



United States Department of Agriculture

Proceedings 19th Central Hardwood Forest Conference

Carbondale, IL

March 10-12, 2014



Forest
Service

Northern
Research Station

General Technical
Report NRS-P-142

December 2014

Proceedings from the 2014 Central Hardwood Forest Conference in Carbondale, IL. The published proceedings include 27 papers and 47 abstracts pertaining to research conducted on biofuels and bioenergy, forest biometrics, forest ecology and physiology, forest economics, forest health including invasive species, forest soils and hydrology, geographic information systems, harvesting and utilization, silviculture, and wildlife management.

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Manuscript received for publication 31 March 2014

Published by:
U.S. FOREST SERVICE
11 CAMPUS BLVD SUITE 200
NEWTOWN SQUARE PA 19073

December 2014

For additional copies:
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Publications Distribution
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Delaware, OH 43015-8640
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19TH

CENTRAL HARDWOOD FOREST CONFERENCE

Proceedings of a Conference held at
Southern Illinois University
Carbondale, Illinois
March 10-12, 2014

Edited by:

John W. Groninger, Southern Illinois University
Eric J. Holzmüller, Southern Illinois University
Clayton K. Nielsen, Southern Illinois University
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Sponsored by:

Southern Illinois University, Department of Forestry
Southern Illinois University, Continuing Education and Outreach
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FOREWORD

The Central Hardwood Forest Conference is a series of biennial meetings dedicated to the sustainability and improvement of Central Hardwood forest ecosystems. The objective of the conference is to bring together forest managers and scientists to discuss research and issues concerning the ecology and management of forests in the Central Hardwood region. The conference has been hosted by different institutes across the region and the 19th Central Hardwood Forest Conference was hosted by the Department of Forestry of Southern Illinois University and the Northern Research Station of the U.S. Forest Service. The conference included presentations pertaining to biofuels and bioenergy, fire, forest biometrics, forest ecology, forest economics, forest health including invasive species, forest soils and hydrology, forest planning, forest recreation, harvesting and utilization, silviculture, and wildlife management. The conference consisted of 49 oral presentations, resulting in 24 papers and 25 presentation abstracts as well as 28 poster presentations and associated abstracts published herein.

The 20th Central Hardwood Forest Conference (2016) will be hosted by the University of Missouri and the Northern Research Station.

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REVIEW PROCEDURES

A blind review process was used in reviewing manuscripts for oral presentations. Each manuscript was peer-reviewed by at least two professionals. Reviews were returned to authors to revise their manuscripts. Revised manuscripts were then submitted to the Northern Research Station, U.S. Forest Service for final editing and publishing. The conference Editorial Committee returned some of the manuscripts to the authors as being more appropriate for other outlets.

ACKNOWLEDGMENTS

The Steering Committee wishes to thank Caitlin McMahon for her assistance in organizing the proceedings, and many thanks to all the professionals who assisted in reviewing manuscripts. The Steering Committee is grateful to Susan Wright and other members of the Northern Research Station communications group for technical editing and publishing of these proceedings. Special thanks to the staff of Continuing Education and Outreach at Southern Illinois University for their assistance in planning and coordinating the 19th Central Hardwood Forest Conference. We also thank Laira Baird, Margaret Bialecki, Josh Nickelson, Bruce Henry, Nathan Hoover, and the following individuals who volunteered as session moderators: Kofi Akamani, Patrick H. Brose, Stacy Clark, Michael Demchick, Daniel C. Dey, Benjamin Dolan, Gary W. Miller, Clay Nielsen, Elizabeth Olson, Logan Park, Ronald Rathfon, Charles M. Ruffner, Michael R. Saunders, Jamie Schuler, Callie Jo Schweitzer, and Curtis Vander Schaaf.

HISTORY OF THE CENTRAL HARDWOOD FOREST CONFERENCE

This Conference is the 19th in a series of biennial meetings that have been hosted by numerous universities and U.S. Forest Service Research Stations in the Central Hardwood Forest region including:

1976 Southern Illinois University
1978 Purdue University
1980 University of Missouri
1982 University of Kentucky
1985 University of Illinois
1987 University of Tennessee
1989 Southern Illinois University and the North Central Forest Experiment Station
1991 Pennsylvania State University and the Northeastern Forest Experiment Station
1993 Purdue University and the North Central Forest Experiment Station
1995 Northeastern Forest Experiment Station and West Virginia University
1997 University of Missouri and the North Central Forest Experiment Station
1999 University of Kentucky and the Southern Research Station
2002 University of Illinois, Urbana-Champaign and the North Central Research Station
2004 Ohio State University, Wooster and the Northeastern Research Station
2006 University of Tennessee, Knoxville and the Southern Research Station
2008 Purdue University and the Northern Research Station
2010 University of Kentucky and the Northern Research Station
2012 West Virginia University and the Northern Research Station

An error in a figure has been reported in a paper published in the 17th Central Hardwoods Conference Proceedings (Fei et al. 2011). This has been corrected in the electronic version online at <http://www.treearch.fs.fed.us/pubs/38058>. For the correction, see page 238 of this paper.

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PLENARY SESSIONS

SILVICULTURAL CONSIDERATIONS FOR MANAGING FIRE-DEPENDENT OAK WOODLAND ECOSYSTEMS

John M. Kabrick, Daniel C. Dey, Carter O. Kinkead, Benjamin O. Knapp, Michael Leahy, Matthew G. Olson, Michael C. Stambaugh, and Aaron P. Stevenson¹

Abstract.—Oak woodlands are characterized by open understories and dense ground flora composed of forbs, grasses, and sedges. They once were common in the western Central Hardwood Forest region and the prairie-forest transition zone where low-intensity fires occurred frequently. In the absence of fire, many of the woodland ecosystems throughout much of this region have succeeded to compositions and structures resembling those of mesophytic forests. Consequently, forest managers are increasingly interested in restoring the structure and composition of oak woodlands by thinning and prescribed burning. Presently, there are few guidelines based upon silvicultural principles for restoring and managing woodland ecosystems. However, many silvicultural concepts, principles, and methods used for managing forests can also be used for managing woodlands, but the application and timing of treatments may differ to meet the objectives of woodland management. In this paper, we summarize findings from a number of studies and offer guidelines for restoring and managing oak woodlands.

INTRODUCTION

Woodlands are natural communities characterized by open to nearly closed canopies of overstory trees, relatively sparse midstory and understory, and dense, species-rich ground flora. In contrast to forests, many of the dominant and codominant trees in the canopy of woodlands have large, spreading crowns (Nelson 2005, Nuzzo 1986, Taft 2009). Shrubs, saplings, and small trees may be present but generally are much less abundant than in a mature forest (Nelson 2005). The relatively open canopy and midstory of woodlands allows sunlight to reach the ground to support a species-rich layer of light-demanding plants dominated by forbs, sedges, and grasses that may be present but seldom are abundant in closed-canopy forests (Kinkead et al. 2013).

Woodlands were once common in the western Central Hardwood Forest region (CHFR) along the prairie-forest ecotone where low-intensity fires occurred periodically (Guyette et al. 2002, Johnson et al. 2009, Kinkead et al. 2013, Taft 2009). These fires largely were set by native people, and fire history studies indicate that before the 1930s the fire frequency was closely related to the population density (Guyette et al. 2002). Fire suppression policies beginning in the region in the 1930s greatly reduced the fire frequency, causing many of the woodland ecosystems throughout much of the CHFR to succeed to compositions and structures resembling those of mesophytic forests (Nowacki and Abrams 2008).

Managers are increasingly interested in restoring the structure and composition of oak (*Quercus* spp. L.) and oak-pine (*Pinus* L.) woodlands by thinning and applying prescribed burning (Dey and

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Hartman 2005, Fule' et al. 2005, Glasgow and Matlack 2007, Hutchinson et al. 2012, Peterson and Reich 2001). Despite the increasing interest, few guidelines based on silvicultural principles are available for restoring and managing woodlands. In this paper, we provide some guidelines for woodland management based upon the contemporary understanding of disturbances and site factors that influence woodland structure and species composition, important metrics for quantifying woodland structure, and silvicultural methods for managing and perpetuating woodlands.

WOODLAND COMPOSITION AND STRUCTURE

In the CHFR, oaks and hickories (*Carya* Nutt.) are the dominant hardwood tree species of woodlands and often occur in association with pines in the Ozark Highlands (Nelson 2005). Numerous ground flora species are considered woodland indicators, particularly graminoids, sedges, and species in the genera *Lespedeza* (Michx.), *Silphium* (L.), *Solidago* (L.), and *Symphyotrichum* (Nees) (Farrington 2010). Most of the woodland indicator species are herbaceous plants that produce flowers and seeds during the summer and are adapted to ecosystems where light penetration is relatively high. These species, often associated with prairie and savanna ecosystems, suggest that stand density has remained sufficiently low to allow sunlight to reach the ground vegetation (Taft 2009).

The Role of Fire

Fire history studies have documented that low-intensity ground fires occurred regularly before European settlement in the forests and woodlands of the CHFR (Guyette et al. 2002) and elsewhere in North America (Ryan et al. 2013). Throughout much of eastern North America and the CHFR, most of these fires were set by humans (Nowacki and Abrams 2008). Surface fires throughout the CHFR occurred frequently; the average fire-return interval ranged from 3 to 19 years but varied over space and time (Guyette et al. 2006). Fire scar data indicated that some locations burned every 2 to 3 years while nearby locations burned once every 20 years or longer because of differences in slope steepness, presence of streams, and occurrence of other features that acted as fire breaks (Guyette et al. 2006). At a single location, there is evidence that the fire-return interval changed over time with changes in the Native American population density (Guyette et al. 2002).

These frequent, low-intensity surface fires are thought to have played an important role in shaping the composition of woodlands (Nelson 2005, Nuzzo 1986, Packard 1993, Taft 2009). Oaks and hickories can persist in association with low-intensity fires because the cotyledons of oak and hickory seedlings remain belowground (Burns and Honkala 1990); if topkilled by fire, the cotyledons remain protected and provide some of the nourishment needed to resprout and remain in the stand. Oak seedlings also establish a large root system at the expense of early shoot growth (Johnson et al. 2009). This larger root system enables oak seedlings to resprout readily after being topkilled. In contrast, maples (*Acer* L.) are disfavored by fire; their cotyledons emerge aboveground and will perish if the seedling is topkilled by a surface fire (Burns and Honkala 1990). Maples also allocate more energy into shoot growth at the expense of root growth, leaving them more vulnerable to mortality following topkill (Burns and Honkala 1990). Grasses, sedges, forbs, and other herbaceous vegetation are also favored by fire compared to vines, shrubs, and other woody vegetation that lose much of their energy reserves if their aboveground tissue is consumed (Brose et al. 2013).

Fire was also thought to have played an important role in reducing stand density and altering forest structure (Nelson 2005, Taft 2009). Shrubs and other small-diameter trees are particularly susceptible to topkill by fire, and frequent, low-intensity fire is thought to have reduced the density of the midstory and understory layers. Surface fire also removes some or all of the leaf litter that can inhibit the germination of many species of grasses, sedges, and forbs. This wide variation in fire-return interval that has been documented to occur within a single location or region is thought to have greatly influenced woodland dynamics (Stambaugh et al. 2007). Frequent fires may have favored the establishment of oak or pine seedlings in woodlands but would prohibit their recruitment. Tree recruitment most likely occurred during fire-free periods (Stambaugh et al. 2007).

Other Disturbances

In addition to fire, disturbances such as severe winds, droughts, ice storms, insects, and diseases also periodically affected woodlands by reducing their density or by altering their species composition (Nelson 2005). As in forests, these disturbances historically contributed to regeneration and stand development patterns. Also, herbivore grazing undoubtedly affected historical woodland structure and composition (Nelson 2005). However, little information is available about how these disturbances shaped woodland character in the past.

Influence of Site Quality

Site quality also affects woodland composition and structure. Dry and nutrient-deficient sites support fewer plant species and lower shrub and understory densities than rich sites (Kabrick et al. 2008). The tree and shrub species that are adapted to these conditions produce litter that dries rapidly and decomposes slowly, allowing them to burn readily. The lower site quality causes trees and shrubs to grow more slowly so that their canopies remain open longer following disturbance (Johnson et al. 2009). Even in the absence of disturbances, the lower shrub and tree seedling and sapling densities allow many of the light-demanding woodland ground flora to persist in the understory (Kinkead 2013). Site classification systems are useful for identifying where site conditions favor the management of woodlands and for predicting how they will respond to management (Kabrick et al. 2008).

WOODLAND SILVICULTURE

Much like forests, woodlands must be managed to sustain their structure and biodiversity and to ensure desirable distribution of woody and herbaceous vegetation in the future (Nelson 2005). Where woodlands are left unmanaged, a dense midstory and understory eventually develops and the overall tree density and canopy cover increases. In addition to the increasing shade caused by the greater stand density and canopy closure, the absence of fire allows a thick layer of leaves to accumulate. Succession to a more shade-tolerant mix of vegetation may occur, particularly in woodlands of moderate to high site quality (Hutchinson et al. 2012). Generally, the intensity of management required to maintain woodland conditions increases with site quality (Nowacki and Abrams 2008). If woodlands are left unmanaged for long time periods, these successional changes may become extremely difficult to reverse due to losses of woodland sedges and grasses and to the additions of shrubs and woody plants that change the nature of the fuels and the response to fire (Nowacki and Abrams 2008).

Table 1.—Some reasons for thinning or burning in forests vs. woodlands

Forests	Woodlands
Thinning	Thinning
Improve stand quality	Reduce density and alter structure
Concentrate growth	Increase light reaching the ground
Utilize trees that will be lost to mortality	Provide growing space to ground flora
Reduce disease and infestation	Reduce disease and infestation
Burning	Burning
Favor desirable tree species during regeneration phase	Favor desirable ground flora species during tending phase
Reduce fuel loading during tending phase	Maintain suitable structure during the tending phase

Because most woodlands in the CHFR have not been managed for many years, a management priority is restoring woodland structure, composition, and function. Once the structure, composition, and function have been restored, it is necessary to plan for regenerating some of the trees in a woodland community. This need arises because some of the trees will succumb to competition-induced mortality as they mature, and others will die of old age or indirectly of injuries suffered through woodland management and other agents causing physiological stress and physical damage. In addition, many woodlands are also capable of producing saw logs, ties, and blocking material. The periodic harvest and sale of these timber products can be used to offset some of the management costs of the thinning and burning that are necessary during later stages. Although a comprehensive management system that includes a plan for regenerating trees is recommended for all woodland management regardless of objectives, this comprehensive plan becomes even more important where producing timber sustainably is an additional objective.

Important Silvicultural Tools, Metrics, and Principles

Many silvicultural concepts, principles, and methods used for managing forests can also be used for managing woodlands. However, the application and timing of treatments may differ to meet the objectives of woodland management (Table 1). Woodland management objectives emphasize conserving the native biodiversity and maintaining habitat rather than optimizing the production of the highest quality wood products. Therefore, treatments are applied at the appropriate frequencies to retain a small number of large trees in the overstory and reduce the number of trees and shrubs in the midstory and understory layers, consume some of the seedlings and leaf litter, and promote the diversity of forbs, sedges, and grasses in the ground layer.

Thinning and prescribed burning may be applied differently in woodlands managed for biodiversity than in forests managed for timber production. In forests, thinning operations are done to improve timber quality and to accelerate the growth of the remaining trees. Although thinning also accelerates the growth of the residual trees in woodlands, it is done primarily to alter stand structure and increase the amount of sunlight reaching the ground to favor light-demanding plant species. In forests, prescribed fire is also used but primarily as a regeneration tool to favor the accumulation of fire-adapted tree seedlings (Brose et al. 2013). Where timber quality is a concern, the application of prescribed fire is generally limited to a short time before or after a regeneration harvest to favor the desirable species. Fire is excluded from the stand during later tending operations to prevent damage to future timber trees (Brose et al. 2013). In woodlands, prescribed fire is used as a tending tool to periodically reduce seedling and sapling density, remove leaf litter, and maintain ground layer species composition.

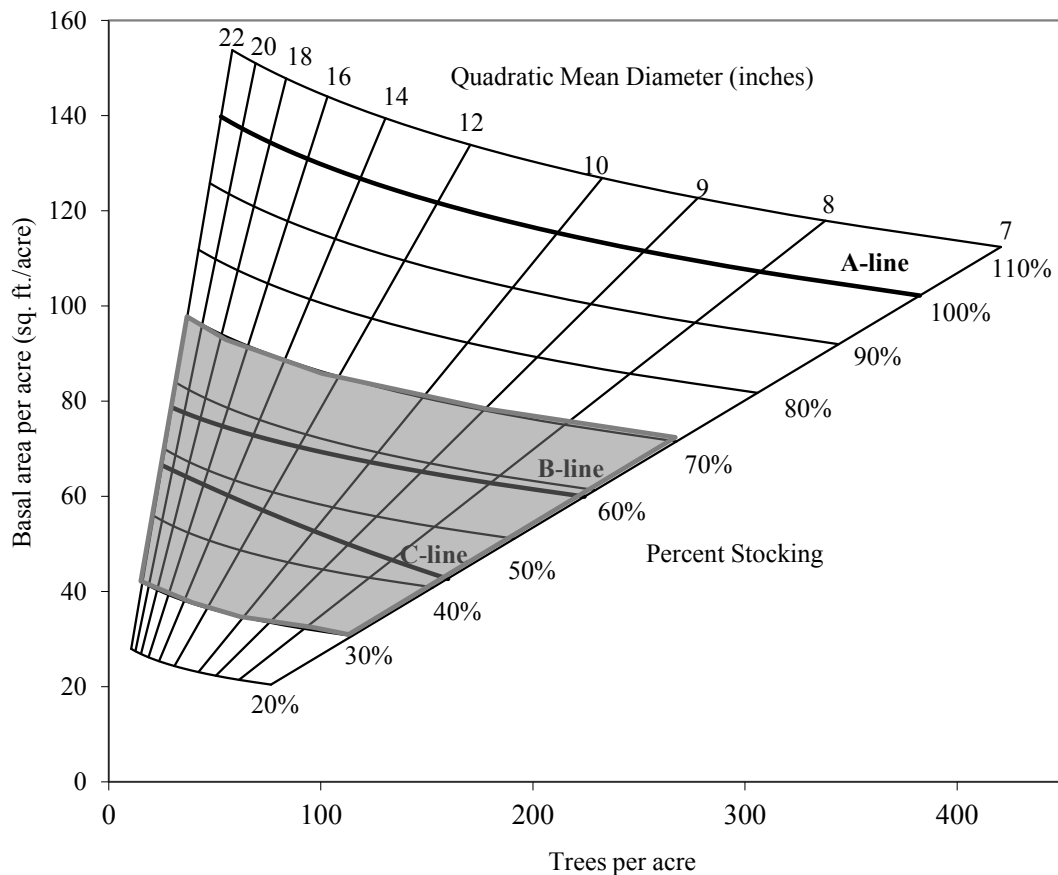


Figure 1.—Stocking chart (Gingrich 1967) for oaks and hickories where the quadratic mean diameter at breast height is >7 inches. Tending operations in woodlands keep the stocking between 30 and 70 percent stocking with closed-canopy woodlands maintained near the B line and open-canopy woodlands maintained below the B line. At stocking levels less than 30 percent, the structure begins to resemble that of a savanna. However, for regenerating forests and woodlands, stocking is reduced below the B line.

Because prescribed burning and thinning are so important to woodland management, drip torches, chainsaws, and herbicides are all important tools for managing contemporary woodlands. However, one of the most important but often neglected tools for managing woodlands is a stocking chart (Fig. 1) (Gingrich 1967). Stocking is an important measure of density related to canopy cover and growing space occupancy (Johnson et al. 2009). Reducing stand density decreases the crown closure and increases the amount of sunlight reaching the ground. Equations are available for estimating canopy closure or light levels as a function of stocking (Blizzard et al. 2013).

Two critical stocking levels are important for managing either forests or woodlands (Fig. 1). The first is the A level, which identifies the average maximum number of trees of a given diameter that can occupy the growing space (Gingrich 1967). The second is the B level, which identifies the average minimum number of trees of a given diameter that can occupy all of the growing space (Gingrich 1967). The B level is an important and biologically meaningful reference because reducing stocking to below the B level ensures that canopy gaps will occur (Johnson et al. 2009). For tending forests, stocking is generally reduced by thinning to the B level and allowing the forest to grow back to the A level (i.e., where full stocking is maintained); for regenerating forests, stocking is reduced to below the B level to allocate growing space to a new cohort of trees. For managing woodlands, stocking is generally maintained at a lower level than for forests. For tending operations in closed-canopy woodlands, stocking levels should be maintained at slightly above or slightly below the B level;

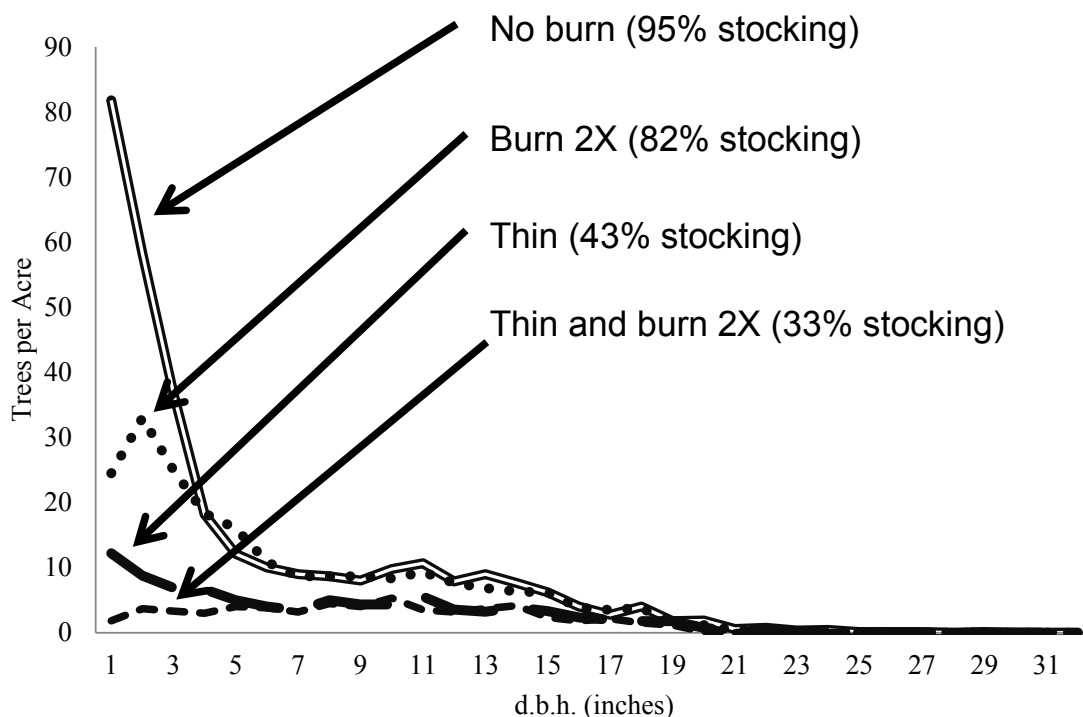


Figure 2.—Diameter distributions coupled with estimated stocking are useful for evaluating the effects of burning and thinning treatments on woodland structure. Greater reductions in stand density can be achieved by thinning or thinning and prescribed burning than by prescribed burning alone. In this example, two prescribed fires were applied 2 years apart (adapted from Kinkead et al. 2013).

for open-canopied woodlands, stocking levels should be maintained below the B level. Although arbitrary, woodland stocking levels during tending operations would remain above 30 percent, the point at which the density would resemble that of a savanna. Much as for forests, woodland regeneration requires stocking to be reduced below the B level.

Stocking estimates can be used along with diameter distributions for assessing the effects of woodland management treatments. For example, in fully stocked stands, two prescribed fires applied 2 years apart will reduce the stem density by 40 percent but will reduce the stocking by only about 15 to 25 percent (Kinkead et al. 2013) by reducing the number of small-diameter stems (Fig. 2). This difference is because trees <5 inches d.b.h. are much more susceptible to topkill or mortality than are larger trees (Dey and Hartman 2005). Greater reductions in stocking require the removal of large trees. For more substantial changes to the stocking, mechanical thinning is needed.

Reducing the stocking only temporarily decreases the growing space occupancy and the canopy cover (Johnson et al. 2009). In the absence of disturbance, stocking increases to the A level over time as the residual trees become larger and new trees become established. During this time the canopy closes and light levels decrease. From the B level, stocking increases on average about 1.5 percent per year but may range from 1 percent per year on low quality sites to 3 percent per year on high quality sites (Dale and Hilt 1989). Consequently, the frequency of thinnings may need to increase as site quality increases to maintain target stocking levels. In woodlands where low stocking is desirable, periodic thinning may be needed, particularly if the stand comprises trees that are too large to be thinned by prescribed fire (Hutchinson et al. 2012).

Table 2.—Historical and contemporary density and diameter of trees > 5 inches d.b.h. of selected Missouri Ecological Sections and Land Type Associates derived from witness tree data from the General Land Office (GLO) and Forest Inventory and Analysis (FIA) databases ^{a,b}

Ecological Section and Land Type Association	Trees per acre		D.b.h. (inches)		Stocking (%)	
	GLO	FIA	GLO	FIA	GLO	FIA
OZ7 Oak Woodland Plains and Hills	50	146	13	9	37	59
OZ7 Oak Woodland/Forest Hills	63	151	13	9	48	60
OZ7 Oak-Pine Hills	66	153	13	9	48	57
OZ7 Pine-Oak Woodland Plains	63	158	13	9	46	57
OZ7 Rugged Hills and Forest Breaks	53	140	13	9	43	53
OZ8 Oak Woodland Plains and Hills	77	138	12	9	48	59
OZ8 Oak Woodland/Forest Hills	57	136	13	10	45	63
OZ8 Oak-Pine Hills	130	138	13	10	99	60
OZ8 Pine-Oak Woodland Plains	65	148	14	9	49	60
OZ9 Oak-Pine Hills	75	138	15	9	59	55
OZ9 Pine-Oak Woodland Plains	63	146	15	9	54	60
OZ9 Rugged Hills and Forest Breaks	111	128	14	9	86	56

^aValues are averages adapted from Hanberry et al. 2014.

^bIn Missouri, General Land Office Surveys were conducted between 1815 and 1850.

Reducing the stocking to below the B level also stimulates the production and development of seedlings, seedling sprouts, and stump sprouts (Larsen et al. 1997). Unless this is desired for regenerating the stand, the high density of sprouts and seedlings may shade the desirable woodland grasses, forbs, and sedges. Under low residual overstory stocking, prescribed fire or some other method of sprout control may need to be applied more frequently than where residual overstory stocking levels remain above the B level, particularly on high quality sites where reproduction establishes more rapidly (Kabrick et al. 2008).

Understanding stocking is also important for interpreting information about the structure of presettlement forests and woodlands. Witness tree data from the General Land Office survey records have been used for identifying restoration target densities for woodlands throughout the CHFR and generally show that there were fewer trees >5 inches d.b.h. per acre than in contemporary forests and woodlands (Table 2, Hanberry et al. 2014). Based on this evidence, many have concluded that forests today have a much greater density than historically and that reducing stand density is a priority for woodland restoration. However, trees at the time of the land surveys were also 1.4 times larger in diameter, and when accounted for using a metric such as stocking, it becomes apparent that even though some woodlands historically had low stocking, many other woodlands and forests had high stocking and at levels comparable to the >5 inch d.b.h. trees in contemporary woodlands and forests (Hanberry et al. 2014). This suggests that the high level of ground flora diversity generally occurring in many woodlands may be due to something other than low overstory stocking such as the removal of leaf litter and the understory vegetation by fire (Kinkead 2013).

Silvicultural Systems

A silvicultural system is a comprehensive plan for regenerating and tending a stand of trees (Nyland 1996). Presently, there are no well-defined silvicultural systems that include a planned series of treatments for regenerating and tending woodlands. Nonetheless, many relevant silvicultural methods are potentially applicable to woodlands as discussed below.

Most of the regeneration methods used in forest management can be applied to woodlands. For example, trees in woodlands can be regenerated with even-age methods including with clearcuts (Roach and Gingrich 1968), the seed tree method where pines are dominant (Nyland 1996), or the shelterwood method (Brose et al. 1999), each with reserves and tended with thinning and prescribed burning. They also can be regenerated with uneven-age methods including with group selection with reserves (Johnson et al. 2009) and tended with thinning and burning. Retaining reserve trees making up 10 to 30 percent stocking is preferable for regenerating woodlands. The reserve trees are retained in the stand to provide habitat for wildlife and partial shade to reduce the density of regeneration that develops after harvesting. Without the reserve trees, the high density of reproduction that develops will eventually shade the woodland ground flora, substantially reducing its cover. If possible, the reserves should be the large-diameter, full-crowned legacy trees that contribute to woodland character.

During the regeneration phase in woodlands, prescribed fire should be excluded until a portion of the reproduction cohort is sufficiently large to escape being topkilled by fire's reintroduction. Here it is important to recognize that in mature woodlands there will only be about 30 to 40 canopy dominant or codominant trees per acre. Thus, managing trees in woodlands is analogous to the silvicultural practice of crop-tree management in which a small number of trees are selected at an early age as the crop trees and are carefully cultured while the vast majority of trees in the stand are left unmanaged. In woodland management, the non-crop trees are subject to removal arbitrarily by burning or deliberately by mechanical thinning.

According to Arthur et al. (2012), the fire-free interval in oak systems should be from 10 to 30 years to allow a sufficient number of trees—30 to 40 per acre—to become large enough to not be topkilled by fire (>6 inches d.b.h.) so that they can recruit into the overstory. If producing marketable timber is also an objective, the fire-free interval may need to be 30 years or longer to allow a critical number of trees to become large enough to not be severely damaged by prescribed fire. These trees are to be treated as the future timber crop so that they can eventually be harvested to offset some of the costs of implementing woodland management treatments. For example, it is reasonable to assume that as trees approach saw log size they become less vulnerable to large losses in value caused by fire scarring of their bark because the damaged outer portion of the saw logs cut from these trees will be removed with the slab wood during milling operations (Marschall et al., in press). If it is assumed that 10 inches d.b.h. is the minimum size threshold for minimizing fire value loss, data from long-term studies of clearcuts conducted on the Sinkin Experimental Forest indicate that more than 20 trees per acre exceed this threshold by age 35 and more than 40 trees per acre exceed this threshold if the stands are thinned at age 20 (Table 3). After the regeneration phase, care must be practiced when reintroducing prescribed burning to prevent mortality of the desired trees or to minimize damage to the future timber crop.

Table 3.—Number of oaks greater than 6 or 10 inches d.b.h. in clearcuts that were not thinned or thinned with a rule-based thinning method to 60 percent stocking in the Missouri Ozarks^a

Stand age (years)	Trees per acre >6 inches d.b.h.		Trees per acre >10 inches d.b.h.	
	Not thinned	Thinned to B level at age 20	Not thinned	Thinned to B level at age 20
15	14	--	0	--
20	58	--	0	--
25	130	128	1	4
30	189	168	6	17
35	214	185	26	42

^aUnpublished data from the Sinkin Experimental Forest.

A 30-year period without prescribed fire may be longer than may have occurred in the pre-European CHFR (Guyette et al. 2006). However, this extended fire-free period is only required if reducing value losses of a future timber crop is a management objective. For woodlands where producing marketable timber for offsetting the costs of woodland management is not an objective, the fire-free period may be as short as 10 years in the CHFR to ensure sufficient recruitment (Arthur et al. 2012).

Reducing the stocking to about 10 to 30 percent during the regeneration phase is important for ensuring that the reproduction cohort can recruit rapidly. Maintaining greater residual stocking levels will substantially reduce the growth rate of the new cohort of trees, increasing the duration of the fire-free interval needed for allowing sufficient numbers of trees to grow larger than the threshold diameters identified above. Reducing the stocking to below 10 percent would increase the growth rate of the new cohort but would leave few large trees in the stand for habitat and for refuge for the ground flora.

Because of uncertainty in fire behavior, silvicultural practices using area regulation are better suited for managing woodlands (Fig. 3). With area regulation, specific stands or land units of the woodland are selected for regeneration or tending. For those selected for regeneration, prescribed fire can be excluded from stands or land units with fire lines, roads, or natural fire breaks to protect the seedlings and allow for recruitment. After a sufficient number of trees have recruited and are no longer in danger of being topkilled or severely damaged, fire can be reintroduced along with other tending methods. Unless the fire-free interval is longer than 10 years, it may be exceptionally difficult to manage woodlands using single-tree selection because this method requires the continuous establishment and recruitment of seedlings and small trees that are vulnerable to topkill by fire. In general, the fire-free interval will need to increase as residual stocking increases, because higher stocking will slow the growth of the regeneration cohort, lengthening the time it takes for desirable regeneration to recruit into size classes less vulnerable to fire damage or topkill.

Longer rotations may be used in woodlands than in forests. Rotations of 100 years are commonly used in hardwood forest management for optimizing the sustained production of timber. However, a longer rotation can be used for managing long-lived species where timber production is not a primary objective. For example, in the Ouachita National Forest, rotations for shortleaf pine-bluestem woodlands were extended from 70 to 120 years for red cockaded woodpecker (*Picoides borealis*) recovery or 160 years for old-growth restoration (Hedrick et al. 2007). Extending the rotation means that woodlands can remain in the tending phase longer where they can be maintained with prescribed fire for more of the rotation.

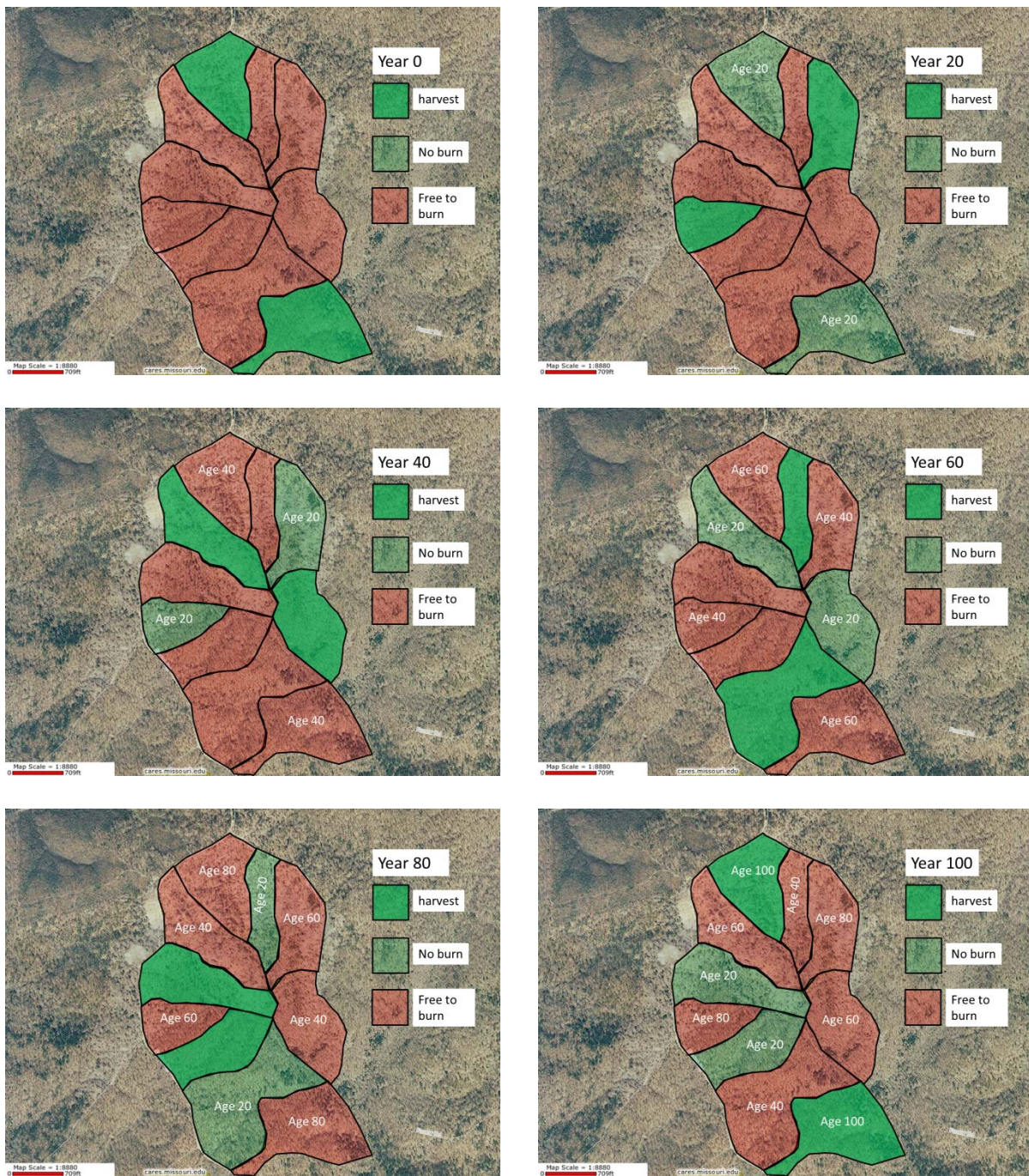


Figure 3.—Area regulation can be used to manage a mosaic of structural states and age classes across an administrative compartment or large woodland tract. In this example, a 200-acre area representing an administrative compartment is subdivided into 10 equal sized fire management units. The boundaries follow old skid trails to facilitate timber removal and fire line maintenance; fire lines elsewhere can be created and maintained with a blower. Regeneration harvests that can include clearcutting with reserves, seed tree method with reserves, or irregular shelterwoods occur in selected fire management units totaling 20 percent of the compartment land area during each 20-year re-entry. Fire is excluded from the regenerated areas for at least 30 years to allow for recruitment and to minimize fire damage to a future timber crop. Other configurations can be generated by changing the rotation age, the re-entry period, and the assumptions about the duration of the fire-free period required for recruitment.

SUMMARY AND CONCLUSIONS

Much like forests, woodlands in the CHFR must be managed to sustain their structure and biodiversity and to ensure desirable distribution of woody and herbaceous vegetation in the future. Many of the tending and regeneration methods used for managing forests can be adapted for managing woodlands although there may be differences in the timing and purpose of treatments. The concept of stocking is useful for managing woodland density and tracking woodland response to management treatments. It generally is easier to use area-regulation methods such as those applied in even-age systems or with the group selection method in uneven-age systems rather than methods using structure control such as in single-tree selection. With area-regulation methods, a stand or management unit can be divided into two phases: the tending phase and the regeneration phase. During the tending phase, prescribed fire can be applied with or without mechanical thinning methods to achieve desirable structure and composition. During the regeneration phase, fire can be excluded from the stand or management unit until a critical number of trees in the regeneration cohort are sufficiently large to escape either topkill or damage so that they can recruit into the overstory.

ACKNOWLEDGMENTS

The authors gratefully acknowledge the two anonymous reviewers whose comments greatly improved this manuscript and all of the woodland managers for their ideas and pioneering work in woodland management.

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FUTURES PROJECT ANTICIPATES CHANGES AND CHALLENGES FACING FORESTS OF THE NORTHERN UNITED STATES

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Abstract

The Northern Forest Futures Project aims to reveal how today's trends and choices are likely to change the future forest landscape in the northeastern and midwestern United States. The research is focused on the 20-state quadrant bounded by Maine, Maryland, Missouri, and Minnesota. This area, which encompasses most of the Central Hardwood Forest region, is the most heavily forested and most densely populated region in the Nation (Shifley et al. 2012). The Northern Forest Futures Project adds detail and context (e.g., Bowker and Askew 2013, Cordell et al. 2012, U.S. Forest Service 2014) to national projections of forest conditions conducted as part of the recent Resources Planning Act assessment (U.S. Forest Service 2012). Analyses for northern forests explore projected conditions from 2010 to 2060 for seven scenarios with differing assumptions about changes in population, land use, harvest removals, and climate. Results suggest that differences in the trajectory of forest change attributable to alternative climate scenarios will become apparent in about 2050. Additionally, other anthropogenic disturbance factors associated with shifts in land use, population change, management practices, and the inadvertent transport of invasive organisms will continue to greatly influence future trajectories of forest change. Some urgent northern forest management issues include: (1) limited structural forest diversity due to age classes clustered in the 40- to 80-year range; (2) decline in forest area in response to increasing urban land area; (3) management of urban trees and forests to improve quality of life for urban residents; (4) damage from invasive species; (5) decline in rates of wood volume growth and carbon sequestration; (6) low level of management of private forests; and (7) barriers to management for nontimber objectives.

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FOREST PLANNING

RESTORATION FOR THE FUTURE: SETTING ENDPOINTS AND TARGETS AND SELECTING INDICATORS OF PROGRESS AND SUCCESS

Daniel C. Dey, Callie Jo Schweitzer, and John M. Kabrick¹

Abstract

Setting endpoints and targets in forest restoration is a complicated task that is best accomplished in cooperative partnerships that account for the ecology of the system, production of desired ecosystem goods and services, economics and well-being of society, and future environments. Clearly written and quantitative endpoints and intermediary targets need to be defined to manage restoration of ecosystem structure, composition, function, and production. Selecting indicators of key ecosystem attributes that are linked to endpoint and target condition, function, sustainability, health, integrity, resilience, and production is important to monitoring restoration success. Indicators are used to track ecosystem trajectory, assess progress toward achieving endpoints and targets, adapt management, and communicate with external publics. Reference sites can be used to help set endpoints and targets with caution. Other science-based ecosystem models or management tools are available to help quantify intermediate targets and endpoints. Continued work to better understand historic ecosystem conditions is fundamental to assessing change, extent of damage, and restoration potential. A hierarchy of forest plans, from regional and landscape to site specific, are useful for defining endpoints, targets, and indicators at appropriate ecological scales and to consider populations, ecosystem function, and socio-economic factors that operate at a variety of scales. The endpoint of restoration is the transition to ecosystem management and sustainability of the desired outcomes and states. This will require continued active management in most cases. Full details of this concept can be found in Dey and Schweitzer 2014.

Literature Cited

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

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A FRAMEWORK FOR ASSESSING CLIMATE CHANGE VULNERABILITY AND IDENTIFYING ADAPTATION RESPONSES IN THE CENTRAL HARDWOODS REGION

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ABSTRACT

Introduction

The Central Hardwood region contains a mosaic of forests, woodlands, savannas, and other ecosystems that will increasingly be affected by a changing climate over the next century. Understanding potential impacts is important to sustaining healthy forests under changing conditions. The objectives of the Climate Change Response Framework (forestadaptation.org) are to develop partnerships among the science and management communities, assess climate change vulnerabilities, and provide tools to integrate climate change information into forest management.

Methods

Two ecoregional projects covering 71 million acres across the Central Hardwood region brought together numerous organizations in a collaborative effort to address climate change. Vulnerability to climate change under a range of future climate scenarios was assessed for 18 forest ecosystems in the Central Hardwood and Central Appalachian project areas. A panel of scientists and managers with expertise in forest ecosystems considered relevant literature and ecological principles together with impact model results to rate ecosystem vulnerability. This expert elicitation method was fundamental to incorporating the local knowledge and experience of the panelists in evaluating vulnerabilities.

Results and Discussion

Panelists first identified potential impacts on ecosystem drivers and stressors. Data from two downscaled climate models bracketing a range of potential futures projected average temperatures to increase by 2 to 8 °F by the end of the century. Although model results for precipitation differed by region, they generally projected decreases in precipitation in summer or fall. Decreased precipitation combined with increased temperatures could potentially decrease soil moisture availability in some areas. These changes have the potential to affect other ecosystem processes such as wildfire dynamics and soil erosion, or increase susceptibility or exposure to insect pests or invasive species.

Potential impacts to tree species were compared and contrasted by the panelists using the Linkages, Landis Pro, and Climate Change Tree Atlas models. All three impact models projected a potential decrease in species having ranges largely north of the region, such as American beech (*Fagus grandifolia*) and sugar maple (*Acer saccharum*), and species currently existing as glacial relicts in the region, such as red spruce (*Picea rubens*) and eastern hemlock (*Tsuga canadensis*). All three impact

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models projected a potential increase in suitability for species having ranges largely south and west of the region, such as shortleaf pine (*Pinus echinata*) and loblolly pines (*Pinus taeda*). Mature trees that are projected to decline may experience slowed growth and reduced productivity but can potentially persist for many years after conditions become unsuitable. Climate change impacts on forest ecosystems may be observed sooner in early growth stages, especially germination and establishment. These impacts will ultimately lead to shifts in species composition within and among ecosystems.

Panelists identified attributes (both positive and negative) for each forest ecosystem related to its ability to cope with climate impacts, collectively known as adaptive capacity. These attributes were based on the current condition of the system given past and current management regimes, with no consideration of potential adaptation actions. Major attributes identified included species diversity, the range of landforms on which the ecosystem could exist, and the ability to withstand or bounce back after a disturbance. Current or past management was also identified as contributing to adaptive capacity in some cases. Ecosystems where past management reduced species, age, or genetic diversity were perceived as having lower adaptive capacity. Ecosystems where current fire or flood regimes differed dramatically from historic regimes were also perceived as having lower adaptive capacity. Management was perceived as increasing adaptive capacity if steps are currently being taken to restore natural ecosystem processes.

Panelists evaluated potential impacts on a forest ecosystem using a continuous scale from positive to negative, generally based on the overall number of positive versus negative impacts on drivers, stressors, and dominant species. Adaptive capacity was also evaluated on a similar scale from low to high. Across both project areas, vulnerability ratings ranged from low to high. Ecosystems dominated by oak species were generally rated low vulnerability, largely due to moderate impacts and high adaptive capacity. Northern and high-elevation ecosystems were generally rated high vulnerability, largely due to high impacts and low adaptive capacity.

Summary

As an increasing amount of relevant scientific information on forest vulnerability to climatic change becomes available, managers require ways to incorporate these broad concepts into forest management plans. We have developed “Forest Adaptation Resources: Climate Change Tools and Approaches for Land Managers” (Swanston and Janowiak 2012) to provide a structured approach for translating broad adaptation strategies into specific management actions and silvicultural practices. These resources, which include an adaptation workbook, are currently being used in collaboration with a number of natural resource managers to develop projects that implement a diversity of adaptation actions while also meeting manager-identified goals.

Literature Cited

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FOREST INVASIVE ADAPTIVE MANAGEMENT ON NATIONAL WILDLIFE REFUGE LANDS IN THE CENTRAL HARDWOOD REGION

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Abstract.—Approximately 2.4 million acres of National Wildlife Refuge System lands are impacted by invasive plants and are the primary challenge for habitat management in the Central Hardwood region. In 2011, biologists, managers, and contracted support staff from six national wildlife refuges in southern Indiana, Illinois, and Missouri developed an adaptive management project to control 42 forest-adapted invasive plant species on refuge lands. Structured decisionmaking was used to identify and refine the management problem, objectives, and alternative management actions, and to assess consequences and tradeoffs among selected management alternatives. Objective hierarchies and an influence diagram were developed to link our monitoring and objectives at two scales, the refuge scale and a management grid scale (1 ha). The project formalized a step-by-step process for prioritizing actions at the refuge scale and for applying management actions at the grid scale. Both inventory and monitoring has provided a feedback loop to inform future management. The grid scale model has allowed formal learning about the effectiveness of prior and ongoing management actions. We demonstrate the approach using data collected from Muscatatuck National Wildlife Refuge (30.9 km²) during 2011–2013.

INTRODUCTION

Invasive plant species (IPS), which affect approximately 2.4 million acres of U.S. Fish and Wildlife Service (USFWS) National Wildlife Refuge System (NWRS) lands, are ecologically and economically costly (Leung et al. 2002). As mandated by the NWRS Improvement Act of 1997 and subsequent policy, the NWRS is to be managed so as to ensure maintenance of the biological integrity, diversity, and environmental health (BIDEH). The BIDEH policy states that where it is feasible and supports the refuge purposes, the NWRS will be managed for historic conditions that were present prior to substantial human-related changes to the landscape. Most refuges report that IPS interfere with their wildlife management objectives (U.S. Fish and Wildlife Service 2003), but control with cost-effective and publicly acceptable methods is a challenge. Refuge managers rank IPS the highest threat to the NWRS, scoring almost double that of any other threat. Costs to combat IPS, including staff and resources for control, are exponentially increasing each year. In fiscal year 2011, the NWRS spent \$15.8 million directly on IPS management (U.S. Fish and Wildlife Service 2013). Further, many federally-listed threatened and endangered species are increasingly impacted by exotic species. Although considered top priority for most National Wildlife Refuge (NWR) managers, there is limited funding and staff to control established infestations or prevent new infestations.

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National Wildlife Refuge managers in the Central Hardwood region do not have a standardized IPS control and monitoring program. They need cost-effective tools to properly plan, prioritize, manage, monitor, and understand IPS infestations. Structured decisionmaking (SDM) and adaptive management (AM) are rarely used to design and implement management strategies for IPS (Blomquist et al. 2010, Bogich and Shea 2008). Based in decision theory and risk analysis, SDM is a carefully organized analysis of problems in order to reach decisions that are focused clearly on achieving fundamental objectives (Hammond et al. 1999). Adaptive management, as framed within the context of SDM, uses flexible decisionmaking that can be adjusted to reduce uncertainties as outcomes from management actions and other events become better understood (Williams et al. 2007). Often in the face of temporal, budgetary, and personnel constraints, managers plan their management strategies with little *a priori* information as to the nature of infestations and have little information as to the effectiveness of management actions. This dilemma is difficult to overcome, as many managers feel the need to use the limited resources to control IPS rather than using those resources to manage IPS (e.g., inventories). However, effective IPS management depends on reliable vegetation monitoring data, and area-wide IPS inventories should be conducted before prioritizing and adopting specific management strategies (Dewey and Andersen 2004).

U.S. Fish and Wildlife Service policies (including BIDEH) mandate managers to adopt integrated pest management (IPM) strategies when managing invasive plant species and inventory, monitoring, and mapping are critical components of a successful IPM program (U.S. Fish and Wildlife Service 2004). Inventories for an IPM strategy should be conducted with the objective of creating accurate species distribution maps that are used to set priorities and select treatments. The most powerful benefit is that monitoring effectiveness is predicted to lower management costs by up to 30 percent (Haight and Polasky 2010). For an inventory or monitoring strategy to be effective, certain critical data must be acquired, including identifying which IPS are present, their location, and their relative abundance (Christensen et al. 2011, Rew and Pokorny 2006). IPS monitoring often focuses on presence only data, however, the collection of true absence information improves the data set and allows the use of traditional statistical methods (Li et al. 2011). Further, true absence data is essential to producing accurate assessments within any species distribution model (Vaclavik and Meentemeyer 2009).

The USFWS Region 3 (R3) refuges in the Central Hardwood region (Illinois, Indiana, and Missouri) that are primarily forested include Big Oaks NWR (IN), Crab Orchard NWR (IL), Cypress Creek NWR (IL), Mingo NWR (MO), Muscatatuck NWR (IN), and Patoka River NWR (IN). In 2009, these refuges began investigating IPS infestations and identified the need to prioritize management actions. Further, managers and biologists recognized the need for coordination to effectively control multiple forest-adapted IPS. The U.S. Geological Survey (USGS) partnered with the refuges to utilize multi-resolution digital aerial color infrared photographs to map some infestations in the spring and fall when nonnative plants were most likely to be most distinct from surrounding native vegetation.² In 2011, staff from each refuge, R3 Division of Biological Resources, and collaborators from the Chicago Botanic Garden convened for a SDM workshop. The goals of the workshop

² U.S. Geological Survey. 2010. Examining the use of color-infrared digital photography to map non-native invasive plants on two Southern Indiana U.S. Fish and Wildlife Service National Wildlife Refuges. 13 p. Unpublished Administrative Report. On file with: Upper Midwest Environmental Sciences Center, 2630 Fanta Reed Road, La Crosse, WI 54603.

were to formulate and clarify the key components of AM, which included objectives, management alternatives, modeling, and monitoring approach (Gregory and Long 2009, Hammond et al. 1999, Keeney and Raiffa 1993, Nichols and Williams 2006, Williams et al. 2007). Through the SDM process, participants decided to: (1) focus on managing IPS; (2) reduce uncertainty in management effectiveness at two spatial scales (refuge scale and 1-ha grid scale); and (3) link management decisions at the grid scale to success at the refuge scale over 7 years. Participants determined that the 1-ha scale was the most appropriate scale to balance effort and resolution of inventories and was an effective scale for management actions and monitoring. Also, staff at most refuges would need 7 years to conduct inventories, apply management actions, and assess effectiveness. Ultimately, the Forest Invasive Adaptive Management (FIAM) project was developed and is presented here. FIAM incorporates a multi-step approach of inventory, prioritization, pretreatment monitoring, treatment, and effectiveness monitoring. FIAM is nearing completion of the setup phase of AM (*sensu* Williams et al. 2007), and pilot work and development of key components are highlighted. Data collected at Muscatatuck NWR using 0.25 ha grids (50 m by 50 m) is used to demonstrate the inventory and management prioritization. However, the other five participating refuges use a 1 ha (100 m by 100 m) grid system because the resolution was found to be adequate for project objectives and greatly reduced effort required to conduct surveys. The refuge-scale inventory phase guided prioritization and management decisions by providing highly detailed maps of the spatial extent and severity of approximately 40 IPS as well as maps of areas that are not invaded.

OBJECTIVES

The stated mission of the NWRS IPS Program is “Through partnerships, prevent, eliminate, or significantly reduce populations of aquatic and terrestrial IPS throughout the NWRS in order to protect, restore, and enhance native fish and wildlife species and associated healthy ecosystems” (U.S. Fish and Wildlife Service 2003). Goals of the program are to: (1) increase the awareness of IPS; (2) reduce impacts of IPS and more effectively meet fish and wildlife conservation goals; (3) reduce IPS impacts on NWRS neighbors and communities; and (4) promote and support safe and effective IPM techniques to combat IPS.

METHODS

Effective management of established IPS can be described as a general 3-step process.

1. Conduct an inventory to identify and prioritize management locations.
2. Apply management actions to those locations.
3. Monitor managed areas to evaluate effectiveness of the action.

Workshop participants developed objectives hierarchies for the refuge scale (Fig. 1) and grid scale (Fig. 2). The ultimate goal was to control IPS at the refuge scale by using actions at a 1-ha scale. Each objective hierarchy started with the NWRS fundamental objectives (i.e., the ultimate reason we seek to control IPS), and moved downward to means objectives (measurable objectives) and the metric that will be measured. An influence diagram was developed to conceptualize factors impacting our objectives and the predicted importance of management actions at the grid scale (Fig. 3). Many factors were considered when developing the influence diagram, but participants chose to focus on three factors of forest composition for monitoring and modeling. The influence diagram also

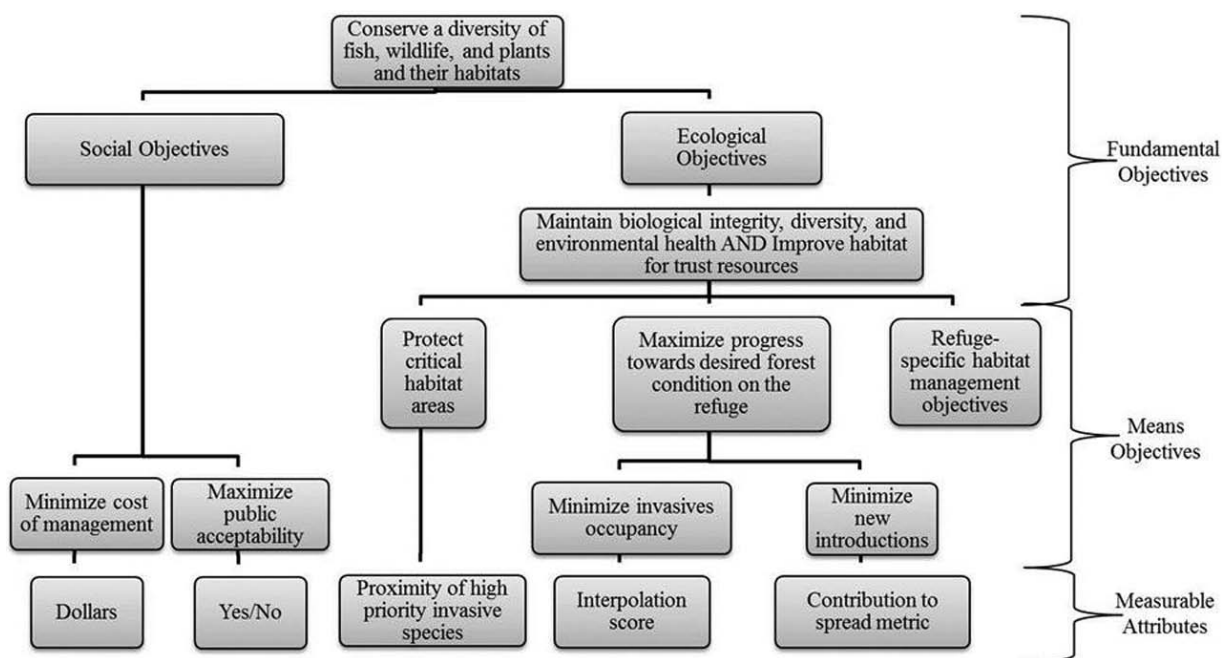


Figure 1.—The refuge-scale fundamental objectives, means objectives, and measurable attributes hierarchy describes the logic and ways to measure success for controlling invasive plant species on National Wildlife Refuges.

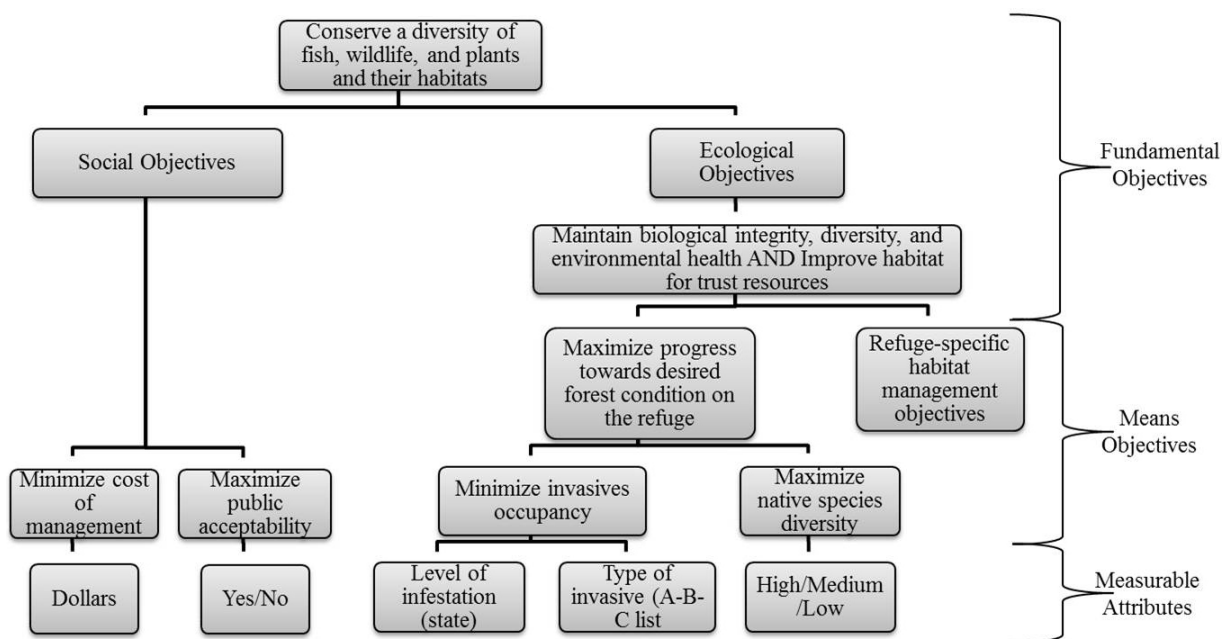


Figure 2.—Grid-scale objectives hierarchy for controlling invasive plant species on National Wildlife Refuges.

provided a visual reference communicating how new information from monitoring would be used to inform future management actions. The primary uncertainty was with how management interacted with the ecology of IPS. Adaptive management at the grid scale was intended to reduce this uncertainty by explicitly learning about the effects of treatments. Additionally, the distribution and potential for spread on the refuges was largely unknown, and the inventory process and prioritization model helped to reduce this uncertainty.

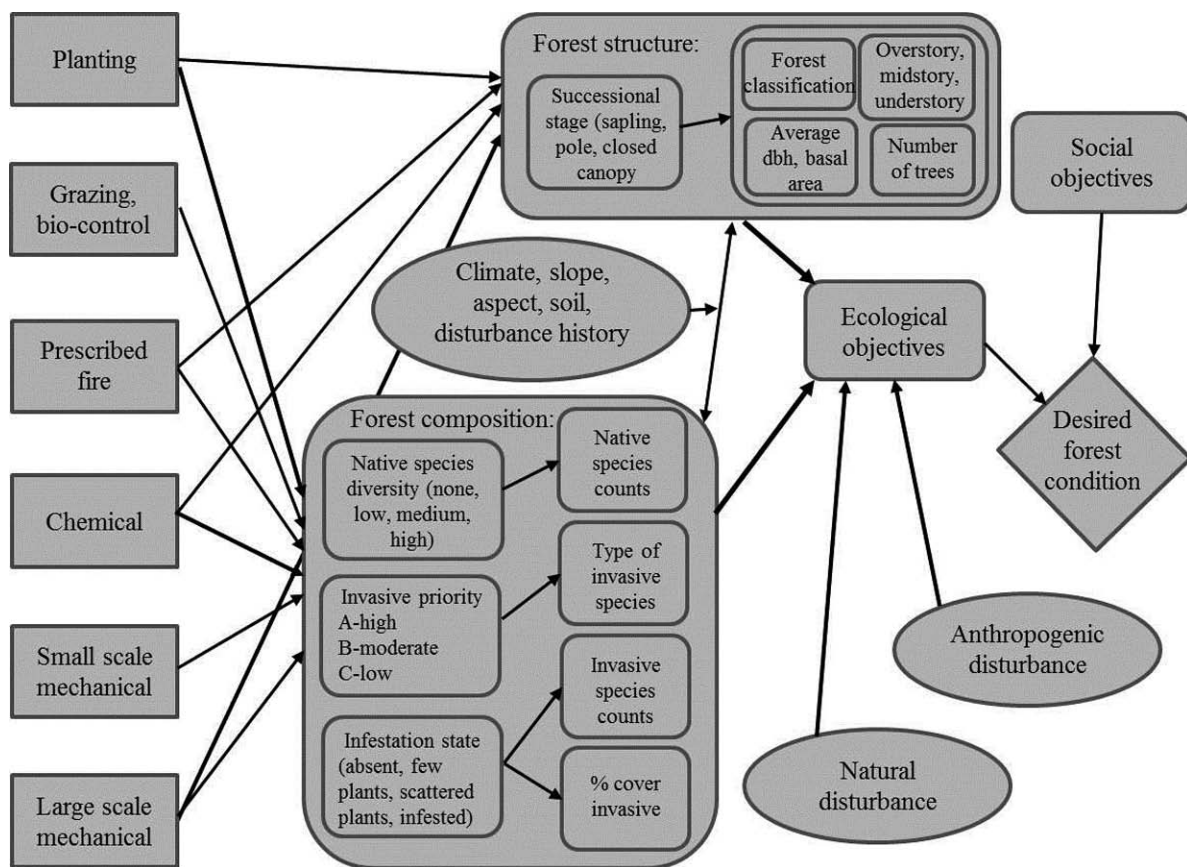


Figure 3.—The influence diagram shows the potential relationships among the management actions, variables outside of the control of managers, and variables controlled by management actions and measured by the refuge staff to assess success at controlling invasive plant species in individual grid cells on National Wildlife Refuges. Key to the influence diagram: diamond = fundamental objectives; rounded rectangles = a variable that is a function of other factors; ovals = uncertainties or chance variables; rectangles = decisions. The thickness of the arrow indicates the strength of the influence, with thicker lines indicating a stronger effect.

Invasive Plant Species Management Steps

Workshop participants formalized a step-by-step process for prioritizing actions at the refuge scale and applying management actions at the grid scale. These steps were built on the basic IPM steps and were useful for developing and revising the objectives hierarchies, monitoring protocols, and linking refuge-scale decisions about where to manage with grid-scale decisions about how to conduct management actions. Basic steps are listed below followed by a more detailed description of the process:

1. Place a grid of 1 ha square cells over the refuge and develop management units for inventory (Fig. 4).
2. Prioritize units for inventory based on site characteristics (e.g., ecological integrity, resistance to invasion, potential spread vectors, disturbance regime, restoration actions, presence of threatened and endangered species, and ease of management).
3. Inventory IPS at each grid across the refuge every 7 years.
4. Create maps based on IPS distribution and state of infestation.
5. Annually prioritize grid cells for management based on: (a) site-specific factors such as growth rate of IPS; (b) spread prioritization factors such as dispersal rate of IPS; (c) proximity to critical areas and potential vectors; d) proximity to public use areas; and e) cost.

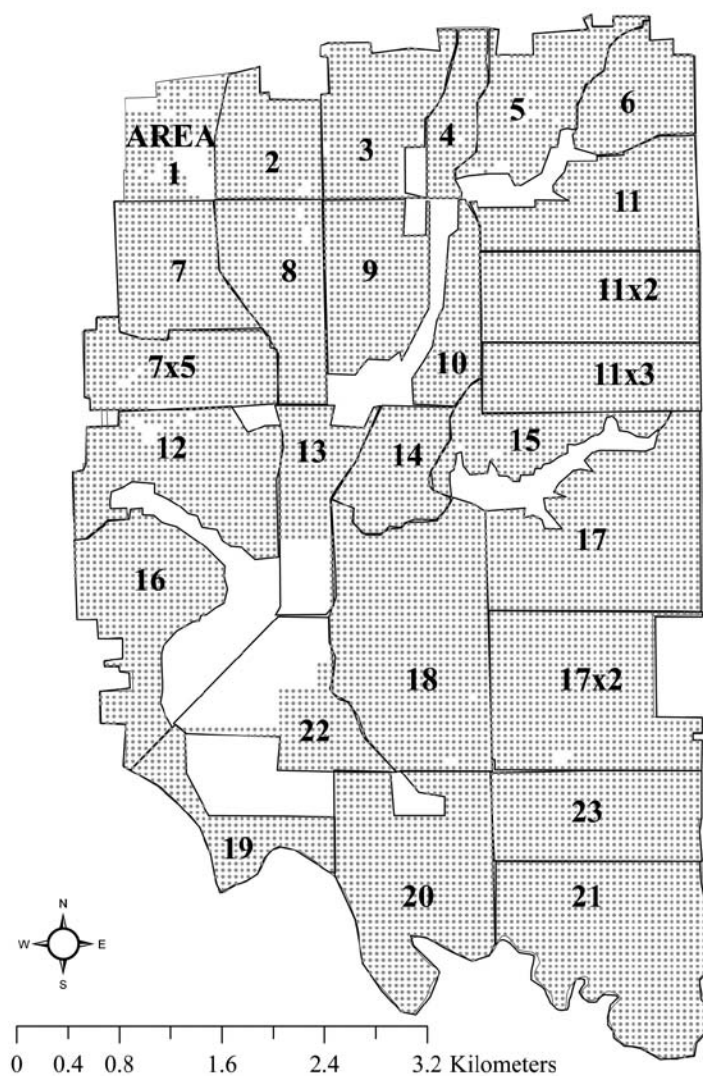


Figure 4.—Invasive plant species management units (areas) and inventory sample points on a 50 m x 50 m (0.25 ha) grid at Muscatatuck National Wildlife Refuge. The grid was overlaid on forested tracts, and most area boundaries consisted of roads, ditches, and wetlands.

6. Decide appropriate management actions.
7. Collect premanagement monitoring data at selected grid cells.
8. Take management action if inventory data and prioritization support action.
9. Collect postmanagement monitoring data to monitor effectiveness.
10. Annually repeat steps 5-8.

Inventory Protocol (Steps 1-3)

The refuge-scale inventory protocol was designed to specifically target project objectives (Lyons et al. 2008, Nichols and Williams 2006) and builds on a forest inventory conducted in 2009-2010. To simplify both inventory and monitoring, workshop participants classified species by their noxiousness (i.e., A-list, B-list, C-list; Table 1) (Morse et al. 2004) and categorized the state of the infestation as:

- Few plants—one plant to a few scattered plants that are not in distinguishable patches.
- Scattered plants—plants have a clumped distribution or are randomly scattered, but not a complete infestation.
- Infested—local area surrounding the sample point completely infested dense stand.

Table 1.—Select invasive plant species (IPS) detected on National Wildlife Refuge (NWR) forests in the Central Hardwood region categorized by level of priority for treatment

Scientific Name	Common Name	Priority rank ^a
<i>Dioscorea bulbifera</i>	air yam	A
<i>Lonicera maackii</i>	Amur honeysuckle	A
<i>Elaeagnus umbellata</i>	autumn olive	A
<i>Dioscorea oppositifolia</i>	Chinese yam	A
<i>Ligustrum vulgare</i>	European privet	A
<i>Alliaria petiolata</i>	garlic mustard	A
<i>Phragmites australis</i>	common reed	A
<i>Berberis thunbergii</i>	Japanese barberry	A
<i>Lonicera japonica</i>	Japanese honeysuckle	A
<i>Polygonum cuspidatum</i>	Japanese knotweed	A
<i>Microstegium vimineum</i>	Japanese stiltgrass	A
<i>Sorghum halepense</i>	Johnsongrass	A
<i>Lonicera morrowii</i>	Morrow's honeysuckle	A
<i>Carduus nutans</i>	nodding plumeless thistle	A
<i>Celastrus orbiculatus</i>	Oriental bittersweet	A
<i>Lythrum salicaria</i>	purple loosestrife	A
<i>Phalaris arundinacea</i>	reed canary grass	A
<i>Lonicera tatarica</i>	Tatarian honeysuckle	A
<i>Euonymus fortunei</i>	winter creeper	A
<i>Populus alba</i>	white poplar	A
<i>Polygonum perfoliatum</i>	Asiatic tearthumb	B
<i>Cirsium vulgare</i>	bull thistle	B
<i>Lysimachia nummularia</i>	creeping jenny	B
<i>Dipsacus laciniatus</i>	cutleaf teasel	B
<i>Heracleum mantegazzianum</i>	giant hogweed	B
<i>Rubus armeniacus</i>	Himalayan blackberry	B
<i>Dipsacus fullonum</i>	Fuller's teasel	B
<i>Humulus japonicus</i>	Japanese hop	B
<i>Pueraria montana</i>	kudzu	B
<i>Rosa multiflora</i>	multiflora rose	B
<i>Paulownia tomentosa</i>	princesstree	B
<i>Lespedeza cuneata</i>	sericea lespedeza	B
<i>Ailanthus altissima</i>	tree-of-heaven	B
<i>Euonymus alatus</i>	burningbush	C
<i>Vinca minor</i>	common periwinkle	C
<i>Sesbania herbacea</i>	coffee weed	C
<i>Glechoma hederacea</i>	ground ivy	C
<i>Clematis terniflora</i>	Japanese virgin's bower	C
<i>Lonicera xbella</i>	showy fly honeysuckle	C
<i>Melilotus officinalis</i>	white sweet clover	C
<i>Salix alba</i>	white willow	C

^a A = high, B = medium, and C = low (Morse et al. 2004).

The refuge-scale inventory protocol was developed and pilot tested at Muscatatuck NWR during the 2011 field season. This protocol was amended from the Federal IPS Mapping Standards (North American Weed Management Association 2002) to more effectively meet AM project objectives. Information from Muscatatuck was used to refine the protocols and was used at other refuges during 2012–2013. The inventory was a rapid assessment of grid cells across the entire refuge based on five major metrics that affect prioritization:

- State of the infestation and type of IPS ranking based on priority for management (Table 1).
- Critical areas metric (distance to special communities, and threatened/endangered species).
- Spread metric (proximity to vectors for spread of IPS including roads, waterways, trails, and refuge boundaries).
- Public acceptability metric (public visibility of infestation).
- Cost metric (determined by logistics to access a grid cell).

At Muscatatuck NWR, ArcGIS 9.3 (Esri, Redlands, CA) geographic information system (GIS) was used to establish grid points with a 50 m spacing (other refuges used 100 m) to create 23 separate sampling areas delineated by wetlands, streams, ditches, roads, and refuge boundaries (Fig. 4). At each point, the IPS were recorded and the state of the infestation was visually estimated for each species. Visibility distance was also recorded as the categorical distance (0-5, 6-10, 11-20, or >20 m) that the observer could reliably identify vegetation. For each grid cell, GIS was used to measure the distance to critical areas, spread vectors, roads, and public use areas. The distance to road, which is correlated with management cost, was used to assess the relative cost to implement a management action.

Prioritization Decision Support (Steps 4-5)

Using a GIS point to raster conversion, a distribution map of autumn olive (*Elaeagnus umbellata*) was created based on survey data at Muscatatuck NWR (Fig. 5), and this procedure was repeated for each target species. If the species was not observed at a point, a 0 (absent) was recorded, therefore, areas that were not invaded were mapped. The procedure provided a layer that defined the approximate boundaries and the level of infestations. Maps of A-list, B-list, and C-list species were used to create a single map for the three priority levels, and the infestation scores were used in the prioritization tool.

Workshop participants developed a decision tool that prioritized management actions based on the state of each grid cell with regard to the prioritization metrics for each of the ecological and social objectives based on inventory data, the relative metric weights, and project budget. This tool was developed further and pilot tested using inventory data from Muscatatuck NWR (unpublished data³). Based on the five metrics from refuge-scale inventory, a management prioritization score was computed for each grid cell using a 5-step process:

1. Site-specific prioritization score assigned based on the probability of changing to another infestation state in the following year (Fig. 6).
2. Spread prioritization score estimated based on infestation state and invasive noxiousness of adjacent cells.

³ Lonsdorf, E.V.; Blomquist, S.M.; Wood, D. Unpublished data. On file with: U.S. Fish and Wildlife Service, Muscatatuck National Wildlife Refuge, 12985 E. U.S. 50, Seymour, IN 47274.

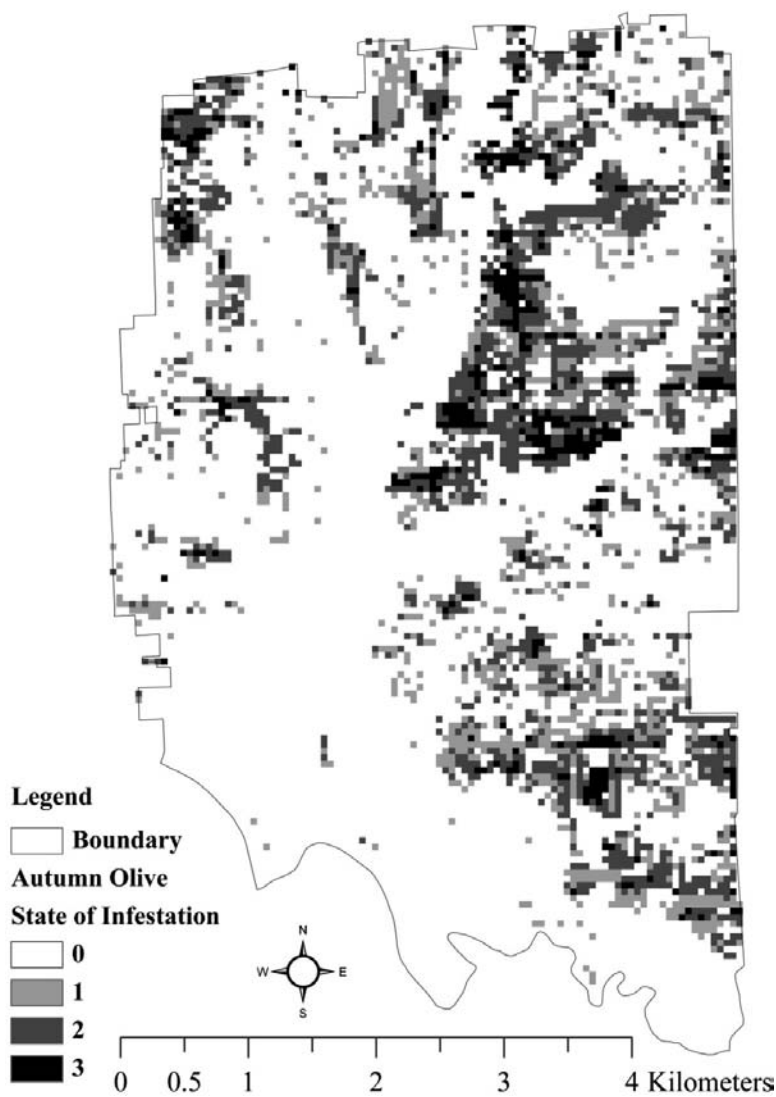


Figure 5.—Autumn olive distribution based on invasive plant species inventory during 2011 at Muscatatuck National Wildlife Refuge. The ordinal scale data for the state of autumn olive infestation were: 0= absent; 1= few plants; 2= scattered plants; 3= infested.

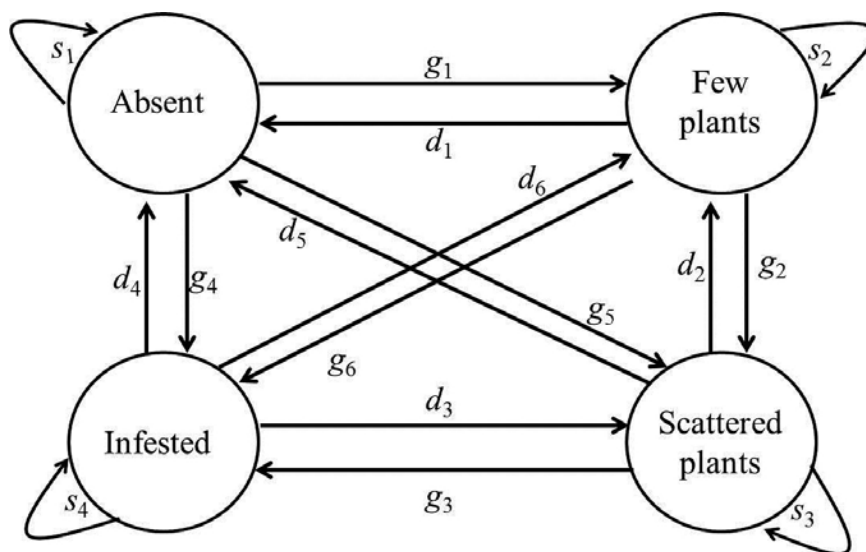


Figure 6.—Transition probabilities for a grid cell to remain in the current state or to change to another state from one year to another. The probabilities of infestation change are to grow (g), decrease (d), or stay (s).

3. Site-specific and spread prioritization score summed and weighted by the distance to spread vectors, critical areas, and public use areas.
4. Management cost estimated for each grid cell based on accessibility.
5. Grid cells selected for management based on the priority score and cost.

Management Actions (Step 6)

Management actions were categorized to reflect a general type of action to simplify the problem for modeling and decisionmaking purposes. Alternative management actions and techniques available for controlling IPS infestations on NWR include:

- No action
- Prescribed fire
- Small-scale mechanical (hand pull, girdle, chain saw)
- Large-scale mechanical (mow, mulch, bulldoze)
- Chemical spray (spot, broadcast)
- Tree planting

Within these categories, the decisionmaker chose the best management practice in each instance based on the species present, accessibility, management restrictions, cost, and other considerations. A challenge for AM projects was to reduce management alternatives to those that will be used at a high enough frequency to promote learning. Thus, categories were simplified based on the frequency of use in forested environments to a final action menu that included: no action, prescribed fire, mechanical, chemical, and planting, as well as selected combinations of these actions.

Grid-Scale Monitoring Protocol (Step 7)

Monitoring at the grid scale was intended to assess the effectiveness of the management actions as well as improve the refuge-scale inventory data. The current protocol collected data on habitat metrics to provide information necessary for state dependent decisionmaking, evaluate management performance, and facilitate improved management through learning. This protocol used a transect-based approach to sample approximately 20 percent of each grid cell selected for management, and thus increased detection of IPS (Chen et al. 2009, unpublished data⁴). Points were sampled on transects premanagement and postmanagement to assess three state-based metrics: (1) state of infestation, (2) invasive noxiousness, and (3) native species diversity in three categories based on species counts.

Management Action Decision Support (Step 8)

State-and-transition models, in which the transitions from one state to the other are unknown, were created to tie treatments to effectiveness monitoring. *A priori* transition probabilities were developed (Fig. 6) for how each category of plant (e.g., noxiousness levels, woody vs. herbaceous) changed with and without management based on staff experience, literature, and agency guidance. These transition

⁴ Blomquist, S.M.; Wood, D. Unpublished data. On file with: U.S. Fish and Wildlife Service, Muscatatuck National Wildlife Refuge, 12985 E. U.S. 50, Seymour, IN 47274.

probabilities represented hypotheses that predicted how each species should respond to management and formalized our primary uncertainty about how our management affected the ecology of IPS. A critical aspect of the AM process is the development of credibility measures for each model, which represent relative beliefs or weights. Learning through AM occurs when each model's prediction is confronted with observations of the system, and these relative belief measures are updated through Bayes theorem (Williams et al. 2007). Stochastic dynamic programming is used to implement this Markov decision process framework (Bogich and Shea 2008). Such state-and-transition models are useful for determining an optimal policy of state-dependent actions and are recommended for land management modeling (Bestelmeyer et al. 2011).

Post-Management Monitoring (Step 9)

Postmanagement monitoring was conducted annually after taking management actions at the grid scale, and followed the same protocol as Step 7. The data on IPS distribution and state was also used to update the prioritization model (Steps 4-5).

DISCUSSION

We presented a systematic approach for IPS control on NWR lands in the Central Hardwood region, but the process can be used for lands managed for natural conditions. Although controlling IPS consumes incredible financial resources for many NWRs, managers rarely track the effectiveness of management actions. The primary reasons are a paucity of common protocols, data management structures, and a perceived lack of resources for monitoring. In most cases, it is unknown if resources used to combat IPS achieve the desired outcome. Before the development of FIAM, refuge managers had few options for prioritizing the control of IPS with various life forms and responses to management techniques. Managers were also challenged with assessing potential for spread beyond current infestations and boundaries. Additionally, the high diversity of habitats throughout much of the Central Hardwood region prevented the use of a single approach to effective IPS management.

The focus of FIAM is on established or recently invaded populations. Our project did not explicitly address early detection rapid response (EDRR) procedures which are effective for preventing IPS establishment and are an essential part of any IPM program (U.S. Fish and Wildlife Service 2004). However, the rapid inventory gave the field crews the opportunity to report EDRR species and give real time reporting of unusual vegetation. For example, during the inventory phase, several IPS were discovered that were unknown to occur on these lands.

Developing the grid system was a compromise between a systematic approach to rapidly inventory NWRs and an appropriate management scale. The 1-ha scale was small enough to put effective management on the ground and large enough to assess the effectiveness of those management actions. FIAM uses a systematic sampling approach to predict and learn about management. The grid-based approach comprehensively samples the management unit regardless of the current distribution, and such systematic approaches are ideal for supporting distribution modeling and long-term monitoring (Stadt et al. 2006).

The explicit structure of AM provided the guidance to elucidate goals, develop hypotheses, identify alternative management actions, and develop procedures to collect relevant data that can be evaluated

and followed by reiteration (Williams et al. 2007). The objective hierarchies are decision analytic tools that were effectively used during the setup phase and can be modified through the implementation of FIAM with learning and shifting priorities. For example, workshop participants decided that forest structure was less influential than forest composition for making management decisions. This helped refine the objectives hierarchies to focus on forest composition and then build decision tools for prioritizing management actions. Ultimately, these hierarchies were useful to link the FIAM project to the NWRS mission, measuring progress toward achieving goals and other factors that may influence success.

This project highlighted the need for modeling capability for AM projects. The prioritization model was effective for deciding the most appropriate locations to focus management efforts and developing predictions about the spread of infestations. Large-scale AM projects with much uncertainty are likely to fail without substantial investment in personnel dedicated to modeling and data analysis.

We have completed pilot testing the inventory and monitoring protocols, but the project remains in setup phase for all refuges as we complete pilot testing during 2014. We anticipate FIAM will be fully implemented, providing feedback to refuge managers and being used for management guidance outside the NWRS by 2015.

FIAM is a good example of the requirements and benefits of developing and implementing the setup phase of an AM project for multiple NWRs. A key component of FIAM development was formalizing priorities for management and predicting effects of management actions used to deal with the high IPS diversity and management. This requirement of AM promoted collaboration and information sharing among refuge biologists and regional biological staff when developing and pilot testing standard protocols. Additionally, collaboration improved data management systems, which tend to be a major hindrance to tracking management actions on many NWRs. An additional key benefit was enhanced overall communication between multiple NWRs across a large region. The 10 management steps were critical to the participants when evaluating and tracking the development status of FIAM through the setup phase. These steps will continue to be used to ensure the project is implemented in an efficient and effective manner.

ACKNOWLEDGMENTS

We thank senior leadership of Region 3 Biological Resources for funding, ongoing support, and helpful comments during project development. Funding for this project was also provided by NWRS Seeds of Success, NWRS Invasives with Volunteers, and NWRS Inventory & Monitoring Initiative. Without the continued support and encouragement of refuge managers this project would not be possible. We thank the dozens of field staff that assisted in collecting data. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. This publication represents the views of the authors and does not necessarily represent the views of the U.S. Fish and Wildlife Service.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

BUILDING SOCIAL-ECOLOGICAL RESILIENCE THROUGH ADAPTIVE COMANAGEMENT IN THE CACHE RIVER WATERSHED OF SOUTHERN ILLINOIS

Kofi Akamani¹

Abstract.—There is growing recognition that the sustainable governance of water resources requires building social-ecological resilience against future surprises. Adaptive comanagement, a distinct institutional mechanism that combines the learning focus of adaptive management with the multilevel linkages of comanagement, has recently emerged as a promising mechanism for building social-ecological resilience. This paper employs the concept of adaptive comanagement to analyze ongoing institutional reforms in the Cache River watershed of southern Illinois. Since the 1970s, efforts have been made to promote collaborative decisionmaking aimed at the restoration of the watershed. However, the current governance system remains vulnerable because little attention has been given to building the capacity of the watershed for learning and adaptation. Adaptive comanagement can contribute to building resilience in the watershed by creating awareness, generating interest, creating opportunities, and building capacity for adaptation.

INTRODUCTION

Over the last few decades, comanagement has attracted a lot of research and policy focus as a promising institutional framework that can integrate the benefits of community-based, market-based, and centralized approaches to resource management (Acheson 2006, Berkes et al. 1989). Yandle (2003: 180) defines comanagement as “a spectrum of institutional arrangements in which management responsibilities are shared between the users (who may or may not be community-based) and government.” In spite of its promise, failures in comanagement programs have been attributed to implementation challenges, such as the lack of political will on the part of governments and their representatives to share power with resource users (Berkes 2010) and the potential for capture by powerful local elite, leading to the entrenchment of pre-existing inequalities (Cinner et al. 2012). Comanagement has also been conceptually critiqued for its inadequate recognition of complexity and the need for learning (Berkes 2004).

Comanagement is evolving into adaptive comanagement, a distinct institutional mechanism that integrates the learning focus of adaptive management with the multilevel linkages of comanagement (Berkes 2009). Adaptive comanagement provides a framework within which different stakeholders across multiple scales are connected through networks from local users to international bodies in an ongoing process of learning and responding to changes in social-ecological systems (Olsson et al. 2004). The field of water resources management, for instance, is increasingly focusing on management approaches that promote learning as a means of dealing with complexity and uncertainties (Akamani and Wilson 2011, Bark et al. 2012, Pahl-Wostl 2007). There is a need for knowledge on strategies that can promote a successful transition to adaptive approaches to resource management (Olsson et al. 2008).

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This paper draws from perspectives on social-ecological resilience and adaptive comanagement to analyze ongoing institutional reforms in the Cache River watershed of southern Illinois. Building on previous works that have assessed the resilience status of governance institutions in the watershed (Akamani 2013) and the challenges in the transition to adaptive governance in the watershed (Akamani 2014), this paper proposes adaptive comanagement as a mechanism for building social-ecological resilience in the watershed. Key concepts are clarified and the case of the Cache River watershed is presented and briefly analyzed. Ways in which adaptive comanagement could inform policies and strategies for building resilience in the watershed, including creating awareness, cultivating interest, creating opportunities, and building capacities for change, are also discussed.

SOCIAL-ECOLOGICAL RESILIENCE AND ADAPTIVE COMANAGEMENT

Anderies et al. (2004: 3) define a social-ecological system as “an ecological system intricately linked with and affected by one or more social systems.” Such linked social-ecological systems exhibit attributes of complex adaptive systems, such as cross-scale interactions, surprise, nonlinearity, and self-organization (Folke 2007, Liu et al. 2007, Pahl-Wostl 2007). The sustainable management of dynamic social-ecological systems requires approaches that build resilience rather than maximizing benefits in the short run (Folke et al. 2011). Social-ecological resilience has three interrelated meanings: the magnitude of disturbance that the system can absorb while remaining in a given state; the capacity of the system for self-organization; and the capacity of the system for learning and adaptation to change (Folke et al. 2002).

The use of centralized, expert-driven approaches to resource management is inadequate for managing complex social-ecological systems (Westley et al. 2011). Olsson et al. (2004: 75) define adaptive comanagement as “flexible community-based systems of resource management tailored to specific places and situations and supported by, and working with, various organizations at different levels.” Adaptive comanagement, an innovative institutional mechanism for managing complex social-ecological systems, has emerged out of the conceptual integration of comanagement and adaptive management (Plummer 2009). Adaptive management is a management approach that recognizes the inherent uncertainty and unpredictability of social-ecological systems and aims at increasing knowledge and reducing uncertainty through constant monitoring (Allen and Gunderson 2011). However, the implementation of adaptive management programs frequently fails due to the absence of an enabling institutional framework (Walker et al. 2004). Through the integration of comanagement and adaptive management, adaptive comanagement is more socially responsive to the aspirations of resource users than adaptive management and focuses more on learning and adaptation than comanagement (Berkes 2009). These attributes make adaptive comanagement a promising mechanism for building resilience in social-ecological systems (Olsson et al. 2004, Walker et al. 2006).

CASE STUDY OF THE CACHE RIVER WATERSHED

The Cache River watershed covers an area of 1,944 square miles near the confluence of the Mississippi and Ohio Rivers in southern Illinois. Over the last two centuries, multiple drivers of change have resulted in significant modification of the watershed (Duram et al. 2004). For instance, the construction of the Post Creek Cutoff in 1915 divided the watershed into two separate drainage basins, the Upper Cache basin and the Lower Cache basin. The 1940s and 1950s saw further

modification of the Cache River, including channelization, dredging, and construction of levees, reservoirs, and water control structures (Cache River Watershed Resource Planning Committee 1995).

Public recognition of ecological crisis in the watershed triggered institutional reforms in the 1970s. The purchase of land by the Illinois Department of Natural Resources (IDNR) in 1970 subsequently led to the formation of the Cache River Joint Venture Partnership (JVP) in 1991. Membership of the JVP currently comprises the IDNR, The Nature Conservancy (TNC), the U.S. Fish and Wildlife Service (USFWS), Ducks Unlimited, and the U.S. Department of Agriculture Natural Resource Conservation Service (NRCS). The goal of the JVP is to protect and restore 60,000 acres of wetlands along the Cache River (Cache River Ecosystem Partnership 1999). The reconnection of the Lower Cache basin and the Upper Cache basin is considered another primary goal of the JVP (Davenport et al. 2010). Between 1993 and 1995, the NRCS and TNC led the preparation of the Cache River Watershed Resource Plan with funding from the U.S. Environmental Protection Agency. The plan was expected to help various organizations in the watershed secure funds for conservation efforts in the watershed (Cache River Watershed Planning Committee 1995).

Assessment of Institutional Reforms

In all, progress appears to have been made in promoting collaboration among various organizations to address the ecological challenges in the watershed. However, from an adaptive comanagement perspective, a number of shortfalls can be identified in the goals and underlying assumptions, as well as in the institutional mechanisms for decisionmaking and implementation in the watershed.

Resource Management Goals and Assumptions

The goals for the current management of the Cache River watershed appear too narrow and static. As such, they do not reflect the need for integrated and adaptive water management. Lant (2003) attributes this outcome to the NRCS and TNC who, as leaders of the planning process, limited the scope of the plan to resource management issues, neglecting socio-economic concerns in the watershed, such as poverty and population decline. Studies show that members of the JVP, such as the NRCS, USFWS, IDNR, and TNC, have been quite successful in using the plan to obtain funding for various conservation efforts, such as wetlands reclamation and soil protection (Adams et al. 2005). However, community members are concerned about the adverse socio-economic impacts of land acquisition and restoration programs (Davenport et al. 2010). Besides its narrow scope, the Cache River Watershed Resource Plan also failed to explicitly recognize the uncertainties in the restoration process and the need to proceed through experimentation and learning.

Planning and Implementation Mechanisms

The institutional mechanisms for the preparation and implementation of the Cache River Watershed Resource Plan failed to offer adequate representation of the various segments of society and provided limited opportunities for the integration of community values and local knowledge. One shortfall of the process is that a 25-member planning committee that was established to represent the five counties in the watershed was mostly composed of large-scale commercial farmers and did not adequately represent the diverse interests of communities in the watershed (Lant 2003). Also, a 15-member technical committee, composed of expert representatives from various organizations including TNC and NRCS, was more powerful in orienting the plan toward ecological issues that did not include the broader socio-economic issues in the region (Adams et al. 2005). As a consequence,

the content of the plan reflected the views of expert scientists affiliated with the various organizations rather than the views of all stakeholders. Furthermore, although the planning committee held four public meetings and sponsored a telephone survey to ascertain the concerns of residents in the watershed, the planning process did not explicitly resolve conflicts in stakeholder perceptions that emerged from the survey (Kraft and Penberthy 2000). Following the adoption of the plan, there have been limited opportunities for community input in the implementation process (Adams et al. 2005). Community members are largely unaware and unsupportive of restoration efforts in the watershed (Davenport et al. 2010). As such, the Cache River Watershed Resource Plan has been critiqued for its lack of legitimacy (Adams et al. 2005) as well as its potential contribution to the erosion of social capital in the watershed (Lant 2003).

APPLYING ADAPTIVE COMANAGEMENT TO THE CACHE RIVER WATERSHED

The analysis of institutional reforms in the Cache River watershed has shown that while progress has been made in the adoption of a collaborative approach to the management of the watershed, further institutional interventions are needed to build a more resilient governance regime. This section discusses various ways in which adaptive comanagement could inform strategies for building resilience in the watershed. The discussion is structured around four key themes on the conditions for social-ecological resilience: awareness about social-ecological complexity; interest and motivation to act; availability of opportunities for change; and capacity requirements (Gunderson et al. 2006, McLain and Lee 1996, Olsson et al. 2004).

Enhancing Awareness

A key challenge in the adoption of adaptive management policies is the reluctance of decisionmakers and other powerful stakeholders to embrace complexity and resilience thinking (Walters 2007). The ability to successfully adapt to social-ecological change requires knowledge and information on social values and the ecological system as well as the uncertainties in their interaction (Dietz et al. 2003, Olsson and Folke 2001). Adaptive comanagement can contribute to awareness about social-ecological complexity in three ways: integration of local and scientific knowledge, promotion of social learning, and emphasis on monitoring and assessments.

First, adaptive comanagement promotes the integration of different types of learning approaches, both experiential and experimental, to understand social-ecological systems (Armitage et al. 2009). Through the use of both science and indigenous knowledge, adaptive comanagement opens up possibilities for indigenous people and other local communities to be involved in the coproduction of locally relevant knowledge as well as linking such local knowledge directly into the decisionmaking process (Berkes 2009). Second, the promotion of social learning through iterative processes of learning by doing is another defining feature of adaptive comanagement. Reed et al. (2010: 6) define social learning as “a change in understanding that goes beyond the individual to become situated within wider social units or communities of practice through social interactions between actors within social networks.” Social learning can contribute to the accumulation of collective social memory, comprising historical experiences, knowledge, values, and institutions that could be drawn upon for future responses to change (Adger et al. 2005, Olsson et al. 2004). Finally, adaptive comanagement can contribute to awareness about social-ecological complexity through its emphasis on monitoring and assessment of environmental outcomes (Armitage et al. 2009). The development of indicators at

appropriate scales for assessing the dynamics of social-ecological systems is critical to providing early warning information about threats to social-ecological resilience and informing appropriate policy responses that preserve diversity and adaptive capacity (Folke et al. 2002).

In the case of the Cache River watershed, there is the need for broadening the scope of knowledge used in decisionmaking through an explicit recognition of local ecological knowledge. Olsson et al. (2004) illustrate the successful integration of local observations and scientific knowledge in the management of Lack Rachen watershed in Sweden within an adaptive-comanagement institutional framework. Social learning could also be enhanced in the Cache River watershed through sustained social interactions among the various stakeholders in the watershed, including local communities, in all stages of decisionmaking from planning to monitoring and evaluation. The promotion of community-based monitoring (Berkes 2007) is one promising approach for integrating local knowledge into resource management in a way that also enhances social learning. For instance, in an evaluation of 18 community-based ecological monitoring and assessment projects in the United States, Fernandez-Gimenez et al. (2008) found that these projects led to several benefits, including enhanced ecological understanding among participants as well as social learning that contributes to adaptive management.

Cultivating Interest

Ostrom (2009) has noted that actors' interest in engaging in collective processes of institutional change boils down to the computation of the costs and benefits of their actions. In many instances, individuals and organizations need to be convinced that it is in their best interest to engage in proenvironmental behaviors (McLain and Lee 1996). Factors influencing the behavior and motivations of actors are multi-dimensional and include cultural, psychological, economic, policy, and institutional factors (Lambin 2005). The use of different types of institutional mechanisms is therefore more likely to succeed in enhancing rule compliance and innovative responses to social-ecological change (Dietz et al. 2003, Westley et al. 2011). Adaptive comanagement relies on a range of monetary and nonmonetary incentives to enhance cooperative behavior (Plummer et al. 2012).

One attribute of adaptive comanagement that enables the provision of economic incentives is its holistic scope. Plummer and Armitage (2007: 65) have argued that, "The instrumental rationale of adaptive comanagement is sustainability: it aims to solve resource problems through a collaborative process which fosters ecologically sustainable livelihoods." The emphasis on sustainable livelihoods and well-being enhancement is important for generating interest since economic incentives are known for their effectiveness in changing behaviors (Vincent 2007). Another way that adaptive comanagement could generate interest is through its contributions to social learning and social memory (Westley et al. 2011). Through the promotion of interaction processes by which social learning occurs, adaptive comanagement can contribute to creating shared meanings, values, and preferences, as well as the building of trust and social capital that enhance collective responses to social-ecological change (McLain and Lee 1996, Olsson et al. 2004, Plummer et al. 2012).

In the case of the Cache River watershed, the use of an adaptive comanagement approach suggests the need to employ a diversity of institutional mechanisms to sustain the interest of stakeholders in the management of the watershed. Karkkainen (2004) attributes the success of the Chesapeake Bay program to the use of different types of mechanisms to gain support for policy implementation,

such as regulatory mechanisms, promotion of voluntary cooperation, use of social pressure, as well as provision of technical and financial assistance. In this regard, the current ecological focus of the Cache River Watershed Resource Plan needs to be broadened to include the livelihood needs of communities in the watershed. Innovative funding mechanisms, such as payments of ecosystem services, could be explored to address the needs of local communities. Beyond these economic incentives, greater attention needs to be paid to promoting inclusive and participatory decisionmaking processes through which social learning and social capital can emerge to promote cooperative behavior in the watershed. For instance, in the case of the community-based monitoring programs discussed earlier, Fernandez-Gimenez et al. (2008) found that besides their contributions to learning, the programs had built trust among participants, thereby enhancing opportunities for rule compliance and collective action.

Creating Opportunities

Institutional structures and processes that promote participation and communication among networks of diverse actors are essential for resilience building (Plummer et al. 2013). Additionally, the availability of arenas or forums for social interaction is critical for enhancing shared understandings and promoting collective responses to social-ecological change (Gunderson et al. 2006). Adaptive comanagement creates opportunities for social interaction through a reliance on multilevel institutional structures that provide vertical and horizontal linkages among actors (Armitage et al. 2009) and a decisionmaking process that is based on communication and conflict management (Plummer and Baird 2013).

First, the institutional structure of adaptive comanagement responds to the need for connecting actors across multiple scales in addressing challenges at any given scale (CIFOR 2008). The multilevel institutional structures allow for horizontal interaction among actors within levels as well as vertical interaction among actors across scales (Berkes 2009, Plummer and Baird 2013). The linking of semiautonomous actors within and across scales enhances the fit between institutions and management challenges (Plummer et al. 2012), sharing of decisionmaking authority at appropriate scales, and the flexibility of institutions in responding to change (Plummer and Baird 2013).

Second, the adaptive comanagement process also emphasizes communication and negotiation as mechanisms for conflict management (Plummer and Baird 2013). Adaptive comanagement recognizes the diversity of interests and knowledge systems of participants and seeks to promote shared understandings and joint problem-solving (Plummer and Fennel 2009). The explicit recognition of conflict in the adaptive comanagement process calls for the use of conflict management professionals, such as facilitators, who could enhance the linkages and interactions among actors and build their capacity for sustaining the iterative problem-solving and learning processes (CIFOR 2008).

The existing institutional structures and decisionmaking processes in the Cache River watershed do not provide adequate opportunities for vertical and horizontal interaction among stakeholders. Regulatory interventions at the level of the state or other appropriate level of action appear necessary to safeguard the role of local communities and other marginalized stakeholders in the management of the watershed. For instance, in their analysis of the involvement of indigenous communities in adaptive water governance in the United States and Australia, Bark et al. (2012) concluded that

the availability of legislation recognizing tribal water rights provided better opportunities for the involvement of indigenous communities in water governance in the western United States than Australia. The establishment of channels of communication and deliberation between local and nonlocal actors is also critical in the Cache River watershed. Given the history of conflict in the watershed, an explicit recognition of conflict as an inherent component of the resource management process and investment in the capacity for conflict management could also enhance the process and outcomes of deliberation among stakeholders in the watershed.

Building Capacities

A central focus of adaptive comanagement is to build adaptive capacity, particularly at the local level and regional levels (Plummer and Baird 2013). Adaptive comanagement “strives to recognize, build on and strengthen local people’s capabilities in addressing the challenges that their changing environments pose” (CIFOR 2008: 1). The capacity to adapt to change is a function of access to various forms of capital (social, economic, human, natural, and physical) and the availability of appropriate institutions and governance systems (Akamani 2012, Walker et al. 2006). Adaptive comanagement has the potential to build and enhance access to the capitals and institutions needed for adapting to change.

With regard to institutions, adaptive comanagement draws from the benefits of comanagement, such as equity, efficiency, effectiveness, and legitimacy, in decisionmaking (Plummer and FitzGibbon 2004), as well as cross-scale linkages that enhance participation, information access, flexibility and response capacity at the local level (Berkes and Jolly 2001). Adaptive comanagement also enhances access to various forms of capital as it seeks to achieve outcomes, such as poverty reduction, enhanced well-being, increased food security, enhanced knowledge, and improved ecosystem health (McDougall et al. 2013, Plummer and Armitage 2007).

In the case of the Cache River watershed, the marginalization of local communities from decisionmaking processes in the watershed coupled with the adverse effect of regional influences on local livelihoods may have eroded community capacity to adapt to change. In addition to modifications in the institutional structures and processes discussed earlier, the pursuit of adaptive comanagement will require external support in building community institutional infrastructure and capital assets. As Bark et al. (2012: 174) have noted, “Without a process to access legal entitlements and without significant government funding for capacity building in...communities and water planning, planning mechanisms will prove less effective.”

CONCLUSIONS

Research on the governance of water resources is increasingly embracing complexity and resilience thinking. In this regard, adaptive comanagement has recently emerged as a promising mechanism that combines the learning orientation of adaptive management with the vertical and horizontal linkages of comanagement. The analysis of transitions in the Cache River watershed has shown that progress has been made in promoting a collaborative approach to managing the watershed. However, the current regime still remains vulnerable due to its lack of prioritization of integrated adaptive management and limited community participation.

The paper has argued for the adoption of adaptive comanagement as a framework for informing institutional reforms for building social-ecological resilience in the watershed. Adaptive comanagement has the potential to contribute to creating awareness about social-ecological complexity, generating interest among actors through economic and noneconomic incentives, creating opportunities for involvement by diverse actors, and building the capacity for institutional change across scales. In spite of its promise, it must be cautioned that adaptive comanagement cannot be seen as a panacea (Armitage et al. 2009). Folke et al. (2011) has noted the difficulty of breaking down the robustness of older regimes and the challenge of consciously designing the multilevel institutional frameworks required for adaptive comanagement and adaptive governance. Similarly, Akamani (2014) has identified a range of challenges from the metaphysical to the practical that constrain the transition toward adaptive water governance approaches. Nonetheless, success in the transition toward adaptive comanagement could be enhanced where favorable conditions exist, such as leadership by key individuals and bridging organizations (Gunderson et al. 2006, Olsson et al. 2008), as well as an enabling policy environment that promotes transparency, participation, and legitimacy in decisionmaking (Folke et al. 2011). These preconditions for successful transition deserve attention in the Cache River watershed.

ACKNOWLEDGMENT

I would like to thank Dr. John Groninger and Dr. Eric Holzmüller for the opportunity and encouragement to present at this conference. Funding for my participation in the conference came from the Department of Forestry, Southern Illinois University.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

FOREST ECOLOGY

USING PHYSICAL PARAMETERS AND GEOGRAPHIC INFORMATION SYSTEM ANALYSES TO PREDICT POTENTIAL RIPARIAN RESTORATION SITES FOR GIANT CANE IN SOUTHERN ILLINOIS

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Abstract.—Riparian buffers have been widely advocated as a best management practice for improving stream and lake water quality. Giant cane (*Arundinaria gigantea*) is a good candidate to include in multispecies riparian buffers designs, as it promotes infiltration of surface runoff and deposition of sediment and associated nutrients. To examine the potential of giant cane as a riparian zone species in the Cache River watershed in southern Illinois, we identified common physical site characteristics for 140 existing southern Illinois canebrakes. Percent slope, soil taxonomy, and pH, along with digital elevation models and land cover mapped with geographic information systems, were used to determine the potential suitability of sites within the watershed for canebrake plantings and general riparian restoration. The following soil characteristics were determined to be associated with giant cane success: percentage of area containing slopes of ≤ 3 percent, fine to coarse-silty textures, pH of 5.3–6.7, effective cation exchange capacity of < 30 units, available water holding capacity ≥ 0.12 , bulk density of 1.37–1.65 g/cm³, and percent clay of 11–55. Eighty percent of existing giant cane sites were found within these slope and soil characteristics. The total area of potential riparian canebrake landscapes based on these parameters is 7,470 ha within the Cache River watershed.

INTRODUCTION

Giant cane (*Arundinaria gigantea*) was historically a dominant component of riparian areas in the lower midwestern and southeastern United States, including southern Illinois (Brantley and Platt 2001, Platt and Brantley 1997, Platt et al. 2009). Today, giant cane occupies only 2 percent of its historical range and is listed as a “critically endangered” species due to factors such as overgrazing by domestic livestock, altered fire regimes, agricultural land clearing, and flood control projects (Brantley and Platt 2001, Noss et al. 1995). Cane is a native bamboo species with a relatively dense rooting network that resists erosion, increases nutrient uptake, and promotes infiltration in riparian zones (Brantley and Platt 2001). Its ability to promote infiltration of surface runoff and deposition of sediment and associated nutrients through its high density culms and extensive shallow rooting network makes giant cane a good candidate to include in multispecies riparian buffer designs (Schoonover et al. 2005, 2006). Giant cane performs as well as, or better than, forest vegetation in nitrogen renovation in groundwater (Schoonover et al. 2010). It also provides significant wildlife habitat benefits, especially in the fragmented midwestern landscape (Blattell et al. 2009).

To determine where giant cane should be considered and targeted for restoration, existing stands need to be analyzed for common landscape or physical characteristics, such as soils or topography. These factors can then be used to define areas suitable for canebrake plantings and general riparian restoration. Restoration budgets are limited, so targeting species to areas where they will have the

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most success is critical for efficient use of funds. Geographic information system (GIS) and remote sensing technologies are well established as excellent tools for delineating and mapping species distributions (Boyd and Foody 2011, Franklin and Miller 2010). These geotechnology tools have been used in a variety of studies such as determining riparian restoration sites (Russell et al. 1997), tracking invasive species (Pande et al. 2007), and mapping vegetation with remote sensing (Akasheh et al. 2008).

METHODS

The Cache River watershed is classified as a hydrologic unit code 12-level basin in southern Illinois. Infrared aerial photography of the Cypress Creek Refuge and Cache River watershed was taken in March 2009 by the U.S. Fish and Wildlife Service (USFWS). The leaf-off imagery was collected by using three spectrum bands of red, green, and near-infrared at a spatial resolution of 0.21 m. Using visual interpretation of the photography, we identified the location of 140 canebrakes. Canebrakes were confirmed with groundtruthing to determine the accuracy of the interpretation process (ArcGIS Desktop: Release 9.2, Esri, Redlands, CA). Groundtruthing proved to be 100 percent accurate as the presence of giant cane was confirmed at all predicted locations.

Soil quality such as chemical composition, moisture retention capability, and texture can be used to guide water quality buffer restoration (Dosskey et al. 2006). Soil characteristics used to measure soil quality are readily available through the Soil Survey Geographic (SSURGO) database and include soil taxonomy classification, pH, bulk density (BD), available water capacity (AWC), effective cation exchange capacity (ECEC), and percent clay (PC).

The SSURGO database includes GIS polygons, referred to as map units, which are given unique keys to identify map units that have common soil characteristics. The aforementioned soil characteristics were assigned to the map units by using the Soil Data Viewer provided by the U.S. Department of Agriculture's Natural Resources Conservation Service (Natural Resources Conservation Service 1995). By assigning the map-unit key to each cane site, we were able to link the soil characteristics to each of our samples by using ArcGIS. Tabular data containing all the soil information for the cane sites were exported for further analysis.

JMP statistical software (SAS Institute Inc., Cary, NC) was used to create frequency tables of all the soil characteristics. Soil parameters were assigned based on the range and frequency of values for each soil characteristic (Table 1). Each soil parameter was then used to select SSURGO map units that were most likely to contain giant cane.

Table 1.—Parameters used to define potential giant cane soil characteristics in Cache River watershed, Illinois

	Slope (%)	Soil taxonomy	pH	ECEC (meq)	AWC	Bulk density (g/m ²)	Clay (%)
Range	0.5 - 27	Fine, fine silty, fine loamy, coarse silty, coarse loam	5.3 – 7.1	0 – 28.9	0.1 – 0.2	1.4 – 1.8	11 – 55
Parameters used in analysis	<3.0	Fine, fine silty, and coarse silty	5.3 – 6.7	<30	>0.1	1.4 - 1.7	11 – 55

In addition, proximity of the 140 canebrakes to streams and roads was determined, and JMP statistical software was used to determine correlations. Based on these correlations, only distance to streams was significant. To further delineate the potential giant cane landscapes, we created stream buffer polygons from the stream segments in ArcGIS to represent the variation in measures of distance to stream. We decided to use three different measures of distance to stream to create the stream buffer polygons. Stream segments were buffered to 50-, 100-, and 200-m polygons by using the ArcGIS line buffer tool. We then selected the overlapping areas of the soil polygons and the stream buffer polygons to create three different potential giant cane landscapes, and determined how many existing canebrakes were within each of the landscapes.

Land cover data were obtained from the 2012 cropland data layer (CDL) to determine the current land covers within potential giant cane polygons. The associated tables were exported and evaluated for percentage of land cover based on the general categories of grassland or forest, cultivated cropland, and developed land.

The three-band visual-near-infrared (VNIR) imagery collected by the USFWS provided the opportunity to also map canebrakes by using supervised image classification, a common process in image classification and species mapping (Akasheh et al. 2008, Verburg et al. 2011). Because the imagery was collected during leaf-off and cane is an evergreen species, we assumed that the cane would have a unique signature relative to the otherwise dormant vegetation. We used the image classification analysis tool in ArcMap 10.0 (Esri) to identify potential canebrakes. Supervised classification uses groundtruth data as signatures to classify remote sensing imagery. Using a hand-held global positioning system (GPS), we collected a data set of well-established canebrakes. Signatures for the image classification process were created from the data set. An iterative process using supervised classification methodologies was used to delineate the different signatures of the digital imagery.

For the first iteration, we delineated seven classifications. Classification 1 included the GPS-collected canebrake polylines to determine the canebrake signature. The second was stream polygons digitized from USFWS imagery. The other five land covers were difficult to determine because the imagery was collected in March during the leaf-off season. Therefore, we obtained land cover data from the CDL from the previous growing season to classify the other five signatures. The classifications were determined by locating individual plots of land on the imagery, then classifying them using the CDL data set. The land covers used for classification were corn, soybeans, forest, roads, and wetlands.

After creating a signature file with the seven land covers, we used a maximum likelihood classification to create a map of the seven land covers. To determine the accuracy of the classification process, we randomly selected eight of the larger clusters of cane pixels from the classified imagery which we thought would be accessible for the groundtruthing process.

Groundtruthing established that only two of the eight randomly selected clusters contained canebrakes. Four of the sites were determined to have cypress (*Taxodium distichum*) trees present; two had neither cane nor cypress. This information was used to conduct the second iteration of the classification process. Based on the groundtruthing results, we assumed that cane and cypress could possibly have similar signatures that the first iteration could not differentiate.

For the second iteration we used GPS-collected areas of cypress trees as an eighth land cover in the classification process. To further delineate cane, groundtruthed noncane areas that were classified as cane in the first iteration were also included in this iteration. It was assumed that using these two new signatures would help to establish a more distinctive signature for canebrakes. As expected, the second iteration reduced the classified cane pixels by roughly 25 percent compared to the first iteration.

To establish contiguous cane pixels and remove extraneous pixels, the output of the second iteration was processed by using the majority filter tool in ArcGIS. Pixels of cane locations were then converted to point locations to be processed with the point density tool in ArcGIS. Eighteen of the densest areas of classified cane were extracted for a second round of groundtruthing, but this second iteration did not increase the accuracy. Consequently, using remote sensing to locate canebrakes was not likely to be feasible.

RESULTS

Close to 50 percent of canebrakes were on land with less than 1-percent slope. Almost 20 percent of canebrakes were on land with a 1- to 2-percent slope. Above 2-percent slope, there was no significant difference in relative occurrence among the slopes (Figs. 1 and 2). The fewest canebrakes were located on southwest- and western-facing slopes, though there was no significant difference among the remaining aspect directions (Fig. 3). The highest number of canebrakes was found between 100 and 105 m above mean sea level (m msl) (Fig. 4). Seventy-six percent of the canebrakes grew between 95 and 110 m msl, and cane was found only between 90 and 150 m msl.

Eighty percent of canebrakes were found on silt loam soils; however, this grouping includes the soil classification Bonnie and Petrolia, which is a mixture of both silt loam and silty clay loam soils (Fig. 5). Bonnie and Petrolia soils, which constitute half of silt loam soils, occur on nearly level flood plains of 0- to 2-percent slope and occasionally on flood-plain steps. They formed in light-colored, recently deposited, acid (although less so for Petrolia), silty alluvium. These soils are classified as poorly drained and very poorly drained. Saturated hydraulic conductivity is moderately high, and permeability is moderately slow. Stream flooding occurs frequently to rarely, and occurs commonly in the winter and spring. In the undrained condition, these soils have an intermittent apparent water table from as much as 0.6 m above the surface to 0.01 m below the surface, typically between October and July. Where drained, an intermittent water table is within 0.3 m of the surface, typically between December and May. The potential for surface water runoff is low to medium (U.S. Department of Agriculture 2011).

About half of the canebrakes (48 percent) were found within 40 m of a stream, with 26 percent found within 20 m (Fig. 6). There was an equal likelihood of finding cane at any distance beyond 60 m. Fifty-five percent of sites were found within 50 m of a stream, 17 percent were located between 50 and 100 m of a stream, and 19 percent were between 100 and 200 m of a stream. However, many of these streams were ephemeral and contained water only during high runoff periods.

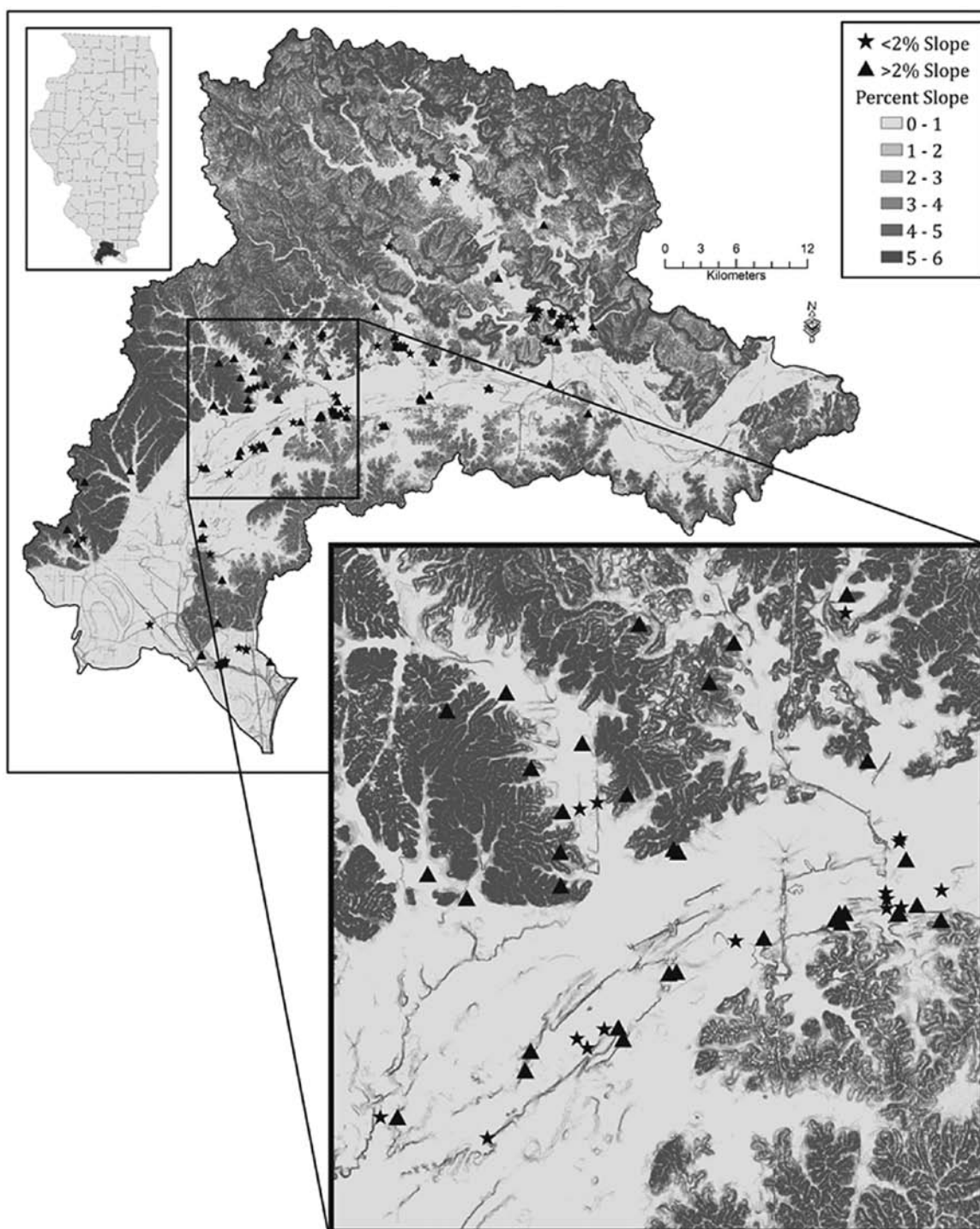


Figure 1.—Slope map of Cache River watershed, southern Illinois, with the canebrakes marked in stars. Low slope areas are marked by the darker shade.

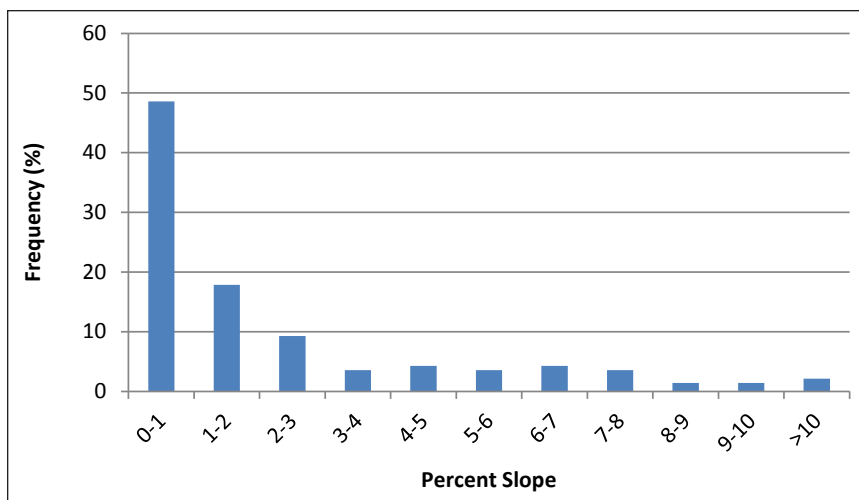


Figure 2.—Frequency of canebrake occurrence by slope class from a survey in the Cache River watershed, Illinois.

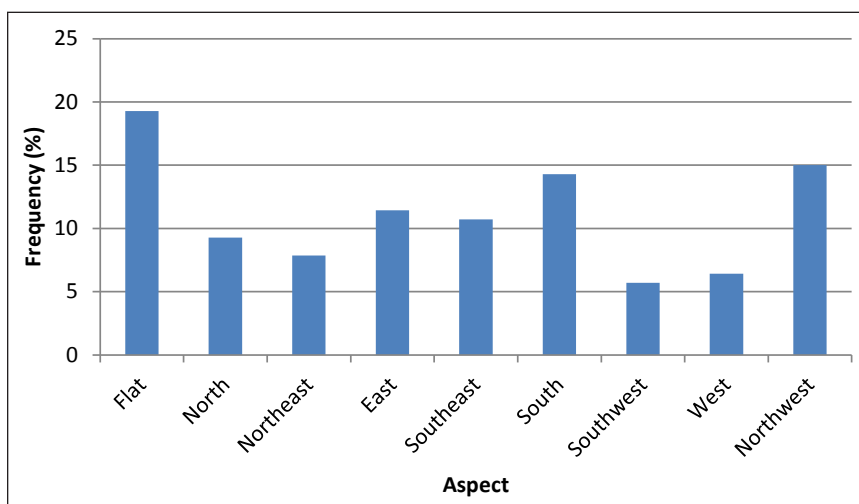


Figure 3.—Frequency of canebrake occurrence by aspect from a survey in the Cache River watershed, Illinois.

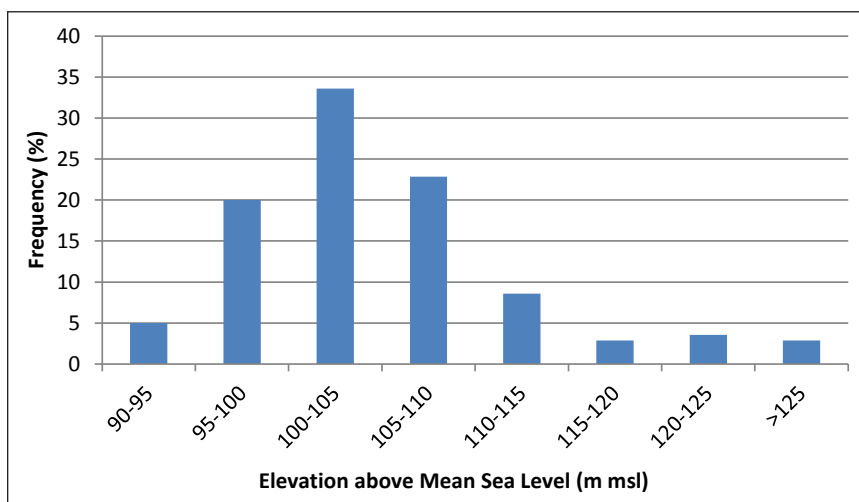


Figure 4.—Frequency of canebrake occurrence by elevation above mean sea level from a survey in the Cache River watershed, Illinois.

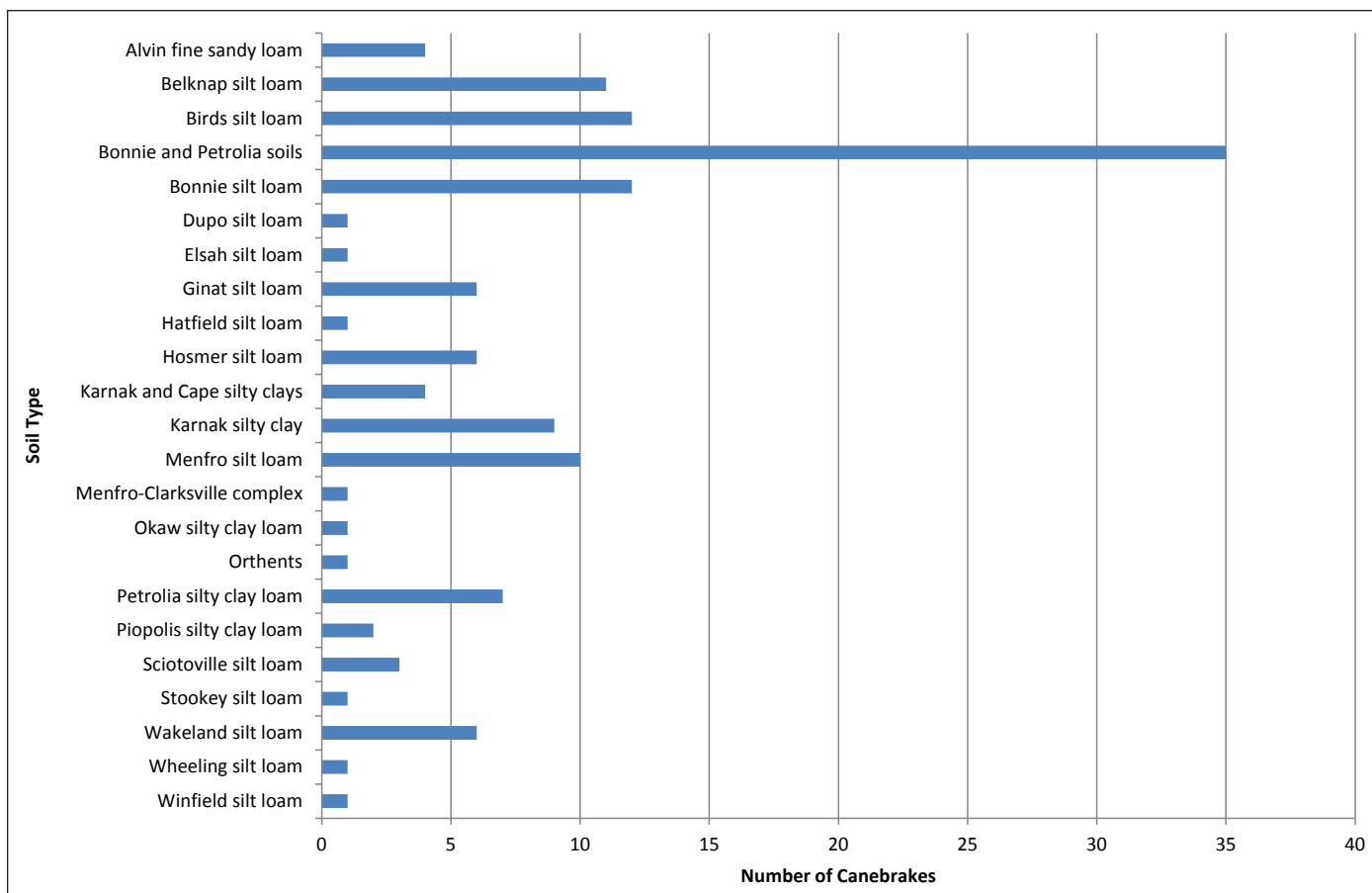


Figure 5.—Soil series within canebrakes from a survey in the Cache River watershed, Illinois.

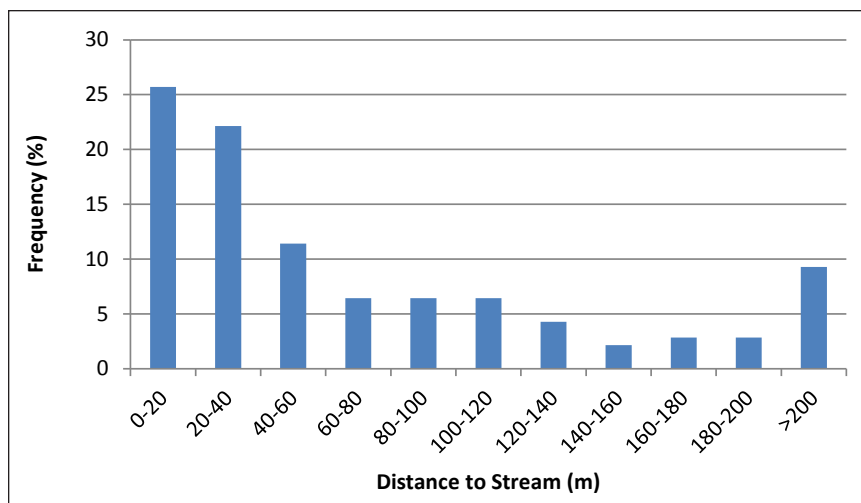


Figure 6.—Proximity of canebrakes to streams, including ephemeral channels, from a survey in the Cache River watershed, Illinois.

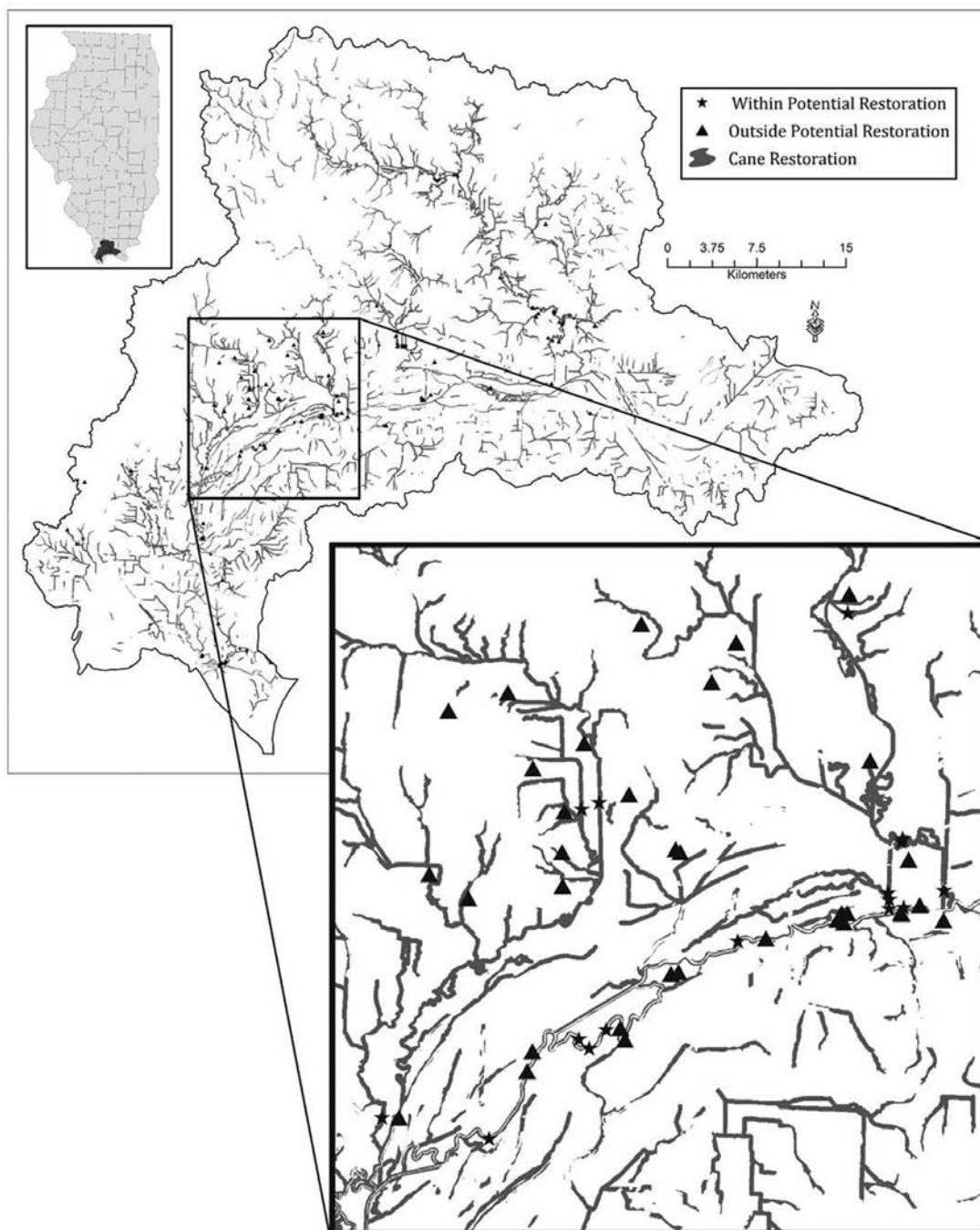


Figure 7.—The existing canebrakes within the potential giant cane landscapes, defined by a 50-m stream buffer and the soil parameters, Illinois.

The results of the spatial analysis to delineate potential areas of giant cane were mixed. When existing canebrakes were compared only to the soil polygons, slightly fewer than 80 percent of the canebrakes were found within the polygons selected by the slope and soil parameters (Fig. 7). However, that number increased to 88 percent by including cane sites within 25 m of the polygons. We made this adjustment by adding a 25-m buffer to the polygon using the ArcGIS buffer tool. This result can be explained in two ways. First, borders are placed subjectively on the SSURGO map units, where they may not correspond to specific demarcations between characteristics. Soils may change gradually rather than abruptly over the landscape, depending on topography. Second, some spatial error occurs both in placing cane site points and in GIS processing.

However, when restricting the potential cane sites further by using the 50-, 100-, and 200-m distance to stream buffers, the proportion of canebrakes within the potential sites was reduced to 75 percent. More specifically, 49 percent of existing canebrakes were found within the 50-m stream buffer. Fourteen percent of the sites were found between 50 and 100 m, and 12 percent were found between 100 and 200 m.

As expected, potential giant cane landscapes >50 m from streams have a greater percentage of cropland than those near streams. Two-thirds of the potential giant cane landscapes within 50 m of a stream were grassland or forest. These land covers provide a much better opportunity for cane to be present than other land covers. However, lands within 50 m of streams do contain 25 percent cultivated crops. Lands between 50 and 100 m of streams have a similar ratio of land covers as those between 100 and 200 m: 46 percent cropland, 48 percent grassland or forest, and 6 percent developed.

DISCUSSION

More than half of the canebrakes were found within 50 m of a stream, and most of those were within 20 m and on a 1-percent slope or less. The total area of potential landscapes in the Cache River watershed is 14,590 ha, although this amount would be reduced to 7,470 ha by areas of existing forest, about 6,500 ha, and continuous open water, roughly 620 ha, neither of which is likely to be considered for cane restoration.

We think that identifying the potential giant cane landscapes demarcated by a 50-m buffer width would be an appropriate way to determine where giant cane could be used as a riparian buffer (Fig. 7). This conclusion is consistent with the statement by Griffith et al. (2009) that ideal cane restoration sites are on flood plains of rivers and streams where the rooting zone of the plant is out of the zone of saturation of the adjacent stream. It is also consistent with studies that present evidence that riparian buffers within 40 m of streams can reduce nitrogen, phosphorus, and sediment loads (Karr and Schlosser 1978; McColl 1978; Schlosser and Karr 1980, 1981a, 1981b). Although roughly one-half of the existing canebrakes are within the potential giant cane landscapes defined by a 50-m stream buffer and the soil parameters, the other half exist under a variety of conditions beyond stream proximity and the soil parameters analyzed. Therefore, more analysis is needed to determine the most appropriate areas for canebrake restoration.

The first round of groundtruthing showed the remote sensing process to be minimally accurate. Only two of the selected sample sites contained cane, and four sites were determined to be cypress trees. The second round of groundtruthing of the supervised classification process yielded very poor results. Only 2 of the 18 potential sites could be verified as existing giant cane.

It was difficult to delineate giant cane by using supervised image classification because of the similarity in reflectance signatures of giant cane and other species within the VNIR imagery. The spatial resolution (0.21-m pixels) was extremely good for image classification purposes, but the single near-infrared band of the imagery limited the ability to separate the cane and cypress signatures, as well as those of other evergreen species on the landscape. Imagery that provides a broader range of the electromagnetic spectrum could potentially solve this issue. Although satellite imagery currently

available can provide this multispectral range, the small size of the typical giant canebrake may not be detectable by the larger spatial resolution of these platforms.

Light detection and ranging (LIDAR) could potentially be an alternative to traditional remote sensing platforms. A study of this type could be improved by LIDAR in two ways. First, LIDAR has the potential for species classification because the platform uses the same visible and infrared spectrum as some satellite sensors. To our knowledge, however, LIDAR has been applied to delineate only levels of vegetation, not species-specific classification. Second, LIDAR could be used in this study through its more common use, digital elevation models (DEMs). As shown, the best restoration areas for giant cane are within level flood-plain zones. A wetness index can determine where the greatest flow accumulation will occur based on elevation data, thus delineating areas of level flood plains. This approach was attempted for this study by using DEMs from the U.S. Geological Survey (USGS), but the USGS data are not detailed enough to create an adequate wetness index for our study area. The much greater spatial resolution of LIDAR offers potential to create a usable wetness index.

The 25 percent of the potential giant cane landscape (50-m buffer) that is cropland could benefit from a riparian buffer capable of reducing sediment and nutrient runoff from cropland. Ideally, at least some buffer width between crops and streams is recommended, though there are no regulations in Illinois requiring farmers to leave such a buffer. Canebrakes create thick networks of rhizomes belowground that effectively stabilize soils along riparian corridors. Giant cane would be very appropriate as a riparian buffer where buffer width is limited because its dense aboveground stands of culms result in litter accumulation and high soil porosity, which promote infiltration, inhibiting—and in some cases eliminating—the transport of sediment and nutrients into adjacent streams (Schoonover and Williard 2003; Schoonover et al. 2003, 2005). Therefore, a more detailed spatial analysis that would show the spatial relationship of crops, cane, and streams would be an important next step in this type of project.

CONCLUSIONS

Although this research showed some of the challenges in using GIS to determine current and potential sites of giant cane, it also provided some of the predominant topographic characteristics of canebrakes. The status of cane as a primarily riparian species was confirmed. We were able to determine appropriate soils for canebrakes located within 50 m of streams. The identification of potential sites could possibly be improved through further analysis. These analyses include potential runoff from agricultural fields. Current DEM information is not detailed enough for this landscape. A LIDAR data set could possibly allow for finer-scale evaluation. A more detailed analysis of the relationship between agricultural fields and the potential sites also could help to define sites in more immediate need of restoration. In the southern portion of the Central Hardwood region where the Cache River basin is located, riparian buffers may be critical for nutrient attenuation in upland riparian sites, where there may be less potential for denitrification relative to lowland areas with shallow water tables and greater tendency to flood (Blattel et al. 2009). Based on the current location of successful canebrakes, giant cane seems to be an excellent candidate for establishing riparian buffers.

ACKNOWLEDGMENTS

This research was supported by a National Science Foundation Integrated Graduate Education Research Traineeship. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the authors and do not necessarily reflect the views of the National Science Foundation. The authors would like to thank the U.S. Fish and Wildlife Service for giving us access to its land and photographs, and Jacqueline Lintzenich for field assistance.

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LONG-TERM IMPACT OF CLEARCUTTING, DEER BROWSING, AND DEFOLIATION ON STAND DEVELOPMENT IN A PENNSYLVANIA MIXED-OAK FOREST

Aaron D. Stottlemeyer¹

ABSTRACT

Introduction

The long-term impacts of even-age forest management and excessive browsing by white-tailed deer (*Odocoileus virginianus*) on regeneration are unknown for hardwood forests in the eastern United States. In 1965, Gene Wood, a graduate student at Pennsylvania State University, initiated a study in a mixed-oak forest in central Pennsylvania to examine changes in forest composition in response to clearcutting and deer browsing (Wood 1971). After the study ended in 1972, no other management was implemented, thus presenting a unique opportunity to collect long-term data on stand development. The primary objectives of our study were to examine forest composition after clearcutting with or without fencing by comparing present-day conditions to those observed in 1965.

Methods

The study area was a 5.4-ha portion of a mixed-oak forest in the Quehanna Wild Area, Moshannon State Forest, in central Pennsylvania. When the stand was first inventoried in 1965, black (*Quercus velutina*), white (*Q. alba*), and red (*Q. rubra*) oaks were dominant and trees were fairly uniform in age at 55-60 years (Wood 1971). Common woody understory species were witch-hazel (*Hamamelis virginiana*) and serviceberry (*Amelanchier arborea*). Tree seedlings and woody shrubs including blueberry (*Vaccinium* spp.), huckleberry (*Gaylussacia* spp.), and sweet-fern (*Comptonia peregrina*) were present, but sparse. Hummer (1970) measured deer density at more than 15 deer per km², a density exceeding the local carrying capacity based on the presence of a severe browse line.

An interior 2.6-ha portion of the stand was clearcut in fall 1966 with no woody vegetation >1 m in height left standing. Shortly after the cutting was completed, eight sampling plots 40 m by 40 m (0.16 ha) in size were established with permanent metal stakes marking the corners. Four plots were located in the 2.6-ha clearcut portion, and four plots were located in the surrounding uncut portion of the 5.4-ha stand to serve as controls. Woven wire fences 2.4 m in height were constructed to exclude deer from one-half (0.08 ha) of each plot, which created four replications of each treatment combination: clearcut with fencing, clearcut with no fencing, control (no cutting) with fencing, and control with no fencing.

We found the original study plots in 2012 and tallied all trees ≥10 cm in diameter at breast height (d.b.h.) rooted inside the 0.08-ha subplot. Additionally, we visually estimated percentage cover of woody shrubs in two 1.8-m-radius nested plots per 0.08-ha subplot. From the inventory data, we calculated the total density of each tree species present and percentage shrub cover for each treatment combination and compared these to Wood's data collected before clearcutting and fencing in 1965.

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Results and Discussion

As of spring 2012 (45 growing seasons after clearcutting), all of the unfenced clearcuts had failed to regenerate with trees. In fact, we found only two trees growing among the four unfenced clearcut plots; both were red maples (*Acer rubrum*) and in the same plot. Instead of regenerating with trees, a dense, continuous layer of woody shrubs covered >50 percent of the ground surface, on average. Apparently in the unfenced clearcuts, excessive browsing of tree regeneration by white-tailed deer allowed the woody shrubs and grasses to spread and capture the new growing space (Horsley et al. 2003). The deer and shrubs together have prevented forest regeneration for nearly half a century.

Fenced clearcuts successfully regenerated, but dominance shifted from oak (72 percent relative density in 1965 to 26 percent in 2012) to red maple (28 percent in 1965 to 69 percent in 2012). This shift from oak to maple may be due, in part, to poor sprouting potential of the larger oaks (Weigel and Peng 2002). Woody shrub cover was <1 percent in these areas, on average.

Fenced and unfenced control plots also shifted in composition, from largely oak (72 percent relative density in 1965 to 26 percent in 2012) to red maple (28 percent in 1965 to 78 percent in 2012). This shift likely resulted from multiple defoliations by oak leaf roller (*Archips semiferanus*) between 1969 and 1972, followed by gypsy moth (*Lymantria dispar*) in the early 1980s and 1990s, which caused many residual oaks to die, thereby releasing the red maples. Total tree density in 2012 in unfenced controls (395 trees/ha) was markedly lower than that in fenced controls (566 trees/ha), likely due to deer browsing of regeneration after insect-related mortality of overstory oaks.

Conclusions

Results from this study provide information concerning the potential long-term influence of clearcutting, excessive deer browsing, and repeated defoliations by insects on stand development in mixed oak forests. Future work should focus on blueberry-huckleberry-sweet-fern as interfering vegetation and methods for reforesting these areas.

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UNDERSTORY VEGETATION COMPOSITION AND ABUNDANCE IN RELATION TO LIGHT, WATER, AND NUTRIENT SUPPLY GRADIENTS IN UPLAND OAK WOODLANDS

Elizabeth K. Olson and John M. Kabrick¹

ABSTRACT

Introduction

The Ozark Highlands of Missouri have experienced a complicated series of exploitive events (Flader 2004). The area was heavily cut over for timber at the turn of the last century and was overgrazed by privately owned livestock through the early 1900s. Decades of fire suppression since the 1940s further altered plant composition and structure. The current state of forest and woodland herbaceous communities may be quite unlike those seen historically.

The main difference between a woodland and a forest lies in the abundance of woody species in the midstory and understory, which affects the ground layer vegetation by blocking sunlight. Forests generally have a well-developed midstory and understory, and therefore have more spring ephemerals and shade-tolerant herbaceous species such as spotted geranium (*Geranium maculatum*) and black baneberry (*Actaea racemosa*). Woodlands have an open midstory and understory; more light penetrates to the ground layer to support a rich diversity of shade-intolerant species like goldenrods (*Solidago* spp.) and tick-trefoils (*Desmodium* spp.).

Ground layer vegetation composition has been shown to be closely aligned to light, water, and nutrient supply gradients. The relationship between environmental gradients and the distribution of woodland and forest plant indicators in the absence of disturbances is poorly understood. The objective of this study was to examine ground vegetation across gradients of light, moisture, and nutrient availability to determine how the composition and abundance of forest and woodland indicator plant species are affected by these gradients in undisturbed stands.

Methods

This study was conducted on the Sinkin Experimental Forest, which is operated and maintained cooperatively by the U.S. Forest Service, Northern Research Station and Mark Twain National Forest. The Sinkin Experimental Forest is located in Dent County, Missouri, approximately 40 km southeast of the town of Salem. It is located in the Current River Hills subsection of the Ozark Highlands, a dissected plateau with steep slopes, narrow ridges, and local relief of about 61 m. The soils of the ridgetops and upper hillsides are highly weathered, droughty, and strongly acidic. The soils of the lower hillsides are less weathered, are underlain by dolomite, and have greater cation exchange capacity and water holding capacity.

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Site selection was based on the presence of mature, fully stocked, closed-canopy oak stands that have not undergone substantial disturbance within the last 15-20 years. The stands were oak-pine and mixed oaks. The dominant trees were shortleaf pine (*Pinus echinata*), white oak (*Quercus alba*), and black oak (*Q. velutina*), which each make up about 20 percent of the basal area. Scarlet oak (*Q. coccinea*), hickories (*Carya* spp.), and northern red oak (*Q. rubra*) were less common. Elm (*Ulmus* spp.), flowering dogwood (*Cornus florida*), blackgum (*Nyssa sylvatica*), black walnut (*Juglans nigra*), and red maple (*Acer rubrum*) were least common (Villwock et al. 2011).

Data were collected in 2008. We sampled one hundred twenty 12.6-m-radius vegetation plots across twenty 4.9-ha stands. Plots in each stand were oriented across a moisture and site quality gradient. To describe the terrain, the aspect, location (shoulder, backslope, footslope), position (upper, middle, lower), and shape (concave, linear, convex) of the slope were recorded at plot center. Below-canopy light was measured to obtain photosynthetically active radiation (PAR). Soil samples were collected and analyzed for available water, cation exchange capacity (CEC), and pH. Diameter at breast height (d.b.h.) was recorded for overstory and midstory trees >5.0 cm d.b.h. Understory vegetation was recorded in four 1-m² subplots; all species were identified and percent cover was estimated in 10-percent classes. Herbaceous species were categorized as forest or woodland indicators, or generalists.

Statistical methods included multivariate ordination, Spearman correlation, regression trees, and multiple linear regressions. Regression trees and multiple linear regressions were used to examine the site, soil, and overstory factors important to forest and woodland indicator species abundances.

Results

A total of 165 species were identified in the 120 plots. Herbaceous plants were categorized as forest indicators (29 species), woodland indicators (65 species), and generalists (14 species). The most common forest indicators were false Solomon's seal (*Maianthemum racemosum*), pointedleaf ticktrefoil (*Desmodium glutinosum*), bearded shorthusk (*Brachyelytrum erectum*), and bellwort (*Uvularia grandiflora*). The most frequently occurring woodland indicators were shining bedstraw (*Galium concinnum*), Bosc's panic grass (*Panicum boscii*), and rue anemone (*Thalictrum thalictroides*). For analyses, the oak species were grouped into the red oak group (mostly black and scarlet oak) and the white oak group (mostly white and post oak).

Nonmetric multidimensional scaling was used to investigate the understory species composition. Aspect was the strongest determinant of ground flora composition; red oak stocking, pH, and CEC were also important factors. Northeast-facing slopes had higher pH and CEC, and greater stocking of non-oak species such as slippery elm (*Ulmus rubra*), pawpaw (*Asimina triloba*), and ash (*Fraxinus* spp.). Rattlesnake fern (*Botrychium virginianum*), *Geranium maculatum* (geranium), bellwort, and *Polygonatum biflorum* (Solomon's seal) were common species on northeast slopes. On southwest-facing slopes, soils were more acidic and red oak stocking was greatest; wild quinine (*Parthenium integrifolium*), late purple aster (*Symphyotrichum patens*), bracken fern (*Pteridium aquilinum*), and blueberry species (*Vaccinium pallidum*, *V. arboreum*, and *V. staminium*) were common in the understory.

Forest indicators (29 species) attained greatest percent cover and richness on northeast-facing slopes. Forest indicators had greatest cover where aspect was between 4 and 90°, total overstory basal area was <18.2 m²/ha, plots were not on upper or lower shoulder slopes, and stocking of non-oak species was >52 percent. Forest indicators had the least cover where aspect was between 110 and 290° and pH was <5.26. A multiple regression model ($R^2 = 0.54$; $p < 0.001$) with soil pH, red oak stocking, pine stocking, and density of red oak stems best explained the variation in forest indicator cover. Soil pH positively affected forest indicator cover, but all tree covariates in the model had negative effects on cover.

Woodland indicators (65 species) did not show preference for aspect; they had similar percent cover and richness on northeast- and southwest-facing slopes. Soil pH was the most important factor influencing woodland indicator cover; the greatest cover occurred where pH was >5.9. However, the majority of plots had pH <5.9; on these more acidic soils, this group of woodland species was tolerant of a wide range of conditions. The following associations are for plots with soil pH less than 5.9. Where non-oak stocking was <30 percent, woodland species cover was maximized on plots with more available water and hill shape was either linear or convex (not concave). Alternatively, where non-oak stocking was >30 percent, woodland species maximized cover on mid-footslopes, lower backslopes, and lower shoulder slopes. This complex pattern may indicate specific woodland species' requirements for measured or unmeasured site factors, or competitive interactions with generalist species. A multiple regression model ($R^2 = 0.38$, $p < 0.001$) with soil pH, available water, total red oak stocking, and overstory pine stocking best explained the variation in woodland indicator species cover. Soil pH and water availability both positively affected woodland species cover; red oak and pine stocking had negative effects.

Discussion

Aspect, tree stocking, and soil pH are currently the dominant forces structuring the ground flora in these undisturbed stands. The cover of both forest indicators and woodland indicators was negatively affected by pine and red oak stocking. High stocking levels may indicate less light penetration or leaf litter accumulation, or a combination thereof, representing a physiological limitation and a mechanical barrier to herbaceous growth. Increases in soil pH led to greater cover of both forest and woodland plants, whereas aspect is a limitation only to forest indicator species.

Midstory tree abundance and PAR were expected to be influential as they both are indicative of the amount of light available to the ground layer plants. We expected to find the forest indicator species strongly correlated with low PAR, high woody density in the midstory and understory, and shadier protected slopes (northeast aspect). Woodland indicator species were expected to be strongly correlated with high PAR, less dense woody vegetation in the midstory and understory, and sunny exposed slopes (south and west aspects). However, because there was minimal variation in midstory basal area and PAR in these undisturbed stands, these covariates were not informative for describing patterns in the understory composition or abundance.

These findings will provide important baseline data for planned future studies, which will examine the interactions among environmental gradients and disturbances including prescribed burning, timber harvesting, and the combination of burning and harvesting.

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GARLIC MUSTARD AND ITS EFFECTS ON SOIL MICROBIAL COMMUNITIES IN A SANDY PINE FOREST IN CENTRAL ILLINOIS

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Abstract.—This study evaluated the impacts of garlic mustard (*Alliaria petiolata*), an invasive species, on soil microbial community dynamics in a pine plantation on sandy soils in central Illinois. In situ soil carbon dioxide efflux was significantly greater in invaded sites. Similarly, in vitro carbon mineralization was significantly greater for soils collected from invaded sites, but only early in the incubation period. Incubations with selective inhibitors showed a decrease in fungi relative to bacteria. Nitrogen-free selective agar plates inoculated with soil slurries supported greater numbers of bacterial colonies on invaded soils. Overall, our studies suggest that garlic mustard invasions have the potential to shift microbial community structure by selectively increasing some bacterial populations and decreasing fungal populations. Furthermore, garlic mustard invasions may significantly affect microbes involved in nitrogen turnover, suggesting that removal of this invasive species may not be sufficient to restore soil microbial community dynamics and ecosystem function.

INTRODUCTION

The number of invasive species has increased dramatically in nearly every habitat across the world (Baskin 2002). Invasive plant species may be introduced accidentally as seeds in foreign goods or via shipping vessels, or purposely for aesthetic, food, or medical purposes. Garlic mustard (*Alliaria petiolata*) is a herbaceous biennial plant of the Brassicaceae family that was introduced to North America by European settlers in the mid-1800s (Anderson et al. 2010). By 2009, garlic mustard (GM) had been documented in 37 states and 5 Canadian provinces (Natural Resources Conservation Service 2009). Studies have focused on understanding what predisposes areas to invasion and the impact that invasive species have on native ecosystem dynamics once established, with the ultimate goal of creating practical preventive and management strategies to reduce ecological and economic impacts alike (Kolar and Lodge 2001, Rejmánek and Richardson 1996).

Invasive plant species have been found to modify soil physiochemical properties as well as change biotic properties, including microbial community structure and function (Belnap and Phillips 2001, Ehrenfeld et al. 2001). Garlic mustard is no exception. Previous studies have suggested that GM alters soil nutrient conditions in its invaded range by increasing the amount of plant-available nitrogen (Morris et al. 2012, Rodgers et al. 2008), a macronutrient that, if present in only low concentrations, commonly limits plant growth. Additionally, GM is one of the few plant species that do not form mycorrhizal associations due in part to its ability to release allelochemicals as native species root exudates, resulting in reduced germination of mycorrhizal associations and colonization (Roberts and Anderson 2001). A survey of the presence or absence of mycorrhizae in >6,500 angiosperm species conducted by Cronquist (1981), found that 82 percent of species surveyed had

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mycorrhizal associations. Furthermore, GM can destroy fungal associations of native plant seedlings, which are necessary for growth, thus perpetuating its invasion (Stinson et al. 2006). In addition to this unique characteristic, allelopathic glucosinolates have been found to affect germination and growth of native plant species through inhibition of soil mycorrhizae (both ectomycorrhizal fungi and arbuscular mycorrhizal fungi) and rhizobial bacteria (Anderson et al. 2010, Murray et al. 2011, Rodgers et al. 2008). Another chemical defense is the release of cyanide from plant tissues (Cipollini and Gruner 2007).

The goal of this study was to examine the effects of GM on soil microbial community dynamics. Our research was conducted in forest soils with and without a history of GM invasion in a pine (*Pinus* sp.) plantation in central Illinois at Sand Ridge State Forest (SRSF). The sandy texture of the soil makes the area susceptible to nitrogen leaching, which can lead to decreased availability of nitrogen for plant uptake (Lee and Jose 2005). Many areas of SRSF have dense populations of GM, but our ongoing study of this site has confirmed that there are areas that are currently uninvaded (NGM). The ability of GM to change the local soil microbial community may be an important factor in understanding its invasibility, likely mediated through increases in the abundance of free-living nitrogen-fixing bacteria (Ehrenfeld et al. 2001, Wolfe and Klironomos 2005). This relationship may help explain why GM is able to establish itself in undisturbed, nitrogen-poor habitats such as those at SRSF. Because GM has been found to alter microbial community structure and nitrogen availability, GM invasion could potentially affect overall productivity and nutrient cycling in this managed forest.

METHODS

Sampling Site: Sand Ridge State Forest

Three study sites were chosen at SRSF near Forest City, IL, for soil sampling. This forest covers >7,500 acres and is the largest state forest in Illinois. Located in northern Mason County, it contains soils predominantly composed of sandy loam, which was formed at the end of the last ice age. In 1939, the Civilian Conservation Corps, managed by the Division of Forestry, oversaw 5,504 acres of the forest tract as a conservation and experimental area. During this time, 2,200 acres were cleared and converted into a series of pine plantations. Today, SRSF has 7,200 acres and is composed of 3,916 acres of dry land oak-hickory (*Quercus-Carya* spp.) woodlands, 2,492 acres of pine woodlands, and 792 acres of open fields and sand prairie (Illinois Department of Natural Resources 2013).

Three separate pine sites (1, 2, and 3) with similar characteristics were selected at SRSF. At each site, six 2-m² plots were randomly selected in GM-invaded and uninvaded areas (12 total plots per site). The three study sites chosen in the forest met the following criteria and were compatible with our sampling parameters: (1) pine overstory, (2) age >50 years, and (3) a minimum area of relatively uniform plant diversity comprising mainly pine and GM. Our study consists of several semi-independent experiments examining soil microbial communities at SRSF. Therefore, the three sites were used differently depending upon the parameters of each sampling method being tested.

In Situ Soil Carbon Dioxide Efflux

In situ soil carbon dioxide (CO₂) efflux was measured by using a portable infrared LI-8100 analysis machine (LI-COR Inc., Lincoln, NE) at all three sites. Ten-cm-diameter polyvinyl chloride collars were permanently placed in the center of each plot at a depth of 2.54 cm from the top rim and left

undisturbed for ≥ 24 hours before measurement. This “resting time” allowed the removal of any excess CO_2 released by roots damaged during collar placement.

The head used for measurements was a standard LI-COR® 20-cm head and the offset of the program was adjusted to 2.54 cm. A temperature probe was placed in the soil and allowed to stabilize before initiating the program. About 10 minutes before sampling, the collars were cleared of live vegetation and litter to reduce the possibility of disturbance effects. Accurate measurements require that the soil remain undisturbed to prevent the loss of CO_2 from the chamber during readings (LI-COR Biosciences 2012). Removal of the litter ensured that we measured CO_2 levels of the soil within the collars and not from CO_2 being emitted from leaf litter. Measurements from the LI-COR were set at the 10-ppm range. Three complete cycles were run per collar per plot and an average reading was recorded and analyzed by using the LI-8100A soil CO_2 flux system embedded instrument software (LI-COR Biosciences 2012). In situ soil CO_2 efflux was compared as a function of garlic mustard density in each plot.

In Vitro Catabolic Response Profiles

A modified version of the catabolic response profile (CRP) approach was used to measure the catabolic differences of the soil microbial community in plots with and without GM invasion. The method developed by Degens and Harris (1997) analyzes microbial communities' short-term respiratory response to simple organic substrates. In our study, nitrogen and carbon substrates were used to assess patterns of microbial communities at SRSF without the need to extract or culture organisms from the soil. The CRP technique is a functional measure of microbial response that is useful in comparing diversity between communities by examining metabolic response patterns of the organisms to the substrates within a sample (Degens and Harris 1997).

Bulked GM and NGM soils were collected randomly from plots within site 1 to a depth of 5 to 10 cm and divided into two sets of six different treatments with four replicates per treatment. The first set of GM and NGM soils was treated with glucose (3 mg/g) with or without the addition of nitrogen. The second set of treatments used sawdust (4 mg/g) with or without addition of nitrogen. Nitrogen was added as $0.0114 \text{ NH}_4^+ \text{NO}_3^-$ mg/g soil following the protocol outlined by Brewer (2004). Deionized water was used in two treatments with no additional substrates to measure unamended soil respiration rates. The samples were stored in quart-sized jars at 25 °C and CO_2 released was measured after 3, 7, 10, 13, and 20 days of incubation on a Model LI-G252 CO_2 Analyzer (LI-COR).

Substrate-Induced Respiration Using Selective Inhibitors

Developed by Anderson and Domsch (1978), substrate-induced respiration (SIR) measures the amount of microbial respiration of soils after amending them with an excess amount of a soil nutrient, typically glucose, in order to trigger microbial activity (Aira and Dominguez 2010). Soil samples were collected randomly from six GM and six NGM plots at site 1 by using a soil corer 2 cm in diameter. Samples were spread out into two rows of four within a 2 m by 2 m plot frame. Each row of four samples was located about 0.75 m from the top and bottom of the plot frame, respectively. Soils were sieved and refrigerated until use. Before analysis soil samples were split into subsamples and weighed into incubation flasks. Moisture was maintained at field capacity at room temperature for 12 hours.

For each subsample, soils (25 g) were amended with glucose, glucose and cycloheximide (a eukaryotic inhibitor), glucose and streptomycin (a bacterial inhibitor), and glucose and the two inhibitors together. Samples measured for basal respiration were unamended. For glucose, the minimum amount necessary for maximum respiratory response was added to soil. For the inhibitors, the minimum amount necessary for maximum inhibition of respiration was added once the minimum glucose amount had been determined. These values were determined for the soils at SRSF by varying concentrations and measuring respiratory response. Inhibitors and glucose were added dry to soil samples. The soil was mixed by stirring, then immediately sealed. Headspace CO₂ was taken at 0, 1, and 5.5 hours. Glucose was added at a rate of 5.0 mg/g soil, cycloheximide at 8.0 mg/g soil, and streptomycin at 14.0 mg/g soil.

Selective inhibition was carried out as described by Bailey et al. (2002). One hour after the addition of inhibitors, glucose was added at the concentration to achieve maximal respiration (over 6 hours) for each soil sample. The incubation jars were sealed immediately after glucose addition and a CO₂ reading was taken before the jars were placed in the dark at 25 °C. Jars were wrapped in aluminum foil to further reduce any effects of light. Headspace CO₂ was withdrawn through a rubber septum from the jar with a gastight syringe for each treatment at 0 and 5.5 hours and measured on a Model LI-G252 CO₂ Analyzer (LICOR). The ratio of fungal to bacterial activity was calculated as the respiratory inhibition caused by the fungicide, divided by the respiratory inhibition caused by the bactericide at 5.5 hours.

Bacterial Isolation

Soils used to determine the number of culturable bacteria on nitrogen-free media were collected from GM and NGM plots on sites 1 and 2. Bacterial number was evaluated by inoculating soil slurries on nitrogen-free (Thompson-Skerman) agar plates. Serial dilutions were prepared to obtain plates with a countable number of colonies. Nitrogen-free plates consisted of 500 ml of nanopure water, agar (10 g), glucose (5 g), ferrous sulfate heptahydrate (5 mg FeSO₄·7H₂O), calcium carbonate (2.5 g CaCO₃), dipotassium phosphate (0.45 g K₂HPO₄), calcium chloride dihydrate (50 mg CaCl₂·2H₂O), potassium phosphate (50 mg KH₂PO₄), magnesium sulfate heptahydrate (5 mg MgSO₄·7 H₂O), sodium molybdate dihydrate (2.5 mg NaMoO₄·2H₂O), and bromothymol blue (1.1 ml of 5 percent ethanol solution). Plates were incubated at 30 °C for 2 to 7 days and then colony counts were performed. Individual colonies that grew on nitrogen-free plates were reinoculated onto another nitrogen-free plate to ensure introduced nitrogen was not a factor in bacterial growth.

Statistical Analyses

Differences between NGM and GM basal and substrate-induced respiration with and without inhibition were determined by using one-way analysis of variance (ANOVA; PROC MIXED; SAS Institute Inc., Cary, NC). All data were evaluated for and met assumptions of ANOVA. In situ CO₂ flux was evaluated by using regression analysis for each site. In vitro respiration rates did not meet assumptions of ANOVA, so they were analyzed by using a nonparametric one-way ANOVA (PROC NPAR1WAY; SAS Institute Inc.) with mean separations determined from Bonferroni multiple comparisons based on ranked data. For bacterial plate counts on nitrogen-free media, colony-forming units were log transformed and plotted as a function of GM density. Linear regressions were performed for each site and then as a whole.

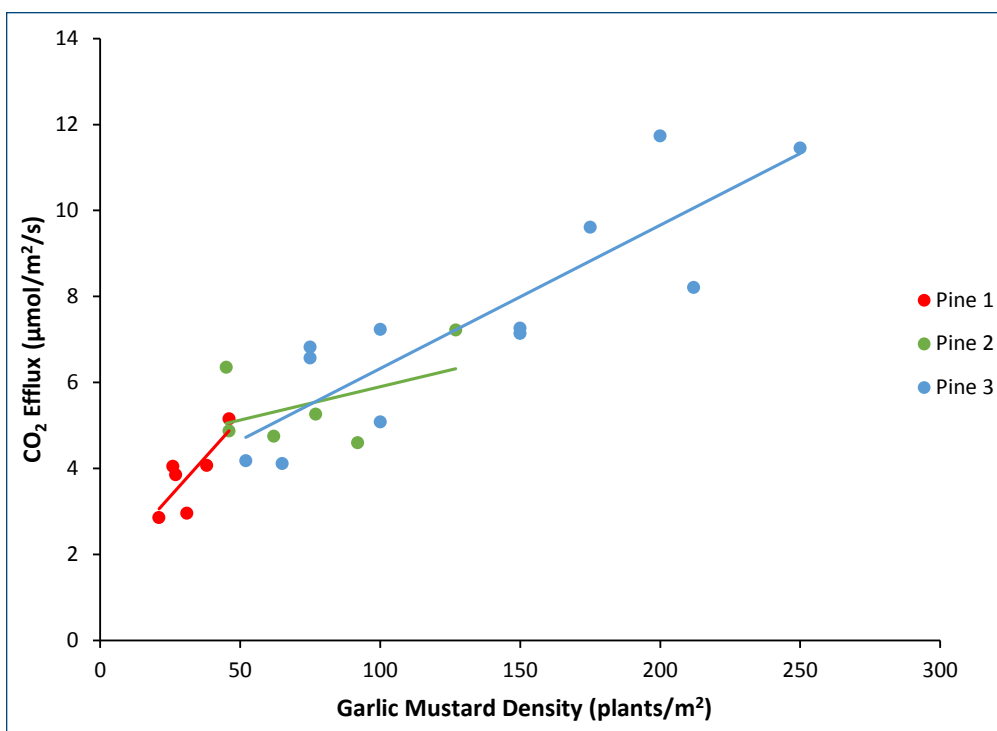


Figure 1.—In situ CO₂ efflux measurements at three different pine sites with varying garlic mustard population densities at Sand Ridge State Forest, Illinois. Pine Site 1: n=6, R² = 0.6146; Pine Site 2: n=6, R² = 0.2148; Pine Site 3: n=12, R² = 0.7617.

RESULTS

In Situ Soil CO₂ Efflux

There was an increase in soil CO₂ efflux as GM densities increased. At two of the three sites (1 and 3) more than 60 percent of the variation in soil CO₂ efflux was explained by the density of GM (Fig. 1).

In Vitro Catabolic Response Profiles

Soil CO₂ efflux was also measured in the laboratory, where the impacts of soil roots would be minimized. Although basal respiration did not differ between GM and NGM soils, soil respiration rates did differ when substrates were added with and without nitrogen. The NGM soils had soil respiration rates similar to GM soils early in the incubation when glucose was added, but only when nitrogen was added with the glucose. Carbon dioxide efflux was significantly greater for GM soils than for unamended NGM soils early in the incubation when we added either glucose with or without nitrogen or sawdust with or without nitrogen. Overall, GM and NGM soils amended with sawdust and sawdust with nitrogen had consistently higher rates of respiration compared to unamended soils, with respiration rates for all treatments declining over the 20-day incubation (Fig. 2). Soils treated with glucose had higher rates of respiration compared to those treated with sawdust (Figs. 2 and 3). Unamended GM and NGM soils did not differ throughout the incubation. Treatments with glucose had higher rates of respiration compared to unamended GM and NGM at all time points (Fig. 3). Early in the incubation (days 3 and 7) GM soils with glucose and with or without nitrogen had higher rates of respiration compared to NGM soils with added glucose. On day 3, NGM soils with added glucose and nitrogen had respiration rates significantly greater than NGM soils amended with only glucose (Fig. 3).

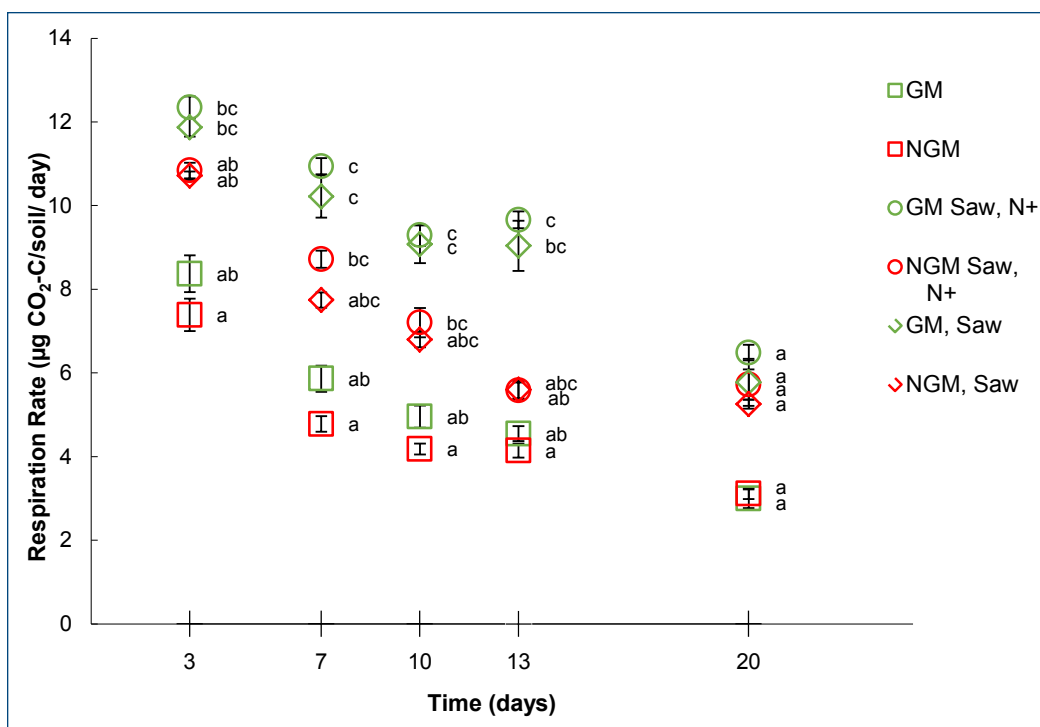


Figure 2.—In vitro respiration rates of garlic mustard and nongarlic mustard soils amended with nitrogen (N+) and sawdust (saw) at 0, 3, 7, 13, and 20 days of incubation at Sand Ridge State Forest, Illinois. Points labeled with the same letter represent means that are not significantly different at $p \leq 0.05$. Error bars represent mean $\pm$ standard error (SE).

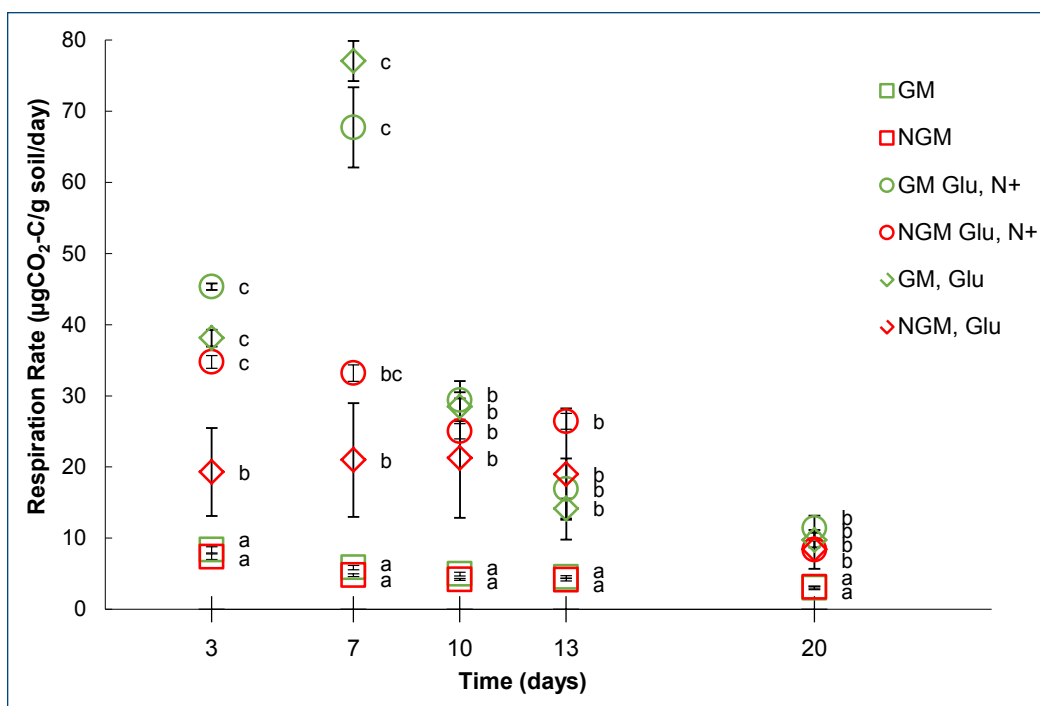


Figure 3.—In vitro respiration rates of garlic mustard and nongarlic mustard soils amended with nitrogen and glucose (glu) at 0, 3, 7, 13, and 20 days of incubation at Sand Ridge State Forest, Illinois. Points labeled with the same letter represent means that are not significantly different at $p \leq 0.05$. Error bars represent mean $\pm$ SE.

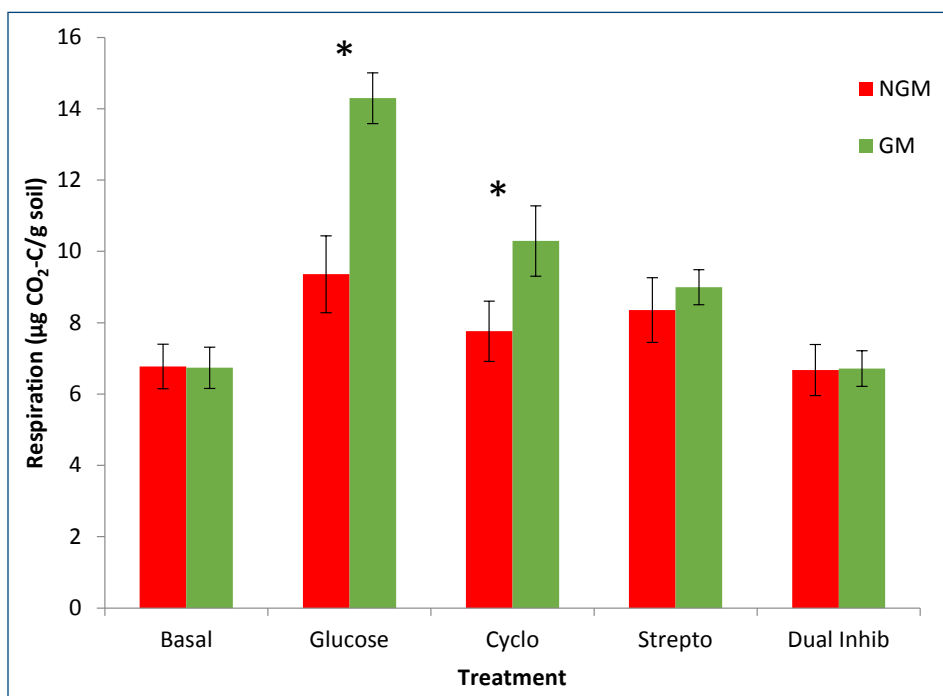


Figure 4.—Substrate-induced respiration after 5.5 hours of incubation in garlic mustard and nongarlic mustard soils at Sand Ridge State Forest, Illinois. Treatments represented are additions of water (basal respiration), glucose, cycloheximide (cyclo), streptomycin (strepto), and cycloheximide and streptomycin (dual inhib) added together. Histogram bars represent mean $\pm$ SE. Asterisks (*) indicate significant difference at $p \leq 0.05$.

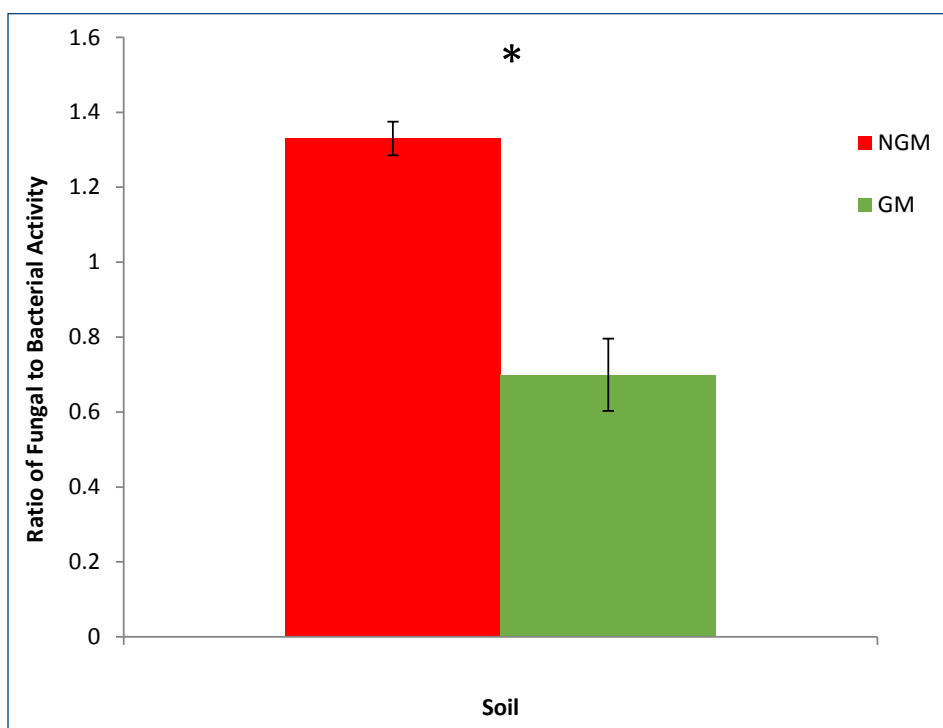


Figure 5.—Ratio of fungal to bacterial activity calculated from fungal inhibition with cycloheximide, and bacterial inhibition with streptomycin after 5.5 hours of incubation in garlic mustard and nongarlic mustard soils at Sand Ridge State Forest, Illinois. Histogram bar represents mean $\pm$ SE. Asterisk (*) indicates significant difference at $p \leq 0.05$.

Substrate-Induced Respiration Using Selective Inhibitors

Microbial respiration in soils treated with glucose was significantly greater for GM soils compared to NGM soils, with rates of about 14 $\mu\text{g CO}_2\text{-C/g soil}$ and 9 $\mu\text{g CO}_2\text{-C/g soil}$, respectively (Fig. 4). After treatment of GM soils and NGM soils with cycloheximide, GM soils had significantly higher respiration rates. Respiration rates for GM and NGM soils did not differ between the other treatments. Based on these values, the calculated ratio of fungal to bacterial activity for NGM soils was significantly greater than for GM soils (Fig. 5).

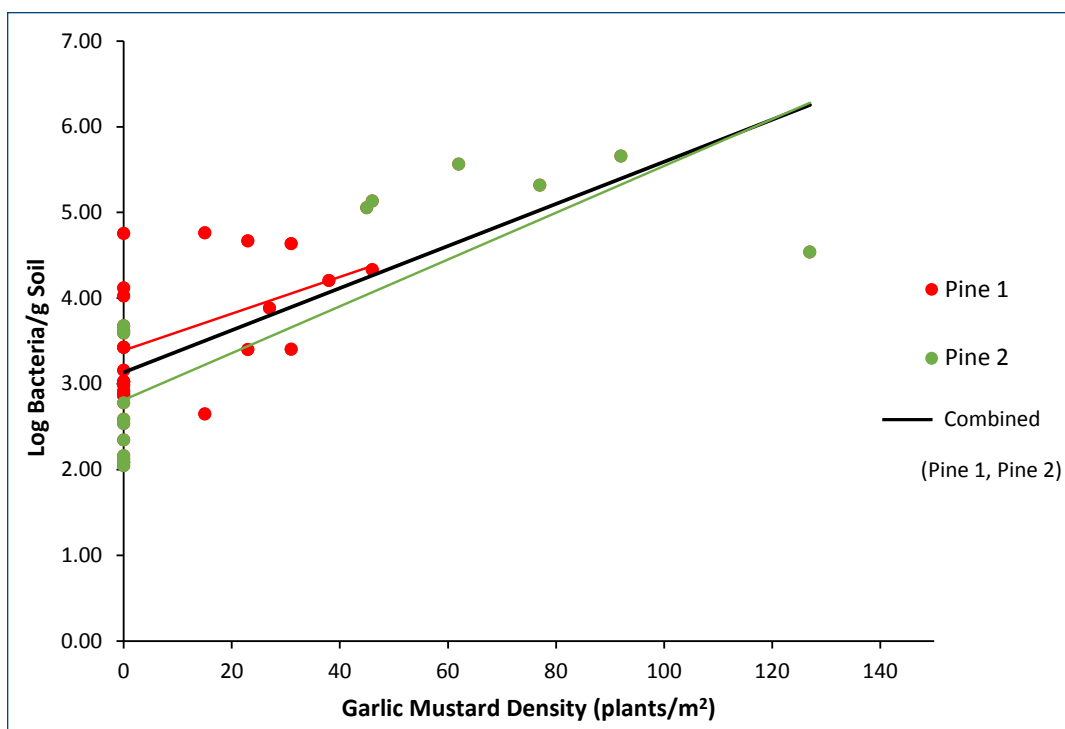


Figure 6.—Relationship of nitrogen-fixing bacteria to garlic mustard density in two selected pine sites at Sand Ridge State Forest, Illinois. Pine Site 1: $n=21$, $R^2 = 0.2178$; Pine Site 2: $n=18$, $R^2 = 0.6391$; Combined Pine Sites 1 and 2: $n=39$, $R^2=0.4831$.

Bacterial Isolation

As GM density increased, there was an increase in the number of bacteria able to grow on nitrogen-free media (Fig. 6). When the sites were analyzed together, nearly 50 percent of the variation in bacterial number was explained by GM density.

DISCUSSION

We hypothesized that GM invasion would alter microbial community dynamics. There are many approaches to evaluating microbial community dynamics. Our first measure at the largest scale was an estimate of soil CO_2 efflux in GM and NGM plots in the field. Soil CO_2 efflux increased as GM density increased across the three sites evaluated (Fig. 1). In situ soil CO_2 efflux reflects contributions from roots as well as from members of the soil microbial community (Hanson et al. 2000). Teasing apart the impacts of GM on the soil microbial community from the impacts of roots or through changes to root dynamics such as increased rhizosphere activity requires an examination of soil CO_2 efflux without the presence of roots.

Soil efflux measured in the lab provided evidence that soil respiration rates did differ when soil was amended with and without nitrogen. These data also support an ample nitrogen supply in the face of a limited carbon supply and an increase in available nitrogen in the presence of garlic mustard that was not observed in NGM soils. Morris et al. (2012) found increased rates of nitrogen mineralization in GM soils compared to NGM soils. These findings indicate that GM has the potential to increase nitrogen availability in soils and also provide a mechanism for the differences in response to carbon substrates in our data.

The response of the microbial community to substrate additions in the GM and NGM soils may also be the result of differences in the composition of the microbial communities. Bacteria typically decompose simpler substrates, whereas fungi decompose complex organic molecules like tannins and lignins (Wall and Moore 1999). This generalization is supported in many experiments that have examined how ratios of substrate carbon to nitrogen favor growth of bacteria compared to fungi, or vice versa (Rousk and Bååth 2007). Therefore, the large increase in respiration observed when soils were treated with glucose in both the CRP and SIR experimental samples suggests an increase in bacteria compared to fungi (Figs. 3 and 4). Alternatively, the defense compounds created and exuded from GM roots may have predisposed the microbial communities to favor glucose use. That the difference in response to substrate additions was a reflection of a change in the composition of the microbial community is also supported by results from the addition of cycloheximide. This addition led to a more significant reduction in respiration in NGM soils compared to GM soils (Fig. 4) and a resultant decrease in the ratio of fungal to bacterial activity in GM soils (Fig. 5). Therefore, there is evidence of changes in the microbial community with GM invasion.

The alteration of soil microbial interactions has also been thought to play an important role in the successful invasion of GM (Rodgers et al. 2008). Studies have found a decrease in the richness and diversity of native bacterial species in the presence of GM (Hammer 2009, Lankau et al. 2011). These changes in diversity may also reflect a change in microbial community function. Nitrogen-free plates allow for identification of bacterial colonies that are likely to fix nitrogen as their sole source of nitrogen. We found a strong relationship between the number of bacterial colonies observed (a proxy for the number of bacteria in the soil sample) and the density of GM on the site sampled (Fig. 6). Rodgers et al. (2008) suggested that GM increases soil nitrogen and that preexisting nitrogen conditions were not a factor in GM invasibility. Taken together with experimental results, their analysis suggests that GM may alter microbial community structure at least partly by increasing the number of free-living, nitrogen-fixing bacteria. These observations are also supported by the substrate addition data (Figs. 4 and 5), which suggest that respiration in GM soils is carbon limited rather than nitrogen limited.

Our results indicate that garlic mustard invasions can have significant impacts on microbial communities, particularly on those microbial guilds involved in nitrogen turnover. There is ample literature to support changes in plant community dynamics as a consequence of increased nitrogen availability in soils (Baer et al. 2004, Davis et al. 2000, De Schrijver et al. 2011). Forest managers charged with ensuring forest integrity should consider that removal of this invasive species may not be sufficient to restore soil microbial community dynamics and consequently ecosystem function. Understanding the effects of invasive species such as garlic mustard on soil microbial communities and subsequent impacts on nutrient cycling can increase our capacity to develop management strategies for addressing specific impacts of invasives on forest systems.

ACKNOWLEDGMENTS

The authors would like to thank Leah Barth, Dan Dreger, Heather Milligan, Dustin Herrmann, Erik Zdansky, and Sam Fan for help with this research. This research was sponsored by National Science Foundation Research Experiences for Undergraduates grants DBI-0755278 (K.D. McConnaughay and S.J. Morris) and DBI-0453285 (K.D. McConnaughay and E.K. Stabenau); Partnerships for Innovation grant EEC-0332714 (S. Liberty, K.D. McConnaughay, and R. Lister); Annie E. Casey Foundation grant 205.0079 (K.D. McConnaughay and R. Wolffe); and a Department of Labor (subcontract to City of Peoria Workforce Development Department) grant (K.D. McConnaughay and E.K. Stabenau).

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WILDLIFE

BAT ACTIVITY IN SELECTION HARVESTS AND INTACT FOREST CANOPY GAPS AT INDIANA STATE FORESTS

Scott Haulton and Kathryn L. DeCosta¹

ABSTRACT

Introduction

Forest managers often prescribe silvicultural methods based on how effectively they mimic the natural disturbance agents that have historically shaped the forests they manage. On Indiana state forests, selection systems are used on most harvested acreage and appear to structurally mimic the effects of naturally occurring, gap-forming disturbances affecting individual trees, groups, or small patches of trees. Forest bats often forage within canopy gaps and along the edges of openings; however, it is unclear whether bats use harvested stands and canopy gaps in intact forest similarly on Indiana's state forests.

Methods

To determine if gap use by bats was related to harvesting, we compared bat activity within naturally occurring gaps in "intact" forest stands (i.e., >15 years since single-tree selection harvest) with activity in recent (i.e., ≤5 years) selection-harvested stands. We used Anabat SD2 (Titely Scientific, Columbia, MO) acoustic detectors to monitor bat activity in randomly located canopy gaps within 15 paired harvested and intact stands at six state forest properties in southern Indiana from June to August 2012. Two to three sites were selected within each stand and a bat detector was stationed at each site at the edge of a canopy gap, with its microphone elevated 2.5-3 m above the ground and directed into the gap. Detectors within paired stands operated simultaneously and were set to begin each night of sampling 30 minutes before sunset and end 30 minutes after sunrise. All detectors were located >200 m from each other and >100 m from stand edge. Recorded bat calls were identified to species group based on call characteristics by using three automated software packages, EchoClass (U.S. Army Engineer Research and Development Center, Vicksburg, MS), Kaleidoscope (Wildlife Acoustics, Inc., Concord, MA), and BCID (Bat Call Identification, Inc., Kansas City, MO). Call files were classified to the species group agreed upon by at least two of the software packages used. Bat activity was quantified for each species group as the number of survey minutes that included at least one classified call file.

Results and Conclusions

Bat activity in recently harvested stands was greater ($P < 0.05$) for two species groups: *Eptesicus fuscus*/*Lasiurus noctivagans*/*Lasiurus cinereus* (low frequency group) and *Lasiurus borealis*/*Nycticeius humeralis*/*Perimyotis subflavus* (mid-frequency group). Activity levels were similar ($P = 0.117$) between stand types for calls characteristic of *Myotis* species. Across selection-harvested stands, the mean number

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of call-minutes per survey hour for low frequency, mid-frequency, and *Myotis* groups was 1.1, 4.0, and 0.7, respectively. Across intact forest stands, the mean number of call-minutes per survey hour for low frequency, mid-frequency, and *Myotis* groups was 0.3, 1.8, and 0.6, respectively. Our results support the use of selection harvesting as a means to improve overall bat foraging activity in mature, intact forest stands, while not adversely affecting the foraging activity of any species group we studied.

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

SWAMP RABBITS AS INDICATORS OF OPTIMAL SCALE FOR BOTTOMLAND FOREST MANAGEMENT

Joanne C. Crawford, Clayton K. Nielsen, Eric M. Schaubert, and John W. Groninger¹

ABSTRACT

Introduction

Specialist wildlife that evolved within forest ecosystems can be sensitive to disturbance regime changes and thereby serve as indicators of optimal scale for forest management. Bottomland hardwood (BLH) forests were once extensive within the Lower Mississippi Alluvial Valley, but land cover conversion has reduced BLH by about 80 percent over the last century. Since 1990, a multiagency effort has been underway to restore BLH forests, with the preservation of wildlife habitat as a primary goal. Habitat loss, coupled with changes in disturbance patterns, has been associated with a decline of the swamp rabbit (*Sylvilagus aquaticus*), a bottomland specialist, throughout the southeastern United States. Afforestation has created a patchwork of early successional stands embedded within a larger agricultural landscape and adjacent to mature riparian stands.

Swamp rabbits rely on habitat components characteristic of both early and mid to late successional BLH stands, but also need upland habitat during periods of prolonged inundation. The small home ranges (HRs) typical of rabbits suggest that all of these habitat components must be within close proximity to one another. Here, we address how the spatial arrangement of BLH plantings influences habitat use by swamp rabbits in southern Illinois. Specifically, we examined the influence of microhabitat and cover type on space use and HR size for 46 swamp rabbits monitored from 2009 through 2012 at seven sites within the Cypress Creek National Wildlife Refuge in southern Illinois.

Methods

Study sites were composed of early and mid to late successional BLH forests with varying amounts of adjacent early successional uplands and agriculture. Early BLH stands were afforested within the last 20 years and were dominated by oaks (*Quercus* spp.), green ash (*Fraxinus pennsylvanica*), boxelder (*Acer negundo*), and sweetgum (*Liquidambar styraciflua*). Tomahawk live traps (Model 205; Tomahawk Live Traps, LLC, Hazelhurst, WI) were placed in areas with rabbit sign, covered with burlap and vegetation, and baited with a quartered apple. We weighed the captured rabbits, determined their sex, and ear-tagged them. We fitted all adult rabbits >1.9 kg with radiocollars weighing 35-42 g. Animals were released immediately after processing at the site of capture. Swamp rabbits were located by triangulation or homing 2-4 times/week during morning (0500-0900 hr), daytime (0901-1759 hr), and evening (1800-2400 hr) sampling periods. We used kernel density estimators to delineate HRs for all animals with >30 radiolocations.

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We conducted microhabitat sampling at all sites May 15-August 15 2012. Sampling was restricted to an area encompassing all radiolocations for rabbits monitored at each site. We sampled randomly placed circular plots (0.02 ha) at a density of 2.5 plots/ha at each site. Within plots, we recorded the proportion of grasses, forbs, and woody ground cover, canopy closure percentage, and basal area. We also recorded the number of trees <10 cm diameter at breast height in three 1-m² subplots within plots. Visual obstruction was recorded up to a height of 1.5 m within subplots.

We digitized land cover patches in ArcGIS® 9.3 (Esri, Redlands, CA) by using 2010 color orthoimagery (National Agriculture Imagery Program, administered by the U.S. Department of Agriculture's Farm Service Agency). Patches were classified as early BLH, late BLH, upland forest or grassland, agriculture, river, or permanent wetland. We also verified land cover assignments by using Google Earth™ and prior knowledge based on trapping and vegetation sampling. Proportion of each cover type within core HRs was calculated in Patch Analyst (Esri), and the distance from each rabbit's core HR center to the nearest patch was measured for all cover types within a 2-km circular buffer.

Results And Discussion

We built two sets of generalized linear models to assess importance of microhabitat and land cover variables on HR size. Model significance was assessed at $\alpha = 0.05$, and results are reported $\pm$ one standard deviation. Average core and 95-percent HRs were 1.9 ± 1.0 and 8.9 ± 2.4 ha, respectively. The majority of swamp rabbits (63 percent) had core HRs in afforested BLH stands. Most rabbits were within 400 m of a river or mature BLH stand, even at sites with extensive early successional upland forest adjacent. Home range size was associated with few microhabitat or land cover variables; core area was weakly positively associated with distance to a permanent wetland ($\beta = 0.001 [\pm 0.0002]$, $p < 0.000$) and average basal area within core areas ($\beta = 0.006 [\pm 0.0002]$, $p < 0.010$).

Conclusions

Swamp rabbits have relatively small home ranges yet require a range of bottomland forest structures. In young stands, crop tree release may create or maintain dense groundcover conditions needed within the small areas used by this specialist species. Older forests may need more intensive canopy removal to mimic natural blowdown or senescence patterns. Implications for implementing these interventions across multiple agency land holdings are discussed.

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

IMPACT OF HABITAT TYPE ON FORAGE QUALITY OF SEEDLING OAK LEAVES IN CENTRAL WISCONSIN

Michael Demchik¹

Abstract.—The objectives of this study were to determine if relative feed value or crude protein in seedling oak leaves was different between three central Wisconsin habitat type groupings. Seedling oak leaves of two species were collected from oak sites that represented either fully stocked or understocked conditions from three possible habitat type groupings: (1) *Acer-Tilia-Fraxinus/Circaea/Acer-Quercus/Viburnum-Geranium* variant, (2) *Acer rubrum/Desmodium*, and (3) *Pinus/Euphorbia/Pinus/Vaccinium-Gaultheria*. The leaves were analyzed for crude protein, acid detergent fiber, neutral detergent fiber, and relative feed value. Northern pin oak (*Q. ellipsoidalis*) had higher levels of crude protein and relative feed value than northern red oak (*Q. rubra*). *Pinus/Euphorbia/Pinus/Vaccinium-Gaultheria* sites had the highest crude protein and the lowest relative feed value, although the relative feed value was still very high. Generally, oak appears to have very good forage characteristics.

INTRODUCTION

The population of white-tailed deer (*Odocoileus virginianus*), hereafter referred to as deer, is unsustainably high in some parts of Wisconsin (Wisconsin Department of Natural Resources 2011). Preferential deer browse on oak (*Quercus* spp.) can result in undesirable species shifts under some stand conditions (Strole and Anderson 1992, Stromayer and Warren 1997). These changes can be large enough that deer are referred to as a “keystone herbivore” (Waller and Alverson 1997). Species shifts are the result of suppression of oak seedling growth below that of competitors that are not as desirable to deer (Russell et al. 2001).

Difference in palatability of species alters feeding preference by deer (Strole and Anderson 1992). Research on oak palatability to livestock and wildlife generally shows that oak (and other tree species) contains higher levels of crude protein (CP) than many other types of forage during some portions of the year (Forwood and Owensby 1995). Levels of CP in oak have been shown to be 11 percent (*Q. infectoria*; Parlak et al. 2011), between 6.7 and 14.5 percent based on time of year (*Q. coccifera*; Koukoura 1988), and 6.3 to 10.3 percent (*Q. macrocarpa*; Forwood and Owensby 1995). Some of the higher CP levels of the year (Forwood and Owensby 1995, Parlak et al. 2011) occur in August, the sampling time for this project.

In Wisconsin, oak is easier to regenerate on poor quality sites (Demchik et al. 2013, unpublished survey²; Schwartz and Demchik 2012) than on good quality sites (Johnson et al. 2002). As a result, high quality oak has undergone a long-term decline in Wisconsin (Perry et al. 2008). The stronger competitive advantage of oak on poor sites may partly explain this difference, but it is also possible that the forage is less palatable on poor quality sites. The objective of this study was to determine if

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² Demchik, M.C.; Kurszewski, K.F.; Johnson, K.N.; Schwartz, K.M. 2013. Survey of northern and central Wisconsin foresters regarding oak silviculture. Unpublished survey on file with author.

relative feed value (RFV; an index of forage quality) or CP of seedling oak leaves differed between three central Wisconsin habitat type groupings.

METHODS

Available sites on publicly owned property were selected for this study to represent at least three fully stocked and three understocked sites for each of three central Wisconsin habitat type groupings (Kotar and Burger 1996):

- **ATiFrCi** (*Acer-Tilia-Fraxinus/Circaea*)/**AQVb-Gr** (*Acer-Quercus/Viburnum-Geranium* variant),
- **ArDe** (*Acer rubrum/Desmodium*), and
- **PEu** (*Pinus/Euphorbia*)/**PVG** (*Pinus/Vaccinium-Gaultheria*)

Generally, stand quality would be, from best to worst, ATiFrCi/AQVb-Gr > ARDe > PEu/PVG. We combined ATiFrCi with AQVb-Gr and PEu with PVG for reasons detailed in Schwartz (2012). Criteria for site selection were that the stand was ≥ 50 years old and >2 hectares in size, and had >40 percent stocking of overstory oak (see Schwartz 2012 for details; Table 1). On each site, 30 leaves from seedling oaks were collected August 12-16, 2012. Only one leaf per seedling was harvested to avoid unnecessarily damaging the seedlings and to get a more representative sample. Species of oak leaf harvested was based on those available on the site. Within a site, species of leaf harvested was the same (Table 1). The samples were oven dried at 66 °C for 4 days. Chemical analyses for CP, acid detergent fiber (ADF), neutral detergent fiber (NDF), and RFV were completed by DHIA Laboratories in Minnesota. Unbalanced analysis of variance in Minitab[®] (Pennsylvania State University, State College, PA) was used for analysis of data.

RESULTS

Overstory stocking (high or low stocking) had no significant impact on any parameter of foliar nutrition of the leaves. Therefore, this factor was eliminated. For both CP and RFV, habitat type ($P = 0.02$, $P = 0.007$; Table 2) and species ($P = 0.03$, $P = 0.01$, Table 3) were significantly different; however, the interaction of habitat type and species was not significantly different. PVG/PEu had higher CP than the other habitat types, yet lower RFV than ArDe but not ATiFrCi/AQVb-Gr. Northern pin oak (*Q. ellipsoidalis*) leaves had higher CP and higher RFV than northern red oak (*Q. rubra*). Because ADF and NDF are used to calculate RFV, statistical comparisons were not made for those two factors; however, the data are included in Tables 2 and 3.

Table 1.—Species of oak and number of sites in each of three central Wisconsin habitat type groupings, 2012: ATiFrCi (*Acer-Tilia-Fraxinus/Circaea*)/AQVb-Gr (*Acer-Quercus/Viburnum-Geranium* variant), ArDe (*Acer rubrum/Desmodium*), and PEu (*Pinus/Euphorbia*)/PVG (*Pinus/Vaccinium-Gaultheria*)

Habitat type	Species	Number of sites
PEu/PVG	<i>Q. ellipsoidalis</i>	5
	<i>Q. rubra</i>	5
ArDe	<i>Q. ellipsoidalis</i>	3
	<i>Q. rubra</i>	4
ATiFrCi/AQVb-Gr	<i>Q. ellipsoidalis</i>	3
	<i>Q. rubra</i>	6

Table 2.—Crude protein (CP), acid detergent fiber (ADF), neutral detergent fiber (NDF), and relative feed value (RFV) for oak seedling leaves collected in 2012 from three central Wisconsin habitat type groupings: ATiFrCi (*Acer-Tilia-Fraxinus/Circaea*)/AQVb-Gr (*Acer-Quercus/Viburnum-Geranium* variant), ArDe (*Acer rubrum/Desmodium*), and PEu (*Pinus/Euphorbia*)/PVG (*Pinus/Vaccinium-Gaultheria*)

Habitat types	Plots	CP (%)	ADF (%)	NDF (%)	RFV
PEu/PVG	10	12.0±0.7	31.2±2.0	46.1±2.1	131.3±9.1
ArDe	7	10.9±0.7	28.4±2.3	38.7±3.3	162.9±17.9
ATiFrCi/AQVb-Gr	9	10.6±0.6	30.5±1.7	42.2±2.1	144.8±10.6

Table 3.—Crude protein (CP), acid detergent fiber (ADF), neutral detergent fiber (NDF), and relative feed value (RFV) for *Q. ellipsoidalis* and *Q. rubra* seedling leaves collected in central Wisconsin, 2012

Species	Number of plots	CP (%)	ADF (%)	NDF (%)	RFV
<i>Q. ellipsoidalis</i>	11	11.8±0.7	28.7±1.8	40.3±2.1	155.0±18.3
<i>Q. rubra</i>	15	10.7±0.4	31.3±1.3	44.6±2.1	136.8±21.9

DISCUSSION

One of the original reasons that I decided to complete this short study was that we have seen oak successfully recruit on poor quality sites (Schwartz 2012, Schwartz and Demchik 2012); however, oak on richer sites struggles to succeed (Demchik et al. 2013, unpublished survey²). One possible reason was that the nutritional quality was lower, resulting in lower preference by deer. This study does not seem to support that explanation. Crude protein was higher on PVG/PEu sites than on ArDe and ATiFrCi/AQVb-Gr sites, and even though RFV is lower on PVG/PEu sites, the quality is still very high (Jeranyama and Garcia 2004). When RFV was calculated for oak leaves from the data in two other papers that had measured oak ADF and NDF, gall oak (*Q. infectoria*) in Turkey (Parlak et al. 2011) showed a range of RFV from 82 to 96 (depending on sampling month) and Pyrenean oak (*Q. pyrenaica*) in Spain showed a range of 124 to 163 (Ammar et al. 2010). Our data showed higher RFV than Parlak et al. (2011) but values comparable to Ammar et al. (2010). Although levels of tannins were not measured (high tannin levels can reduce forage usability to livestock and wildlife), this level of RFV is comparable to good quality cattle forage. The high RFV and CP on the habitat type with the worst site quality (PVG/PEu) suggest that forage quality issues are not an important factor in the higher levels of success in regeneration seen on PVG/PEu sites. Furthermore, these data suggest that the preferential deer browsing on oak is due to high overall palatability of the genus.

Also surprisingly, northern pin oak had both higher RFV and higher CP than northern red oak. As the interaction of species by habitat types was not significant, the uneven representation of northern red oak and northern pin oak on different habitat types was not the origin of this effect. The sample year, 2012, was a severe drought year in central Wisconsin; this drought may have had a disproportional impact on northern red oak. Curtis (1959) ranked northern red oak as less drought tolerant than the other associated oak species. The impact of northern red oak's inferior drought tolerance when compared to northern pin oak (a very drought-adapted species) is impossible to determine from our data set. Northern red oak may simply have lower forage value than northern pin oak or it may be a relic of the exceptionally dry summer of the sample year.

Overall, it appears that oak's higher success at regenerating cut sites on PVG/PEu sites when compared to other habitat types is not related to the forage value of the leaves. Oak on all three habitat type groupings were very high in RFV and CP.

ACKNOWLEDGMENTS

I wish to thank the Wisconsin Department of Natural Resources and county foresters for allowing site access for this work. I would like to thank Tyrone and Tess Radke, who assisted with sample collection and Kevin Schwartz, who classified these sites by habitat type as part of another project. I thank Sophie Demchik for the comments and editing. The funding for this study was provided by McIntire Stennis Cooperative Research Grants.

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IMPACT OF DEER BAIT SITES ON *PEROMYSCUS* MICE IN SOUTHERN ILLINOIS

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and Clayton K. Nielsen¹

ABSTRACT

Introduction

Wildlife populations are heavily influenced by food availability and predation rates. Changing the distribution of high quality food sources can often alter the distribution of wildlife populations. In particular, increases in food abundance can result in immigration; earlier breeding; and increased productivity, survival, and density of wildlife (Morris et al. 2011). Increases in prey abundance, however, may also lead to increases in predator densities, which can in turn decrease survival and increase emigration (Cooper and Ginnett 2000, Hamilton et al. 2002). Furthermore, predation risk can cause sublethal changes in prey behavior such as different habitat use, increased vigilance, decreases in home range size, and decreased reproductive output (Lima and Dill 1990).

Establishing bait sites for studying and hunting game species is a common practice, but nontarget species such as deer mice (*Peromyscus maniculatus*) may also be attracted to and consume this supplemental food (Rollins 1996). As granivores, *Peromyscus* species provide important ecological services such as distributing seeds and serving as prey for a wide variety of predators. These mice have short generation spans that allow them to respond quickly to changes in their environment. Because bait tends to be more ephemeral (days) than naturally occurring concentrated food sources, such as fruiting trees (months), monitoring small mammal communities may show how quickly they can respond to changes in food abundance. Knowledge on how *Peromyscus* mice respond to these changes may help predict how other similar and endangered species may respond in these situations. Furthermore, a new spatial arrangement of prey may trigger other species to respond, causing additional changes in population parameters. Our objective was to characterize the potential impacts of short-term bait sites on *Peromyscus* population parameters.

Methods

Touch of Nature Environmental Center (Makanda, IL) contains more than 1,200 ha in the southern part of the state. We conducted a paired-grid study at this location on a 50-ha forested site dominated by oaks (*Quercus* spp.) and hickories (*Carya* spp.). Two plots were established about 700 m apart so that mouse home ranges would not overlap but forest composition would be similar between the plots. We monitored for white-tailed deer (*Odocoileus virginianus*) to make sure they were present on each plot. We placed Sherman traps (H.B. Sherman Traps, Inc., Tallahassee, FL) 10 m apart on an 8 by 8 grid in each plot. Traps were baited with bird feed, and we trapped for 3 nights

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each week for 9 consecutive weeks in August-October 2012. We applied a uniquely numbered tag (5 mm high by 11 mm long; National Band and Tag Co., Newport, KY) to each captured *Peromyscus* mouse and recorded age, sex, and capture location.

During weeks 4-5, we maintained 2.3 kg of corn at the center of the grid in one plot, similar to what would be used for research or by hunters. We monitored for deer use of the bait pile. We used weekly mark-recapture data of *Peromyscus* mice to run Huggins robust-design models in the RMark package in program R. We created a priori models to determine effects of bait, session, time, age class, and sex. We ranked models by using the second-order Akaike information criterion (AIC_c) and considered any model within 2 Δ AIC_c to be competing.

Results And Discussion

Our top model included age class and an interaction between time and session in both capture and recapture parameters. Capture probabilities ranged from 0.30 to 0.57 during the prebait session, 0.54 to 0.63 during the bait session, and 0.37 to 0.90 during the postbait session. Recapture probabilities were 0.50-0.66 during the prebait session, 0.56 in the bait session, and 0.10-0.45 during the postbait session.

We found that age and an interaction between time and session were important factors influencing capture and recapture rates. Capture rates increased with time, and recapture rates decreased with time. These results are most likely indicative of a population with low immigration rates: as time progressed, the number of unmarked individuals decreased and the number of marked individuals increased within the relatively stable population. Adults had higher capture and recapture rates than juveniles, but sex had no effect in any of our top models. This age bias may be associated with drought conditions during summer 2012 that may have reduced survival and reproductive rates and therefore decreased the number of young within the population during our study (Yahner 1992). We found that the bait pile did not influence *Peromyscus* survival, emigration, or immigration. Mast availability in the study area may have buffered the effects of concentrated food on *Peromyscus* populations. The short timeframe of 2 weeks of baiting may have also buffered the population from predators who may be unable to pinpoint this potential food concentration at such a timescale (Boutin 1990).

Conclusions

Our study provides preliminary support that short-term baiting does not affect *Peromyscus* mice population parameters in southern Illinois. Future studies should be conducted in different habitats and seasons to better understand responses of populations in those areas. Altering the durations of concentrated food sources may cause changes in population parameters if *Peromyscus* or predators have more time to key in on these food concentrations and prey locations.

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A REVIEW OF BEST MANAGEMENT PRACTICES AND THE MITIGATION OF STREAM-BREEDING SALAMANDERS IN THE EASTERN DECIDUOUS FOREST

Thomas A. Maigret and John J. Cox¹

Abstract.—Timber harvest has been implicated as a causative factor in the decline of amphibian populations and diversity in many areas of the world. The adoption of best management practices (BMPs) is intended to minimize the impacts of timber harvest on the biotic community, including amphibians and their habitat requirements. Herein, we synthesize the current scientific literature on the impact and effectiveness of BMPs in preventing population declines of stream-breeding salamanders and associated habitat loss in the eastern deciduous forests of North America. We frequently found sedimentation to be a suspected cause of population declines; many studies also described a correlation between basal area and salamander abundances. In addition to summarizing research, we offer recommendations to improve the efficacy and implementation of BMPs in the central and eastern United States. We also make suggestions for further research, such as increased testing of BMP methods and an increase in long-term studies.

INTRODUCTION

In the past half century, numerous amphibian species have been experiencing significant declines, a worldwide phenomenon that has been interpreted as indicative of a global decline in ecosystem function (Blaustein and Bancroft 2007, Houlihan et al. 2000). Chief among the primary threats to amphibian populations is habitat loss and degradation, especially among species associated with freshwater wetlands or riparian areas (Lannoo 2005, Semlitsch 2000). Timber harvest has been implicated in the degradation of amphibian habitat, including freshwater habitats (Corn and Bury 1989, Vesely and McComb 2002).

Over the past few decades, a greater awareness of the negative effects of timber harvests has resulted in numerous efforts to improve regulations pertaining to logging. Usually termed best management practices (BMPs), these regulations have been repeatedly revised to incorporate new areas of concern such as freshwater fisheries protection or wildlife habitat, but also to reform and target specific practices that have the most demonstrated negative ecosystem impacts.

Streamside management zones (SMZs) are an example of a BMP strategy to help maintain the integrity of freshwater riparian and headwater stream habitats, which are generally more susceptible to damage from timber harvest than terrestrial ones (Chizinski et al. 2010). Although originally designed to protect water quality, the goal of many of these regulations has been expanded to incorporate both abiotic and biotic ecosystem components. These management zones rely on use of riparian buffer strips or stringers to reduce sedimentation into streams, help reduce postharvest increases in stream temperature and light intensity, and provide both core riparian habitat and

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corridors for forest-dwelling species. Consequently, SMZs have become one of the most important components of BMPs used to prevent or reduce expected post-timber harvest declines of aquatic biota, including stream salamanders, defined as those species that require freshwater streams for part of their life cycle (Maigret 2013).

Of all the biota present in headwater streams, stream salamanders can be considered among the most important for ecosystem function. In many deciduous forests they are the most abundant vertebrates, with a biomass equal to or exceeding that of other vertebrate taxa including birds and mammals (Burton and Likens 1975, Petranka and Murray 2001). Stream salamanders also play important roles in nutrient cycling and energy flow (Petranka 1998); as predators of detritivores, they can influence leaf litter accumulation rates and potentially carbon cycling (Wyman 1998). Additionally, stream salamanders have been suggested to be suitable bioindicators of ecosystem integrity (Welsh and Ollivier 1998).

TIMBER HARVEST IMPACTS ON STREAMSIDE SALAMANDER POPULATION DYNAMICS

Throughout the past few decades, several studies have sought to determine how timber harvest affects stream salamander populations. Using a research design framework, we present a review of these research findings from the eastern deciduous forests of North America, most of which were focused on the family Plethodontidae.

Stand Age Studies

One method commonly used in studies evaluating timber harvest impacts on salamanders involves determining the relationship between timber stand age (time since harvest) and resident salamander populations. This method has many benefits, including a streamlined logistical structure, ease of replication, and the convenience of omitting the installation of a treatment midway through data collection. However, the statistical inferences drawn can be more tenuous than from studies using a control-impacted (CI) or before-after control-impacted (BACI) design.

Over the past few decades, many studies have evaluated stream salamander populations in stands of assorted ages (Connette and Semlitsch 2013, Crawford and Semlitsch 2008, Ford et al. 2002, Lowe and Bolger 2002, Lowe et al. 2004); all found that stand age was positively associated with stream salamander abundance. Lowe and Bolger (2002) studied populations of *Gyrinophilus porphyriticus* in first-order streams of a mixed softwood-hardwood forest in New Hampshire by using a stepwise multiple regression analysis. Their best supported model for predicting *G. porphyriticus* abundance contained the following factors: (1) the presence of brook trout (*Salvelinus fontinalis*), (2) a variable representing a combination of years since harvest and substrate embeddedness by sediment, and (3) landscape configuration, defined as whether the streams were paired (a stream with a confluent first-order stream) or isolated. It is also worth noting that they found years since harvest and substrate embeddedness to be highly autocorrelated. Substrate embeddedness was defined by Lowe et al. (2004) as “a measure of fine sediment accumulation among the larger substrate particles of the streambed.”

Ford et al. (2002) calculated abundance, species richness, and a Shannon's diversity index for salamanders in Appalachian cove hardwood stands of northern Georgia. Using an analysis of covariance, they found species richness and Shannon's diversity to be higher in stands ≥ 85 years old than in stands aged 15, 25, or 50 years. Abundance was positively correlated with stand age in *Desmognathus aeneus* and *D. ocoee*. Stand age was found to be significantly associated with abundances of *Eurycea bislineata*, but abundances peaked in 25-year-old stands. Species richness, Shannon's diversity, and abundances of *D. ocoee* and *D. quadramaculatus* were also found to be positively correlated with basal area although abundances of *E. bislineata* were negatively correlated with basal area.

Lowe et al. (2004) sought to investigate stage-specific effects of logging-associated sedimentation and *S. fontinalis* presence on *G. porphyriticus* in mixed hardwood-softwood forests of New Hampshire. Using multiple regression analysis, they determined that larvae abundance was negatively related to the presence of *S. fontinalis*, but unrelated to substrate embeddedness. Conversely, adult abundance was primarily associated with substrate embeddedness. They concluded that in streams where fish are absent and with high levels of salamander larvae, salamander populations may be buffered from logging-associated sedimentation. In streams with fish and corresponding low larval abundances, the reduction in adult survival associated with sediment inputs from logging may threaten salamander persistence.

Crawford and Semlitsch (2008) examined salamander populations of riparian forests adjacent to headwater streams in western North Carolina and found abundances of *E. wilderae* to be significantly reduced in stands ≤ 40 years old compared to stands 41-80 years old and those ≥ 81 years old; this was true for both night and daytime sampling. Additionally, they found that terrestrial habitat usage was significantly greater with increased stand age for *D. monticola*, *D. ocoee*, and *E. wilderae*. Out of numerous covariates, only leaf litter depth was found to have a significant association with salamander abundance.

Connette and Semlitsch (2013) examined populations of *D. ocoee* and *E. wilderae* in streams with different histories of timber harvest. In accordance with previous research, they established a positive relationship between timber stand age and salamander abundance. A negative association between abundance and distance to forest edge was found as well. Additionally, it was determined that the populations increased faster in younger regenerating forest stands than in forest stands closer to climax. Salamander abundance was estimated to reach a peak at 100 years postharvest.

Paired Analyses

Other research has been designed to reflect paired watershed analyses, whereby separate watersheds or other land divisions with different logging treatments are studied. These watersheds often incorporate different SMZ characteristics and include comparative control watersheds within a CI study design. A CI study design presents the advantage of generating comparisons dependent on treatment, but requires the assumption that population levels have not fluctuated temporally due to an uncontrolled variable. Research using a CI design to investigate timber harvest effects has also been widely practiced (Knapp et al. 2003, Moseley et al. 2008, Peterman and Semlitsch 2009, Peterman et al. 2011).

Knapp et al. (2003) conducted research in mixed mesophytic forests of southwestern Virginia and southern West Virginia and found that gravid *D. ochrophaeus* females weighed more in uncut treatments than in treatments subjected to timber harvest. However, no differences in number of gravid females or in number of eggs per gravid female were found.

Moseley et al. (2008) found that abundance of *Desmognathus* spp. was affected negatively by increased removal of timber volume, and was positively associated with increased time since disturbance in a northeastern West Virginia mixed mesophytic hardwood forest. They further concluded that higher percentages of imbedded substrate were associated with lower abundances of *Desmognathus* spp. Their conclusion is shared by Peterman and Semlitsch (2009), who focused on *E. wilderae* and *D. quadramaculatus* in forests of western North Carolina. In that study, larval *E. wilderae* abundances in streams with unharvested forest buffer widths of 0 m, 9 m, and 30 m were compared along with unharvested control streams. Abundances in the streams with a 9-m buffer were not significantly different from 0-m buffer streams, and abundances in streams prescribed a 30-m buffer were not significantly different from the control. However, the 9-m buffer was found to have significantly fewer salamanders than the 30-m buffer and the control. A corresponding pattern in sedimentation was noted, and suggested as the mechanism of decline due to the filling of interstitial spaces used as refugia for larval salamanders.

In examining five western North Carolina forest headwater streams, Peterman et al. (2011) found significantly higher densities of adult and juvenile *D. monticola* and *D. quadramaculatus* at a treatment without a stream buffer, and substantially higher densities at sites with a 9-m unharvested buffer in comparison to unharvested controls. They considered their findings to be attributable to evacuation, characterized by the abandonment of logged areas by stream salamanders in search of more suitable habitat.

Before-After Control-Impacted Design

A BACI study approach is often used to determine the impact of perturbation events on one or more biological or nonbiological entities. These studies are frequently preferred to CI designs because BACI analysis can alleviate the chance that unmeasured covariates are influencing observed effects (McDonald et al. 2000). Considering that salamander populations are often distributed unevenly on a landscape (Wyman 1988), an assumption of pretreatment site homogeneity could potentially weaken experimental conclusions (deMaynadier and Hunter 1995). However, the design can be subject to flaws: the life histories of different salamander species can dictate how much time must have passed between the installation of a treatment and data collection to ensure the integrity of conclusions applied to diverse age classes.

The BACI approach has been used in several studies of timber harvest impacts on stream salamander populations (Maigret 2013, Perkins and Hunter 2006, Williams et al. 2002). Statistical inference from these studies can often be improved over other designs, but they typically are more expensive and time consuming (deMaynadier and Hunter 1995) and often require timber harvests to be tailored to specific study designs. Consequently, BACI projects can be difficult to implement and maintain over long study periods and across multiple funding and data-collecting entities.

Using a BACI study design, Williams et al. (2002) found that a population of *D. brimleyorum* was significantly lower after a removal of 0.04 ha of vegetation from a stream bank in the Ouachita Mountains of eastern Oklahoma. They also noted a large increase in the amount of sediment in the stream but did not measure it.

Perkins and Hunter (2006) conducted a study in a northern hardwood to mixed-conifer-hardwood Maine forest, and found abundance of *E. bislineata* in a partial timber cutting to be more similar to unharvested control streams than clearcuts mitigated by 25- to 35-m stream buffers. The similarity was not significant, however.

Conducting research in ephemeral headwater streams of a mixed mesophytic forest of southeastern Kentucky, Maigret (2013) used a BACI study design with three different treatments: a no-harvest control, a set of watersheds where the current State BMP requirements for SMZ application were applied, and a set of watersheds where intermittent stream SMZ requirements were applied to ephemeral streams under augmented requirements. Maigret (2013) found lower abundances of *Desmognathus* spp. in watersheds subjected to current State BMP requirements for SMZ application, and further found that the treatment where an augmented SMZ was applied was not statistically different from the control. However, these results were found only in a treatment versus control watershed comparison. When preharvest data were compared to postharvest, the abundances followed similar patterns but lacked statistical significance. Similar patterns were found when analyzing *E. bislineata* abundances, but they were not significant in CI or BACI comparisons.

Identifying Cause-Specific Sources of Stream Salamander Declines

Of the studies we reviewed, only a surprisingly small variety of likely causes of stream salamander population decline from timber harvest were described. Logging-associated sedimentation was one factor in many of these studies that was found or hypothesized to be negatively associated with salamander abundance (Crawford and Semlitsch 2008, Lowe and Bolger 2002, Lowe et al. 2004, Moseley et al. 2008, Peterman and Semlitsch 2009). Stand age was identified as being associated with higher abundances of salamanders (Crawford and Semlitsch 2008, Ford et al. 2002, Lowe and Bolger 2002,) as well as higher Shannon's diversity and species richness (Ford et al. 2002), and with increased terrestrial habitat usage (Crawford and Semlitsch 2008). Stand age is a factor that was demonstrated to be associated with numerous other microhabitat factors important to stream salamander populations, including leaf litter depth and soil moisture (Crawford and Semlitsch 2008, Lowe and Bolger 2002). Therefore, timber stand age could be interpreted as a proxy for these more specific microhabitat features.

EVALUATING POTENTIAL SOLUTIONS

Timber harvests can potentially influence habitat characteristics in two important ways for stream salamanders: by increased sedimentation and filling of interstitial spaces in the substrate (Swank et al. 2001), and by degradation of the surrounding terrestrial habitat to the point that it interferes with the terrestrial component of the life cycles of many stream salamander species (Petranka et al. 1993). The preservation of forest buffer strips attempts to address both of these suspected proximate causes of stream salamander population declines.

Stream management zones are designed to help prevent or reduce soil erosion and associated sedimentation in two ways: by the restriction of unimproved stream crossings, roads, or landings within the SMZ, and by preservation of standing timber within buffer strips. Forest roads, including haul roads and skid trails, have been found to be the main source of sediment deposited in headwater streams during and immediately after harvest (Bowker 2013). Studying the same harvests at the same location as Maigret (2013), Witt et al. (2013) found that truck use of improved crossings over headwater streams reduced levels of total suspended solids by 86 percent. Furthermore, the preservation of a buffer strip, even if improved crossings are not mandated, has been shown to reduce sedimentation by physically obstructing sediment paths (Bowker 2013).

The preservation of some standing timber is likely the most effective, most practical, and perhaps the only solution for preserving the required quality and quantity of microhabitat factors necessary for salamanders and other stream-dwelling species to persist. For example, Peterman and Semlitsch (2009) found that a 30-m SMZ buffer and control did not differ in *E. wilderae* abundance. Likewise, Maigret (2013) found that retention of even the nearest streamside canopy tree was associated with increased *Desmognathus* spp. And although many have suggested a more liberal underharvested or unharvested buffer (Crawford and Semlitsch 2007), the amount of forest buffer required to maintain species viability on short and longer temporal scales is likely to be highly dependent on forest type, local hydrology, geology, and climate, as well as the salamander species and their prey base. Until we have a better understanding of the various spatial and compositional aspects of SMZs required to maintain stream salamander populations and other aquatic organisms, we argue that maintaining a precautionary approach to design and implementation of stream buffers would be wise. If a canopy retention requirement is included, some loggers may give the stream more buffer than the legal minimum to ensure not violating the requirement. This behavior was noted during the harvests associated with Maigret (2013), Bowker (2013), and Witt et al. (2013).²

To summarize these potential solutions, we propose the following list of minimum requirements for SMZ regulations in deciduous forests of the eastern United States. These should not be interpreted as the only regulations which may be of benefit, or as regulations in full detail, but rather as general ideas which would likely prove to benefit stream salamander populations during and after timber harvests.

- The SMZ regulations must apply to streams which flow perennially, intermittently, and ephemeral.
- The buffer widths mandated by SMZ regulations should increase with increasing bank slope.
- The buffer widths mandated by SMZ regulations should require at least some canopy retention.
- Within each buffer:
 - Use of heavy equipment should be prohibited.
 - Construction of roads, landings, or trails should be prohibited.
 - Use of improved crossings should be mandated.
- Dumping of materials into streams, whether perennial, intermittent, or ephemeral, should be prohibited.

² Stringer, J.W. 2013. Personal communication. Extension Professor, University of Kentucky, Department of Forestry, Lexington, KY 40546.

FUTURE RESEARCH

Specific and tactical studies instead of broad, homogenous ones may prove more informative to development of strategies for preserving stream salamander populations while maintaining or modifying contemporary timber harvest practices. For example, Lowe et al. (2004) showed what stage-specific differences can exist in *G. porphyriticus*. Few stage-specific studies have been conducted on other stream salamander species (but see Peterman and Semlitsch 2009), and therefore for many salamander species, we remain largely ignorant of the habitat requirements that may extend well beyond SMZ boundaries. Aside from life stages, vast differences in behavior, spatial ecology, and reproduction occur between species which are often grouped together as stream salamanders (Petranka 1998). For example, the finding by Ford et al. (2002) that basal area and abundances have a strong positive relationship for *D. ocoee* and *D. quadramaculatus* but a negative correlation for *E. bislineata* suggests that such differences likely make some species more susceptible than others to the effects of certain types of timber harvest.

Investigations of specific aspects of timber harvesting are few. Many details about tree selection and road construction remain untested, and may prove to be disproportionately informative to SMZ design and policy considering the small investments of money and time typically required to conduct these studies. Comparing, for example, different types of improved stream crossing techniques used by logging machinery may prove more worthwhile than studying improved crossings versus unimproved crossings in general.

A major concern for the forest industry is the cost of increased regulations to protect the biotic and abiotic resources that might be jeopardized by timber harvests. In many states, the application of BMPs is nonbinding and entirely at the discretion of loggers. Therefore, in-depth economic analyses of the costs of measures to ensure the protection of natural resources, including stream salamander populations, may be warranted to show how the adoption of such measures can be balanced with profitability. For example, Bowker (2013) videotaped the installation and removal of improved stream crossings and found that using a reusable skidder bridge or steel culvert to cross headwater streams was more time efficient (and therefore more cost efficient in labor and fuel) than using an unimproved ford. An analysis determining the value of timber left behind to satisfy SMZ canopy preservation guidelines may also be useful. Highlighting the potential to remove trees of high value and leave trees of low value while still retaining sufficient canopy cover, may help persuade logging companies to implement BMPs in states where they are not required. Although companies may be by definition profit driven, public image is important to many businesses in the forest industry, as shown by the popularity of Forest Stewardship Council and similar accreditations. In any event, providing evidence that complying with regulations designed to protect stream salamander species can be done without damaging profitability may be key to the implementation of regulations that help maintain the biological integrity of forest streams.

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FACTORS INFLUENCING AVIAN HABITAT SELECTION BETWEEN OAK-HICKORY AND MESIC FORESTS IN SOUTHERN ILLINOIS

Kevin P. Sierzega and Michael W. Eichholz¹

ABSTRACT

Introduction

Oak (*Quercus* spp.) regeneration has declined drastically over the past century in eastern deciduous forests predominantly because of decreased disturbance (i.e., fire). Many forests are undergoing mesophication, a positive feedback system that occurs within closed-canopy systems wherein shade-tolerant, late successional, mesic species such as maples (*Acer* spp.) and American beech (*Fagus grandifolia*) outcompete fire-adapted oak. The Central Hardwood region is predicted to be the first major system to convert to climax succession. Numerous neotropical migrant populations have declined in this region. Oaks are a keystone species and provide many resources for forest birds; the ability of mesic species to provide similar resources is untested. To determine whether patches of distinct oak forest and mesic forest provide similar resources, we examined two nonexclusive habitat-selection hypotheses about avian abundance and distribution: (1) habitat heterogeneity and (2) availability and distribution of food resources. We predicted oak-dominated patches would provide greater heterogeneity and more food resources than mesic patches.

Methods

We conducted spring-migration bird surveys April 15-May 15, 2013 and breeding bird surveys May 20-July 15, 2013 in distinct oak-dominated ($n=10$) and mesic-dominated ($n=9$) sites located in the Shawnee National Forest (SNF) in southern Illinois. Sites were ≥ 10 ha in area within upland deciduous habitat ranging in elevation from 130 to 230 m. Sites represented a gradient of relative percentages of oak and hickory (*Carya* spp.) basal area (BA) and mesic tree species BA. When applicable, a mesic site was paired with a proximate (i.e., <500 m away) oak site. We used distance sampling for both point-count transects ($n=3-6$ points/site) and line transects ($n=2$ transects/site). Points were located >250 m apart for independence and >75 m from site boundaries. Surveys were conducted one to two times during spring migration and four times throughout the breeding season. At each point, we collected microhabitat data (e.g., leaf litter depth, stem density, and vertical obstruction) and measured diameter at breast height of all tree species within a 10-factor prism sample to characterize tree composition.

We calculated avian community metrics between site types with the BiodiversityR package. Our measure of abundance was detection rate, which we calculated as the total number of each species detected at a site divided by the total survey time at the site. We included solely individuals detected <75 m from the observer and excluded edge species from our analysis. We examined relationships between individual species abundance rates and explanatory variables during the breeding season with the BiodiversityR package. We calculated density by using the “unmarked” R package and modeled species response with site covariates. We used repeated-measures analysis of variance in PROC.

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MIXED SAS 9.3 (SAS Institute Inc., Cary, NC) to examine differences between abundance and main effects of year and district, and year only, from a 9-year breeding bird survey data set in the SNF.

Results and Discussion

During migration, only species richness was higher in oak sites, whereas Shannon diversity and Jaccard's evenness were slightly higher in mesic sites. Richness, diversity, and evenness were slightly lower in oak than mesic sites during the breeding season. We identified higher mean abundance of all neotropical migrants in oak sites. Transient migrants (e.g., Nashville warbler; see table for scientific names) were present in greater numbers in oak sites. Only the Kentucky warbler displayed a positive relationship with relative percentage of mesic tree species BA, whereas the red-eyed vireo and worm-eating warbler exhibited negative relationships. The worm-eating warbler showed a significant positive relationship with relative percentage of oak-hickory tree species BA. The wood thrush displayed slight selection toward oak sites. We found higher densities (i.e., >20-percent difference) of the red-eyed vireo and Acadian flycatcher in oak sites and higher density of the tufted titmouse in mesic sites. We identified significant 9-year declines for five forest-breeding migrants in the SNF: hooded warbler, Kentucky warbler, Louisiana waterthrush, scarlet tanager, and worm-eating warbler. The cerulean warbler and many common species also declined.

Conclusions

Multiple migratory songbirds of the Central Hardwood region are experiencing rapid decline. Oak-hickory forests may offer additional resources for avifauna. The worm-eating warbler requires dense leaf litter for nest concealment, and leaf litter is deeper in oak stands than in maple stands. Furthermore, oak and hickory species yield high densities of Lepidoptera whereas maples and American beech contain low densities. Likewise, foraging efficiency of migrant birds is higher in oaks than in maples. Our preliminary results are consistent with these findings. We will sample for Lepidoptera in spring 2014 to examine our food availability hypothesis and will continue our surveys to increase statistical power. It is vital to understand factors influencing avian habitat selection between oak and mesic forests to guide management towards the conservation of many bird species.

Appendix A.—Scientific names of forest-breeding birds mentioned in text

Common name	Scientific name
Acadian flycatcher	<i>Empidonax vireescens</i>
Cerulean warbler	<i>Dendroica cerulea</i>
Hooded warbler	<i>Wilsonia citrina</i>
Kentucky warbler	<i>Oporornis formosus</i>
Louisiana waterthrush	<i>Parkesia motacilla</i>
Nashville warbler	<i>Vermivora ruficapilla</i>
Red-eyed vireo	<i>Vireo olivaceus</i>
Scarlet tanager	<i>Piranga olivacea</i>
Tufted titmouse	<i>Baeolophus bicolor</i>
Wood thrush	<i>Hylocichla mustelina</i>
Worm-eating warbler	<i>Helminthos vermivorus</i>

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ELK HABITAT SELECTION IN GREAT SMOKY MOUNTAINS NATIONAL PARK

Elizabeth Hillard and Laura E. DeWald¹

Abstract.—Great Smoky Mountains National Park (GSMNP) in North Carolina and Tennessee now has an established elk (*Cervus elaphus*) population 10 years after reintroduction. Although elk typically elect more open habitat, elk in GSMNP are showing they are capable of doing well in predominantly forested habitats. Evaluating how the established herd of elk is using forested areas in GSMNP is important for the health and management of the elk, and for the protection of the diverse flora within the park.

We assessed habitat selection of forest cover type, understory density class, disturbance use history, and distance to nonforested areas by using geographic information system (GIS) raster layers and fecal pellet counts. Elk trails were mapped and fecal pellet counts were used to index habitat selection. Plots were established to determine if there were relationships between elk selection and habitat components related to food and cover. In GSMNP elk selected successional and flood plain forest types, ericaceous understory classes of light to medium density, areas with concentrated settlement use history, and forests close to areas of open fields and recent human disturbance. The availability of species-specific woody browse was an important factor driving habitat selection. Successional forests contained the highest percentage of elk-preferred browse species.

Overall, elk in GSMNP are selecting forested areas that have more open canopies maintained by disturbances, and are selecting undisturbed continuous forests less because they do not contain preferred or abundant forage. If future monitoring detects pellets in more closed continuous forests, food sources in preferred younger forests may have become depleted, indicating that more intensive habitat management strategies should be considered. This understanding of resource selection by elk will be used to guide the management, monitoring, and future research of elk habitat management in GSMNP.

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

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FIRE

EFFECTS OF LONG-TERM PRESCRIBED BURNING ON STRUCTURE, COMPOSITION, AND TIMBER QUALITY OF OAK-HICKORY FORESTS IN THE MISSOURI OZARKS

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ABSTRACT

Introduction

Prescribed fire is commonly being used as a management tool for restoring or maintaining woodlands in the Central Hardwood Forest region. Woodlands are characterized as having canopies that are more open than those of forests, with lower abundance of woody stems in the midstory and understory layers, and a dense, diverse ground flora that is dominated by herbaceous species. Frequent fire may promote the structure and composition associated with woodlands by reducing the encroachment of woody stems in the subcanopy layers and encouraging the development of herbaceous vegetation (Hutchinson et al. 2012, Kinkead et al. 2013, Peterson and Reich 2001). However, many questions remain regarding the application of prescribed fire, especially over long timeframes. This paper reports on the effects of over 60 years of prescribed burning at regular intervals in oak-hickory forests of the Missouri Ozarks.

Methods

This study was established in 1949–1950 in University Forest Conservation Area (previously University State Forest) in Butler County, Missouri (36°55' N, 90°15' W). A randomized complete block design with two blocks that were located about 1.6 km apart was used. Within each block, six 40 m x 40 m plots were established, and two plots in each block were randomly assigned one of the three following treatments: Control (no prescribed fire); Annual (prescribed fire applied every year); and Periodic (prescribed fire applied every 4 years). Prescribed burns were conducted from March to May during scheduled burn years. Prior to the installation of this study, the stands consisted of all-aged oak-hickory forests that had not been burned for at least 20 years (Paulsell 1957).

In summer 2013, we recorded fire scar presence and diameter at breast height (d.b.h.) for all trees ≥ 10 cm d.b.h. In each plot we established five circular subplots (0.01 ha) and tallied all woody stems ≥ 1.5 m tall but < 10 cm d.b.h. by species. All woody stems < 1.5 m tall were tallied by species in four 1-m² quadrats in each subplot, and all standing vegetation < 1 m tall was clipped from four additional 1-m² quadrats in each plot to determine biomass. Analysis of variance was used to determine treatment effects on stand structure, composition, and the presence of fire scars.

Results

Total basal area of overstory trees ≥ 10 cm d.b.h. was significantly greater on the Control plots (23.2 m²/ha) than on the Periodic plots (17.0 m²/ha) in 2013. The basal area on Annual plots (19.3 m²/

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ha) did not differ from that on the other two treatments. There was no significant treatment effect on the number of overstory trees per hectare despite there being 335 trees on Control plots, 250 trees on Annual plots, and 186 trees on Periodic plots. Species in the white oak group contributed >70 percent of the basal area and trees per hectare on the Annual and Periodic plots but only around 50 percent on the Control plots. On the Control plots, hickories (*Carya* spp.) made up 10 percent of the basal area and 20 percent of the trees per hectare but represented <5 percent for either variable on Annual and Periodic plots. In total, 60 percent of the canopy trees on Periodic plots had at least one fire scar and 7 percent of trees on Annual plots were scarred.

Large midstory (3.00–9.99 cm d.b.h.) stems were the most common on Control plots (510 stems per hectare), with 30 stems per hectare on Periodic plots and zero stems per hectare on Annual plots. Small midstory (1.50–2.99 cm d.b.h.) stem density did not differ among treatments despite zero small midstory stems on Annual plots, 1,586 stems per hectare on Control plots, and 1,288 stems per hectare on Periodic plots. There was very little herbaceous vegetation biomass on Control plots (1.6 kg/ha), and the biomass of herbaceous vegetation did not differ between burned plots (Annual = 382 kg/ha; Periodic = 481 kg/ha).

Conclusions

Our results demonstrated that long-term prescribed burning at intervals of 4 years or less created the structure and composition associated with woodlands by reducing stand basal area and increasing production of herbaceous vegetation. Burning annually eliminated woody stems from the subcanopy layers, but periodic burning resulted in continual resprouting that retained small saplings in the midstory layer. However, there has been no apparent recruitment of new canopy trees in burned plots since this study was initiated in 1950. Given the prevalence of fire scars on the periodically burned plots, this fire regime is likely to reduce timber quality and may affect the longevity of canopy trees.

Acknowledgments

This research was partially funded by a Research Council grant sponsored by the Office of Research at the University of Missouri. The study plots are maintained and managed by the Missouri Department of Conservation with special thanks to Mark Pelton. We also thank Michael Hullinger and Dave Bourscheidt for their help with data collection and organization.

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PREScribed FIRE AND OAK SAPLING PHYSIOLOGY, DEMOGRAPHY, AND FOLIVORE DAMAGE IN AN OZARK WOODLAND

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Abstract.—Prescribed fire is a tool in wildlife management for restoring and maintaining midwestern oak woodlands. The success of some of the wildlife management objectives depends upon opening the canopy, new oak (*Quercus* spp.) saplings entering the canopy, and removal of cedar (*Juniperus virginiana* L.). We examined population characteristics of oak saplings based on changes in canopy light penetration, gas exchange, abiotic environment, and folivory. After 60 years of fire suppression, one portion of the study area was initially burned in 1980 (continuously burned), another portion was initially burned in 1999 (recently burned), and a third portion is still unburned (unburned). Fire opened the canopy from 6 percent in the unburned area to 8 percent and 40 percent in the recently burned and continuously burned areas. Saplings from the white oak group and red oak group responded to increased light availability with higher net photosynthetic rates. The resprouting ability of all oaks resulted in low mortality. Cedar mortality and recruitment were higher in burned than unburned areas. Sapling foliar area and total folivory was greater in burned than in unburned areas. Our data suggest reasons why, under the current biennial fire regime, potential oak canopy recruits will likely remain as large multitemmed sprouts and fail to enter the canopy.

INTRODUCTION

Historically, oak (*Quercus* sp.) woodlands were common throughout much of the midwestern United States (Nuzzo 1986). Oak woodland communities are characterized by widely spaced canopy trees and high light penetration to the ground layer (Anderson, 1998, Cottam 1949, Curtis 1959). Frequent fires historically played an integral role in the development and maintenance of midwestern oak woodland communities (Abrams 1992, 1996; Anderson and Fralish 1975; Gleason 1913). In the extended absence of fire, oak woodlands have become closed canopy woodlands (Anderson and Brown 1986, Cole and Taylor 1995, Nuzzo 1986). Oak sapling growth may then be limited by low light conditions caused by a closed canopy (Burns and Honkala 1990, Crow 1988, Lorimer 1994).

Leaf nitrogen (N) concentrations, plant water stress levels, and light are probably three of the most important variables affecting leaf quantity, quality, and feeding by folivores (Baraza et al. 2004, De Bruyn et al. 2002, Warring and Cobb 1992). Prescribed burning has been hypothesized to increase quantity and quality of plant foliage (Adams and Rieske 2003), and therefore, insect folivory (Rieske 2002). However, there have been very few studies of oak sapling leaf quantity, quality, and folivory as a function of prescribed fire.

In this paper we examine the efficacy of prescribed fire for increasing canopy light penetration, the physiological and demographic response of oak saplings to canopy light penetration, and insect folivory on oak saplings in a woodland area in the Ozark Mountains region of southwestern Missouri

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managed for deer and turkey with varied prescribed fire regimes. Physiological responses (e.g., gas exchange) of saplings were measured to better understand mechanisms influencing population-level responses to management efforts, while folivory was measured to assess potential impacts of leaf area loss on plant responses to light. We tested the following predictions: (1) prescribed burning increases canopy light penetration; (2) oak saplings are light limited in unburned woodlands and will respond positively to increased light with increased net photosynthetic rates; (3) total sapling density and dominance will increase in response to increased light availability; and (4) foliar loss to insect herbivores will be higher in burned woodlands and will be associated with leaf quality and abiotic conditions that result from increasing canopy light penetration.

METHODS

Study Site

This study was conducted at Bull Shoals Field Station (BSFS) within the 809.4 ha (2000 acre) Drury Conservation Area (DCA) in Taney County, Missouri (36° N latitude, 93° W longitude). The climate at DCA is continental. Growing season (April through September) mean daily maximum temperature was 28 °C, and mean annual precipitation was 1097 mm during this study. DCA has karst topography with elevations ranging from 180 to 340 m (600 to 1100 feet). Soils at DCA are predominantly Gasconade-Opequon-Clarksville association (Nigh and Schroeder 2002).

At DCA, two primary woodland habitat types existed in 1999. The habitat types included closed canopy oak woodland (60 percent of area with 80-95 percent overstory cover) and open oak woodland (35 percent of area with 40-60 percent overstory cover), with 5 percent of the habitat considered glade (<30 percent overstory cover). The oak woodland with high overstory cover is thought to be the result of 50 years of fire suppression. The current open woodland habitat at DCA is thought to be a result of the reintroduction of fire to the area in 1980. Periodic (generally every 2 years) prescribed burning since 1980 by the Missouri Department of Conservation (MDC) over 35 percent of the area was used to manage for deer and turkey and increase understory species diversity, resulting in open woodland habitat. In the spring of 1999, the MDC implemented prescribed burns over an additional 50 percent of the woodlands with biennial frequency to begin managing for wildlife. Ten percent of the area has not burned in over 60 years.

We established three experimental habitat areas (100 m x 100 m) representing the following:

- Unburned (n = 2): last known fire in 1950.
- Recently burned (n = 2): fire resumed in 1999 and repeated every 2 years.
- Continuously burned (n = 2): fire resumed in 1980 and repeated every 2 years.

An overview of understory species richness, midstory structure, and overstory basal area in the three habitat areas in 2008 is provided in Tables 1, 2, and 3. The data in this paper were collected between 2001 and 2003.

Table 1.—Descriptive statistics for understory species richness in twelve 1-m² plots in three habitat areas with different burn histories located in southwest Missouri

		Habitat area		
		Unburned woodland	Recently burned woodland	Continuously burned woodland
Species richness	Mean	21.67	31.80	60.38
	Standard deviation	6.22	4.22	13.87
	Minimum	16	27	30
	Maximum	30	39	76

Table 2.—Descriptive statistics for midstory tree structure in six 0.01-ha plots in three habitat areas with different burn histories located in southwest Missouri

		Habitat area		
		Unburned woodland	Recently burned woodland	Continuously burned woodland
Density (stems/0.01ha)	Mean	13.33	2.67	0.75
	Standard deviation	9.99	1.51	0.89
	Minimum	29	4	2
	Maximum	3	0	0
Height (m)	Mean	3.48	5.87	3.42
	Standard deviation	1.38	3.27	4.25
	Minimum	11.20	1.09	9.50
	Maximum	1.46	0	0
Dominant species		<i>Rhamnus caroliniana</i>	<i>Cornus florida</i>	<i>Quercus alba</i>

Canopy Light Penetration

We used leaf area index (LAI) as a measure of canopy light penetration (CLP) to determine if prescribed burns resulted in a more open canopy. Based on the complementary relationship of LAI and CLP, we defined CLP as:

$$CLP = 12 - LAI$$

where 12 is the theoretical maximum value for LAI (Boyles and Aubrey 2006). Values were converted in this way for a more intuitive illustration of canopy light penetration as opposed to interception. We obtained LAI indirectly using an AccuPAR PAR-80 light interception device (Decagon Devices, Inc., Pullman, WA). Measurements were collected 1.2 m above ground level at five randomly spaced points along a 100 m transect within each habitat area. These five measurements were averaged to calculate one LAI value, and therefore, one CLP value for each transect per sample period. Measurements were collected throughout two growing seasons (2002 and 2003).

Table 3.—Descriptive statistics for overstory basal area (m²/ha) in six 0.1-ha plots in three habitat areas with different burn histories located in southwest Missouri

Species		Habitat area		
		Unburned woodland	Recently burned woodland	Continuously burned woodland
Total	Mean	23.75	26.04	17.27
	Standard deviation	7.69	6.86	8.79
	Minimum	15.16	18.71	4.87
	Maximum	36.41	37.32	29.78
Red oak species	Mean	12.58	13.69	5.95
	Standard deviation	7.95	7.46	6.93
	Minimum	3.00	4.25	0.00
	Maximum	23.67	26.29	19.00
White oak species	Mean	5.57	7.53	9.61
	Standard deviation	3.83	4.18	7.74
	Minimum	0.76	4.52	1.45
	Maximum	9.43	14.61	23.20
Hickory species	Mean	3.38	2.10	1.22
	Standard deviation	1.78	1.39	0.88
	Minimum	1.58	0.26	0.00
	Maximum	6.49	4.11	2.30
Cedar	Mean	1.43	1.62	0.15
	Standard deviation	2.10	1.66	0.30
	Minimum	0.00	0.00	0.00
	Maximum	4.96	4.49	0.81
Other species	Mean	0.60	1.08	0.29
	Standard deviation	0.77	0.95	0.39
	Minimum	0.00	0.00	0.00
	Maximum	2.06	2.46	1.12

Abiotic Environment

Air temperature, relative humidity, soil temperature, soil moisture, solar radiation, and wind speed were monitored using two portable weather stations (Campbell Scientific, Logan, UT). These abiotic parameters were logged every 30 seconds with averages calculated every 30 minutes. The weather stations were rotated through six combinations of the three woodland habitat areas during the summer of 2003. During any one of the six rotations, data were collected for a minimum of 10 days at a time. Data were collected from May 19 (Julian Day 139) to August 25 (Julian Day 237). Raw data were revised to remove the first and last day of each 10-day period. In addition, at least one day was randomly selected to be removed to equal 7 days of data per rotation. For each variable, the mean for the 7 days in each habitat area was calculated. The means of these data sets were then compared to a reference mean obtained over the same time periods from a permanently placed weather station (Campbell Scientific, Logan, UT). The permanent weather station was selected as a reference because it is located in a large open prairie-like field at BSFS within 1 km of each woodland area.

Physiological Characteristics

We measured leaf level net photosynthesis of oak saplings (stems < 1.0 m) at ambient (A_{AMB}) and saturating (A_{MAX}) light levels using a portable gas exchange system (LI-6400, Li-Cor, Lincoln, NE). A single leaf of five randomly selected saplings located along a transect within each habitat area was measured. Ambient light levels were measured at each sample leaf using the quantum sensor on the gas exchange system, and light levels were held constant in the measurement chamber to obtain A_{AMB} . A_{MAX} was then obtained at saturating light levels (1500 $\mu\text{mol}/\text{m}^2/\text{s}$). Temperature (25 °C), CO_2 concentration (360 ppm), and flow rate (500 $\mu\text{mol}/\text{s}$) were also maintained at constant levels within the measurement chamber. The same permanently tagged individuals were measured once per month throughout one growing season.

Population Characteristics

Three permanent variable area belt transects (Dobrowski and Murphy 2006), hereafter referred to as transects, with an average area of 102 m^2 (4 m wide with various lengths) were established within each habitat area. During plot establishment we identified, tagged, and mapped all oak (*Quercus* spp.) and cedar (*Juniperus virginiana* L.) individuals less than 2.0 m in height. We categorized oaks into two groups due to small sample sizes of individual species. Hereafter, all members of the red oak group (*Quercus falcata* Michx., *Q. marilandica* Muenchh., *Q. rubra* L., *Q. shumardii* Buckl., and *Q. velutina* Lam.) are collectively referred to as red oak, and all members of the white oak group (*Quercus alba* L., *Q. macrocarpa* Michx., *Q. muhlenbergii* Engelm., and *Q. stellata* Wangenh.) are collectively referred to as white oak.

We assessed oak and cedar sapling characteristics by measuring stem number of all individuals within each transect during June or July of 2001, 2002, and 2003. We determined stem density, mortality (absence of an individual previously present), and recruitment (introduction of a new individual) in 2001-2002 and 2002-2003 at each plot.

Leaf Quantity, Quality, and Folivory

Total sapling leaf area was determined across the three habitat areas throughout the growing season. Within each habitat area, three 50 m by 5 m (250 m^2) belt transects were established. Within each transect, 15 red oak and 15 white oak saplings less than 2 m in height were randomly chosen and tagged for a total of 45 oaks per habitat type. The area of each leaf was determined using a CI-420 leaf area meter (CID Bio-Science, Inc., Camas, WA).

To assess leaf quality, leaf tissue nitrogen (N) was determined using micro-Kjeldahl analysis following the methods of Anderson and Polis (1999). Due to the destructive nature of leaf N analysis (i.e., removing an entire leaf at the petiole), five oak plants were randomly selected adjacent to each tagged plant, and 15 oak plants were used per habitat area each month. Leaves were dried at 60 °C and ground, and then approximately 0.1 g samples were weighed with 0.0001 precision. Following digestion of leaf samples and the N concentration assay, samples were analyzed using a Shimadzu UV 1601 spectrophotometer (Colombia, MD). Predawn and midday water potential measurements were obtained on the same plants using a pressure chamber (PMS Instrument Co., Corvallis OR) to assess water stress. Predawn measurements indicate levels of soil water availability and midday measurements, along with photosynthetic rates, indicate limitations to carbon gain by stomatal

Table 4.—Mean ($\pm$ standard deviation) monthly canopy light penetration (CLP = 12 – Leaf Area Index) in three habitat areas with different burn histories located in southwest Missouri

Month	Habitat area		
	Unburned woodland	Recently burned woodland	Continuously burned woodland
April	10.9 $\pm$ 0.1	11.2 $\pm$ 0.1	12.0 $\pm$ 0.1
May	9.1 $\pm$ 1.0	10.1 $\pm$ 0.5	11.5 $\pm$ 0.3
June	7.8 $\pm$ 2.3	10.1 $\pm$ 1.1	11.6 $\pm$ 0.3
July	7.6 $\pm$ 1.9	10.2 $\pm$ 0.4	11.5 $\pm$ 1.2
August	7.9 $\pm$ 0.5	10.3 $\pm$ 0.3	11.6 $\pm$ 0.9
September	7.8 $\pm$ 0.4	9.9 $\pm$ 0.2	11.7 $\pm$ 1.1

regulation. Stems less than 3.0 mm in diameter were cut using a clean razor blade. The pressure applied to the leaf blade (in megapascals) to force water out was equivalent to the amount of soil moisture (predawn) and water stress (midday) that an oak was experiencing.

To estimate folivore damage, each leaf on each tagged oak plant was visually scored using a ranking system based on the methods of Futuyma and Wasserman (1980) to estimate the area removed due to insect herbivores. Individual leaf scores were then averaged into one overall score for each oak plant. Visual scoring took place in mid June, July, August, and September. The visual scores for the month of August were applied to the total available leaf area to estimate the amount of leaf area that was removed by herbivores from each habitat area. Leaf area loss due to herbivory was obtained by calculating the mean of the range of percent leaf area removed that corresponded with each rank. The mean percent of damage per plant (obtained from the visual score data) was then multiplied by the total leaf area of each plant in each habitat area to estimate the leaf area removed. The leaf area removed per habitat area was estimated by summing the leaf area removed from each plant.

RESULTS AND DISCUSSION

Canopy Light Penetration

The reintroduction of fire positively influenced CLP (Table 4). Canopy light penetration was higher in continuously and recently burned than unburned habitat areas, and CLP declined through June in unburned areas but remained constant in burned areas.

The higher CLP in habitat areas treated with prescribed fire is an important result as the long-term goal of burning is to open the canopy and restore woodland communities back to their presettlement characteristics, of which CLP is a key driving force and a defining characteristic (Curtis 1959, Leach and Ross 1995, Nuzzo 1986, Taft 1997). Other studies have also demonstrated that prescribed fire is an effective tool in opening closed woodland canopies in the Midwest (Anderson and Brown 1986, McCarty 1998). If canopy opening results in photosynthetically active radiation being greater than saturation levels for oaks, then oak growth should not be light limited.

Earlier maximum leaf expansion in habitat areas treated with prescribed fire suggested burning may have altered leaf phenology. This potential effect of fire on overstory leaf phenology has not been studied, but it is possible that altered phenology is an important component of understanding fire effects on sapling physiology and growth. For example, fire may alter leaf phenology by temporarily

increasing plant-water status (Borchert 1994, Eamus 1999) which may result in earlier bud break and leaf expansion (Saha 2001). However, it could be that seasonal patterns are an artifact of canopy and subcanopy tree composition.

Abiotic Environment

Forest microclimate below the canopy is expected to vary with canopy light penetration and influence growth and survival of tree seedlings and saplings (von Arx et al. 2013). There may be threshold canopy densities that support the regeneration of tree saplings through both light penetration and associated microclimate. We measured air temperature, relative humidity, soil temperature, soil moisture, solar radiation, and wind speed at each woodland habitat type to provide a baseline for interpreting how canopy light penetration might affect the microclimate below the canopy (Table 5). The continuously burned areas generally had higher air and soil temperatures and solar radiation but had lower relative humidity and soil moisture than the more closed, recently burned or unburned habitat areas. In the recently burned habitat areas there were higher air and soil temperatures, higher soil moisture, and greater solar radiation than in unburned habitat areas. These data suggested that in the recently burned habitat areas, microclimate was still more similar to unburned habitat areas than continuously burned habitat areas. Therefore, the amount of water stress to oaks is expected to be the greatest in continuously burned habitat areas, and oak sapling gas exchange is potentially going to be limited by water stress in open canopies (e.g., closing stomata) even though light is not limiting. Soil moisture may also explain extended overstory leaf expansion.

Physiological Characteristics

There was a positive relationship between CLP and A_{AMB} suggesting that red and, especially, white oak saplings are capable of responding to increased light availability (Table 6). Higher light saturated photosynthetic rates (A_{MAX}) among saplings in burned habitat areas indicate that oak saplings are light limited in closed canopy woodlands. Elevated A_{AMB} in burned habitat areas suggest a positive response to increased light availability, especially in white oaks. These physiological patterns may be due to both increased light and nitrogen availability, as both light and nitrogen are common limiting resources for plant photosynthesis and growth (Fahey et al. 1998, Reich et al. 1997, Sipe and Bazzaz 1995, Walters and Reich 1997). Burning has been shown to increase nitrogen availability in forests (Boerner 1988, Raison 1979, Reich et al. 1990), but our estimates of foliar nitrogen concentrations did not support this, nor did water stress limitations as measured by predawn and midday water potential (Table 7). Therefore, light availability was likely the major limiting resource within closed woodland habitat areas in our study, a result found in forests in the northeastern United States (Finzi and Canham 2000).

Population Characteristics

Increased CLP (Table 4) and subsequent enhanced photosynthetic capacity (Table 6) did apparently lead to increased dominance for red and white oak in recently burned woodland areas (Table 8). Red oak rootstock density was greater than white oak in all but the continuously burned woodland areas (Table 8). White oak had higher basal area in recently burned woodland areas than would be expected based on its proportional contribution to density. White oak density was three times higher in recently burned than unburned woodland areas, but dominance was 21 times greater. This suggests

Table 5.—Differences between mean values of environmental conditions in three habitat areas with different burn histories located in southwest Missouri relative to an open prairie field

Environmental parameter	Habitat area		
	Unburned woodland	Recently burned woodland	Continuously burned woodland
Air temperature (°C)	-1.66	-0.75	+1.79
Relative humidity (%)	+4.16	+1.77	-4.14
Soil temperature (°C)	-4.73	-2.47	+3.21
Soil moisture (%)	+3.76	+3.92	-8.10
Solar radiation (W/m ²)	-375.82	-359.50	-170.16
Wind speed (m/s)	-0.59	-0.39	-0.34

Note: Comparisons between habitat areas and open field are based on 7 days of comparisons collected from May 19 (Julian Day 139) to August 25 (Julian Day 237).

Table 6.—Mean ($\pm$ standard deviation) maximum (A_{MAX}) and ambient (A_{AMB}) photosynthetic rates of red and white oaks in three habitat areas with different burn histories located in southwest Missouri

	Habitat area		
	Unburned woodland	Recently burned woodland	Continuously burned woodland
Red oak A_{MAX} ($\mu\text{mol CO}_2/\text{m}^2/\text{s}$)	5.05 $\pm$ 0.21a [†]	5.40 $\pm$ 0.22a	8.54 $\pm$ 0.63b
Red oak A_{AMB} ($\mu\text{mol CO}_2/\text{m}^2/\text{s}$)	1.48 $\pm$ 0.21a	3.82 $\pm$ 0.42b	5.80 $\pm$ 0.74c
White oak A_{MAX} ($\mu\text{mol CO}_2/\text{m}^2/\text{s}$)	5.65 $\pm$ 0.19a	8.14 $\pm$ 0.28b	9.47 $\pm$ 0.37b
White oak A_{AMB} ($\mu\text{mol CO}_2/\text{m}^2/\text{s}$)	1.58 $\pm$ 0.21a	4.44 $\pm$ 0.43b	6.77 $\pm$ 0.69b

[†]Significant differences (Tukeys; $P = 0.05$) between habitat areas within a row are indicated with different letters.

Table 7.—Mean ($\pm$ standard deviation) values for leaf parameters of oak plants (< 2.0 m height) as a function of habitat areas with different burn histories located in southwest Missouri

	Habitat area		
	Unburned woodland	Recently burned woodland	Continuously burned woodland
Total leaf area (cm ²)	23,780 $\pm$ 507a [†]	35,314 $\pm$ 714a	90,705 $\pm$ 1,4700b
Leaf area removed (cm ²)	3,110 $\pm$ 79a	6,947 $\pm$ 159c	4,472 $\pm$ 105b
Foliar nitrogen content (%)	1.88 $\pm$ 0.31b	1.80 $\pm$ 0.53b	1.53 $\pm$ 0.27a
Pre-dawn water potential (MPa)	-0.57 $\pm$ 0.49a	-0.55 $\pm$ 0.44a	-1.42 $\pm$ 1.27b
Mid-day water potential (MPa)	-1.51 $\pm$ 0.82a	-1.42 $\pm$ 0.76a	-2.21 $\pm$ 1.18b

[†]Significant differences (Tukey's; $P = 0.05$) between habitat areas are indicated with different letters.

Table 8.—Mean ($\pm$ standard deviation) population characteristics of cedar, red oak, and white oak saplings (< 2.0 m in height) in three habitat areas with different burn histories located in southwest Missouri

Population characteristic [†]	Unburned woodland			Recently burned woodland			Continuously burned woodland		
	Cedar	Red oak	White oak	Cedar	Red oak	White oak	Cedar	Red oak	White oak
Density (#/m ²)	0.018 $\pm$ 0.007	0.200 $\pm$ 0.038	0.022 $\pm$ 0.006	0.009 $\pm$ 0.004	0.270 $\pm$ 0.088	0.070 $\pm$ 0.008	0.15 $\pm$ 0.041	0.09 $\pm$ 0.010	0.21 $\pm$ 0.033
Dominance (cm ² /m)	0.067 $\pm$ 0.056	0.811 $\pm$ 0.413	0.030 $\pm$ 0.009	0.0007 $\pm$ 0.000	1.180 $\pm$ 0.110	0.461 $\pm$ 0.250	0.004 $\pm$ 0.002	0.210 $\pm$ 0.046	1.650 $\pm$ 0.440
Mortality (%)	10.00 $\pm$ 10.00	7.10 $\pm$ 4.00	8.35 $\pm$ 8.35	88.10 $\pm$ 1.00	8.90 $\pm$ 3.20	7.65 $\pm$ 2.85	53.40 $\pm$ 21.90	2.68 $\pm$ 1.96	2.90 $\pm$ 1.96
Recruitment (#/m ²)	0.023 $\pm$ 0.010	0.051 $\pm$ 0.029	0.013 $\pm$ 0.013	0.033 $\pm$ 0.033	0.034 $\pm$ 0.015	0.050 $\pm$ 0.009	0.500 $\pm$ 0.200	0.073 $\pm$ 0.030	0.160 $\pm$ 0.083

[†]Density = number of individuals per meter squared. Dominance = sum of basal areas/cm² for each species per area sampled. Mortality = absence of an individual that was previously present. Recruitment = presence of a new individual. Values are means across 3 years (or 2 years for mortality and recruitment).

that the increase in CLP in recently burned areas has already resulted in increased carbon gain for white oak, but not red oak.

Increased recruitment in white oaks in recently burned woodlands (Table 8) may be attributed to increased stump sprouting, seed production, increased germination rates, or a combination of these factors. The similarity in mortality of red and white oaks between woodland types (Table 8) suggests tolerance to fire. Red and white oak experienced topkilling by fire but resprouted immediately, generally with more stems of smaller diameter (data not shown). The resprouting capabilities of oak in response to fire have been well documented (Abrams 1992, Stearns 1991) and are likely the reason mortality did not differ between woodland areas (Reich et al. 1990). Under continued prescription of the biennial fire regime, we suggest that red and white oak will persist, but only as multistemmed shrubs unlikely to be recruited into the canopy. We have no evidence of canopy recruitment (data not shown).

The maintenance of oak woodlands depends upon the right frequency of new individuals entering the canopy (Ladd 1991, Leach and Ross 1995, McCarty 1998). Studies have shown oak resprouts develop rapidly once burning is withheld (Anderson 1998, Bowles and McBride 1998, Crow 1988) and may grow up to 6 m in less than a decade (Cain 1995). Therefore, if burning were excluded from this area for a decade, sprouts should grow through the susceptible stage of topkilling and likely enter the canopy. After a cohort group reaches this height, a biennial burn cycle should help keep competition in check. However, in order to determine an ideal burn frequency, it is imperative to understand how long it will take for red cedar saplings to reach fire-tolerant size. We found that the current biennial fire regime appears to be effective at removing (high mortality), although not eradicating (high recruitment), red cedar saplings (Table 8).

Leaf Quantity, Quality, and Folivory

Increased light along with increased photosynthetic rates probably played a role in the difference in foliar area between the habitat areas (Table 7). Our results were similar to those reported by Dudt and Shure (1994) and Baraza et al. (2004) that indicate that plants grown in a high light environment have greater leaf area than shade grown plants. We found that oak saplings in continuously burned habitat areas lost 5 percent of their total leaf area, while in the unburned and recently burned habitat areas oak saplings lost 13 percent and 20 percent of their total leaf area to folivory, respectively (Table 7). The percentage of leaf area removed in the more closed areas was within the range (10 to 20 percent) found by Hochwender et al. (2003). However, the damage amounts in the continuously burned habitat areas were much lower than the 18 ± 1 percent found by Hochwender et al. (2003) and Marquis and Forkner (2004). Oak saplings in the closed canopy habitat, independent of recent burning, had 13 percent more foliar N and were 30 percent less water stressed than oak saplings in the continuously burned habitat areas. We found that less water stressed saplings had more foliar damage, which supports the notion that insects do not always have a positive response to water stressed foliage (Warring and Price 1990). Ultimately, we found that insect folivores had the greatest potential impact in recently burned habitat areas.

CONCLUSIONS

We found that prescribed burning increased canopy light penetration. Oak saplings were light limited in unburned woodlands and responded to increased light with increased net photosynthetic rates and leaf area even though abiotic conditions were more stressful (higher temperature and lower soil moisture). Total sapling density and dominance increased in response to increased light availability only after continuous burning for 20 years, yet there was no recruitment into the overstory. Foliar loss to insect herbivores was greatest in recently burned habitats, but clear associations with leaf quality and abiotic conditions that result from increasing canopy light penetration were not found. We suggest that prescribed fires should occur at longer intervals along with physical gap formation to promote oak regeneration into the canopy, and that changes in folivory or abiotic conditions associated with an open canopy are not currently limiting oak canopy recruitment.

ACKNOWLEDGMENTS

Funding for this research was provided by a grant from the Missouri Department of Conservation to D. Alexander Wait and funding from Missouri State University to the authors. We thank the Bull Shoals Field Station for use of facilities, and Pam Brown and Becky Gehringer for help in the field.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

KANAWHA STATE FOREST, WV: WILDLAND FIRE FUEL LOAD ASSESSMENT AND GEOSPATIAL DISTRIBUTION

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Abstract.—Fire has become a major concern along the wildland-urban interface (WUI), and there is increased awareness that fires could result in serious damage to people and property in residential areas occurring in forested landscapes. Part of the long-term strategy the West Virginia Division of Forestry outlined in the 2010 West Virginia Statewide Forest Resource Assessment was to conduct a comprehensive monitoring program of wildland fire fuels. In an effort to predict and prevent catastrophic fire activity, identifying areas of the WUI that are at risk for wildfire is one of the initial steps in developing mitigation plans and priority levels. In early 2013, an extensive fuel sampling inventory was performed to quantify the fuel loading in the 9,388 acre Kanawha State Forest located 7 miles south of the state capitol, Charleston, WV. Results included fuel load volumes, a spatially distributed fuel load assessment model, and effects of site and vegetation characteristics. Data from the fuel sampling will be used in forthcoming evaluations that incorporate additional elements of fire hazard into a comprehensive fire risk assessment.

INTRODUCTION

In the last 150 years, fire in the area that is now Kanawha State Forest was predominately caused by industrial activities involving coal mining, logging, and the supporting railroad transportation networks. In 1908, over 1.7 million acres burned in West Virginia causing the state to pass the West Virginia Reform Law in 1909 which established the Division of Forestry and defined its focus as wildfire protection. In 1938, West Virginia purchased the first 6,705 acres of what is now the 9,388 acre Kanawha State Forest to serve as a recreation area to be developed by Civilian Conservation Corps and to provide forest fire protection for the area (West Virginia Division of Forestry 2009).

There are still widespread occurrences of uncontrolled wildfire in West Virginia, and the fire program mission of prevention, preparedness, and suppression remains the objective of the Division of Forestry. The large majority of wildfires are caused by debris burning and incendiary actions, and in the past 11 years there have been over 10,000 fires burning over 227,000 acres statewide (West Virginia Division of Forestry 2013a).

As part of the 2008 Farm bill mandate, West Virginia conducted the 2010 West Virginia Statewide Forest Resource Assessment, which included approximately 12 million forested acres statewide (West Virginia Division of Forestry 2010). One of the long-term strategies identified by the West Virginia Division of Forestry (WVDOF) was to locate and map all high risk fire areas in the state by 2015, providing information to make the necessary actions to successfully maintain the fire program mission.

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Wildland fire risk incorporates site-specific characteristics that both prevent and promote fire across a landscape, climatic conditions, as well as ignition variables. Kanawha County is historically among the counties with both the highest number of fires and the highest number of acres burned, and the vast majority of both are attributable to incendiary actions and other anthropogenic sources. As such, in the 2010 forest assessment, Kanawha County was identified as one of the 14 “hot” counties and a high priority area of wildfire concern based on a composite of fire occurrence, topography, and wildland-urban interface concerns (West Virginia Division of Forestry 2010).

Formal wildland-urban interface (WUI) definitions specify minimum structural density and vegetation levels, however the WUI is generally recognized as the area where housing and dense vegetation intermingle (Haight et al. 2004). Smoke levels are also a concern in the WUI from visibility reduction, public safety, and the health effects of breathing fine particulate matter. Among the other values at risk (VAR) in, or in close proximity to, Kanawha State Forest are heritage sites of historical importance, recreation resources, natural resource extraction infrastructure, and valuable timberland.

The major forest cover type in Kanawha State Forest is mixed mesophytic hardwoods, and eastern fuel types have not been as extensively modeled as western types. Several studies identify the existing fuel type shortcomings, especially in areas with ericaceous shrubs and timber litter. Suggested alternatives or customization as well as pictorial depictions (Brose 2009, Brose et al. 2009) are available to assist with determining appropriate local fuel types. Anderson’s original 13 fuel types (Anderson 1982) have been improved upon by the 40 fuel type models identified by Scott and Burgan (2005), including an increase in the number of fuel types for high-humidity areas and types for forest litter and shrub understory. When using national tools such as the National Fire Danger Rating System and Wildland Fire Assessment System (West Virginia Division of Forestry 2013a, 2013b), certain fuel types used in the models may not accurately represent the specific local vegetation profile. Despite the increasing eastern fuel type resources and improvements, potentially inaccurate fuel type models used in some national tools can result in ambiguous or incorrect fire risk predictions unless careful interpretation is made.

In the absence of a large scale disturbance, fuel loading by seasonal pattern is a relatively static variable (Cooke et al. 2007). However, after two large disturbances in 2012, all previous fuel loading data were irrelevant. A high incidence of incendiary action and escaped debris fires, an undetermined volume of additional fuel loading from recent storms, potential local fuel type model differences, and proximity to the WUI necessitated fine-scale field data to be collected at Kanawha State Forest. Knowing current fuel loads and existing vegetation characteristics would enable WVDOF to make a more precise fire risk assessment than that from a national level analysis.

The objective of this project was to gather data to address the need to establish a general geospatial and quantitative assessment of potential wildfire fuel hazard areas in proximity to the WUI and VAR. Ignition causes and locations obtained from the WVDOF were mapped as part of the 2010 forest assessment. Together, fire hazards and ignition patterns can be used to determine fire risk, and mitigation and resource allocation can be concentrated in areas of greatest concern.

The urgency in determining fuel loading precluded conducting a full scale fire risk assessment that included climatic and topographic variables. Fuel load volume was the only component quantified by the data collected, with the intention that the baseline established by this study can provide more accurate data for future analyses and other management uses.

METHODS

Sampling

A sampling strategy was developed by the West Virginia University Appalachian Hardwood Center according to FIREMON protocol (Lutes et al. 2006). Data collection was done in January 2013 by the West Virginia Division of Forestry. Landform types were divided into three strata and grouped by ridgetops (RT), north, northeast, and east aspects (N, NE, E), and south, southwest, and west aspects (S, SW, W), in which 30 sample plots each were randomly distributed, totaling 90 plots. Northwest and southeast aspects were not selected for sample plot locations in this initial assessment because the fire hazard influence from fuel loading on these neutral topographic positions was expected to fall between high and low values of the other strata. The FIREMON fuel load (FL) method specifies sample measurements to be taken using the planar intercept technique. Measurements for 16 metrics were recorded, including 1-hour fuels (0- to 0.25-inch diameter), 10-hour fuels (>0.25- to 1-inch diameter), 100-hour fuels (>1- to 3-inch diameter), and 1000-hour fuels (>3-inch diameter), which were further separated into decay classes 1 (completely sound), 2 (moderate decay), and 3 (extensive decay) and weighed in tons per acre. Duff/litter profile depth and the litter component of total profile depth were measured. Litter defined the loose layer of mostly twigs, grasses, leaves and needles, and other debris that had not begun to decompose. Duff defined the organic layer below litter and above mineral soil in an advanced state of decomposition, inclusive of any root systems. Additionally, percentages of live and dead tree cover vertically projected to the ground, percentages of live and dead herbaceous cover vertically projected to the ground, and average tree and herbaceous heights were measured.

Analysis

Data Analysis

Plot data on fuel tonnage was calculated using equations described by Van Wagner (1968). Averages and totals were calculated on a per acre basis and identified by aspect, and a weighted averaging model was developed using eight fuel loading variables. Properties of each fuel type sample category were subjectively considered by their predicted effect on fire behavior (Scott and Burgan 2005) and were assigned weighted averages accordingly. Both 1-hour and 10-hour fuel readily reach ignition temperature and strongly determine fire behavior and spread (Anderson 1985) and were weighted 30 percent and 20 percent, respectively. Percentages of dead herbaceous cover and litter depth were weighted 7.5 percent and 5 percent, respectively, accounting for their ability to sustain fire combustion and spread by providing fuel bed continuity. Duff/litter depth, 100-hour, and 1000-hour fuels are attributed to extended flaming front and smoldering burn time and can strongly influence fire severity and effects (Lutes et al. 2006) and were assigned 20 percent, 10 percent, and 5 percent weights, respectively. Most 1000-hour fuel was blowdown from recent wind events and was still in decay class 1 and 2. Increased fire effects reflect the combustibility of sound or moderately decayed fuel compared to that of extensively decayed fuel. Dead tree cover was assigned 2.5 percent for its potential addition to fine fuel volumes and ladder fuels for an unlikely crown fire occurrence. SAS (SAS Institute Inc., Cary, NC) was used to perform statistical analyses used to develop a natural breaks algorithm to categorically quantify fuel load variables as a means to illustrate the distribution of fuels most influential on fire behavior and effects.

Spatial Analysis

Geospatial analysis was done by the WVDOF using ArcGIS 10.0 (Esri, Redlands, CA). All fuel load raster data were reclassified into high, medium, and low classes defined by natural breaks and were assigned values of 3, 2, and 1, respectively. Using the inverse distance weighting method, fuel load raster layers were derived using point sample data in Geostatistical Analyst, and values were interpolated within Kanawha State Forest boundaries. Raster analysis consisted of 9.84-square-foot grids, and using the Spatial Analyst Weighted Overlay tool, reclassified layers were assigned the corresponding weighted average influence.

Distribution of reclassified fuel load layers was overlaid with corresponding aspect location. Using spatial analyst tools, ridgetops were delineated with a 50 foot vertical buffer. The algorithm using weighted averages only incorporated fuel load volumes, not the effect aspect has on fuel load moisture levels and resulting fire behavior. However, for illustrative purposes, spatial distributions of the amalgamated fuel loading variables generated by the embedded weighted average algorithm were differentiated by aspect (Fig. 1).

Proximity of WUI areas to distribution of fuel loads in Kanawha State Forest was defined by overlay of raster data identifying structure locations and weighted model fuel load classes (Fig. 2).

RESULTS

Several results were influenced by the cyclical nature and disturbance-based conditions of the time when sampling was done in January 2013. Phenological stage influences litter volumes as well as herbaceous cover, and high levels of 1000-hour fuels were due to the significant volume of down woody debris from two large storm events in 2012, a derecho in June and Hurricane Sandy in October.

Total 1000-hour fuels across all sample strata averaged 20.1 tons per acre, with the N, NE, and E strata grouping having the highest average of 25.5 tons per acre. Across all strata, 100-hour fuels averaged 0.40 tons per acre and were about the same on the N, NE, and E grouping and RT strata, which averaged 0.46 and 0.45 tons per acre, respectively. Across all strata, 10-hour fuels averaged 0.96 tons per acre, averaging the highest with 1.1 tons per acre on N, NE, and E strata. The 1-hour fuels averaged 0.63 tons per acre across all strata with the highest averages of 0.7 on the RT strata. Duff/litter profile depth across all strata averaged 4.5 inches and was the highest with 4.7 inches on S, SW, and W strata. Litter depth component of total duff/litter profile depth across all strata averaged 2.8 inches, and averaged the highest on N, NE, and E strata with 3 inches. Percent dead tree cover across all strata measured 0.66 percent, averaging highest at 0.79 percent in the N, NE, and E strata. Percent dead herbaceous cover across all strata averaged 1.0 percent and averaged the highest at 1.3 percent on the RT strata (Table 1).

Within each stratum, the number of acres was further separated into low, medium, and high classes, and the percentage of acres that had fuel loading levels within each class is shown in Table 2. Fuel loading assessment across the Kanawha State Forest on all sampling strata included 3,214 acres categorized as low, 5,436 acres categorized as medium, and 738 acres categorized as high.

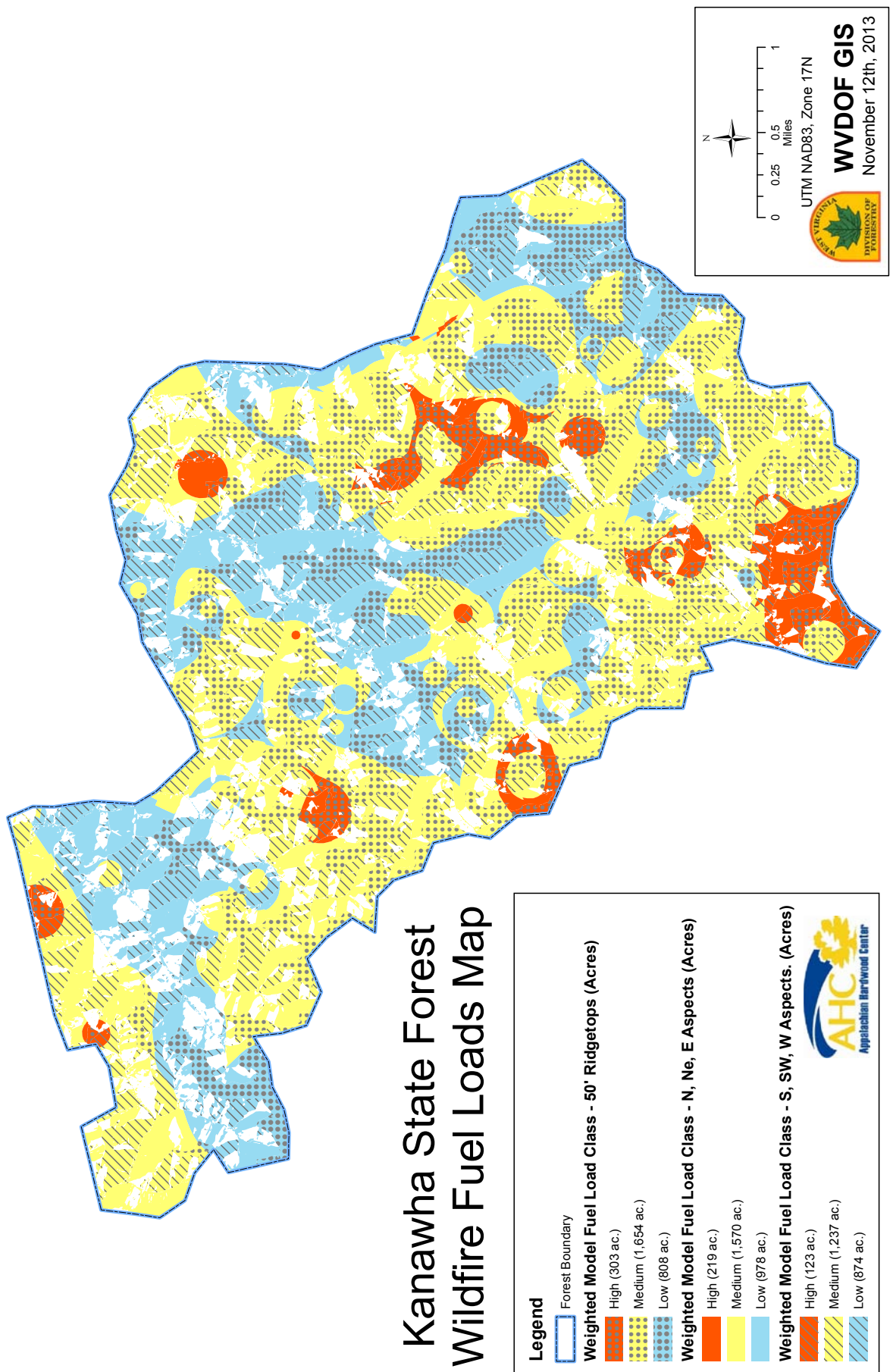


Figure 1.—Overlay of fuel load assessment map and sampling strata map.

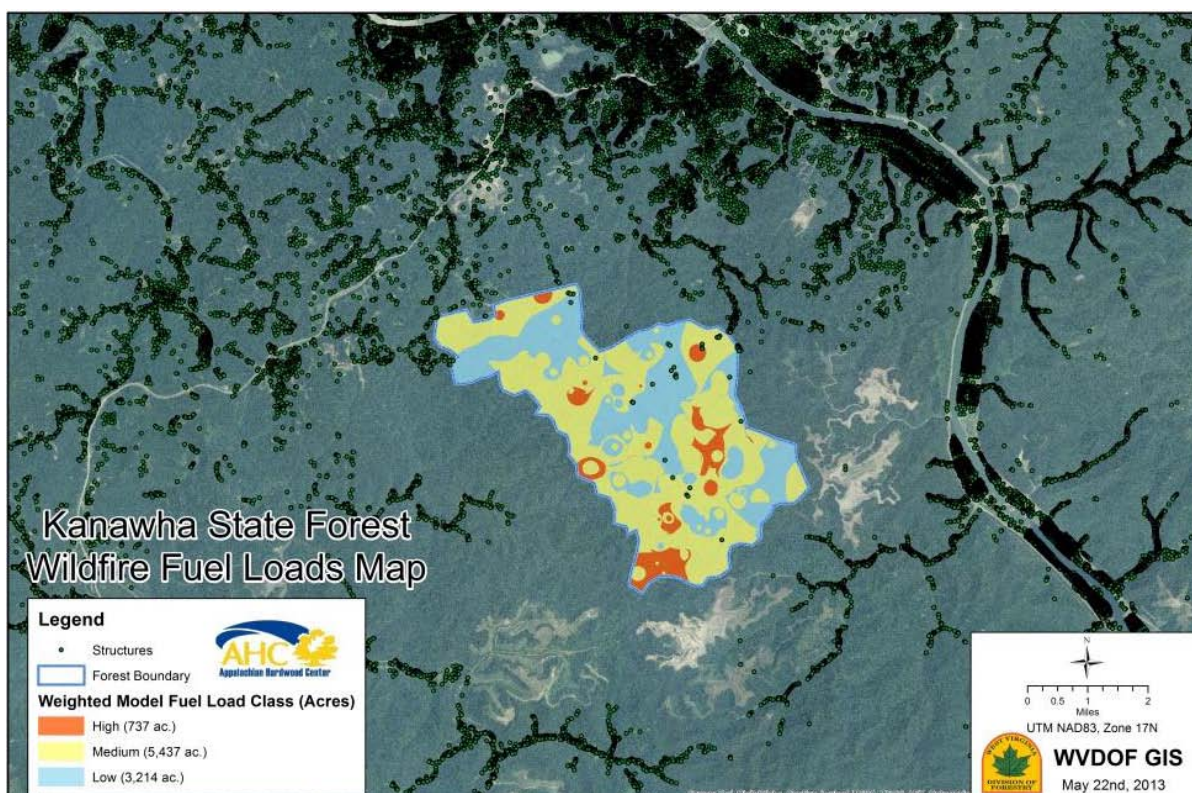


Figure 2.—Fuel load assessment in the wildland-urban interface context.

Table 1.—Fuel load sample averages by sampling strata, category and weighted model averaging for deriving fuel load assessment classes.

Fuel load category	Weighted average	Sample strata			
		All strata	N, NE, E	S, SW, W	Ridgetop
	%	<i>Average per acre</i>			
1000-hour (tons)	5	20.08	25.54	12.42	22.30
100-hour (tons)	10	0.40	0.46	0.29	0.45
10-hour (tons)	20	0.96	1.06	0.81	1.00
1-hour (tons)	30	0.63	0.63	0.56	0.70
Duff/litter profile (inches)	20	4.45	4.20	4.74	4.42
Litter depth (inches)	5	2.77	3.04	2.79	2.48
Dead tree cover (%)	2.5	0.66	0.79	0.58	0.62
Dead herbaceous cover (%)	7.5	1.00	1.24	0.42	1.33

Table 2.—Total acreage and percentage of acreage by fuel load assessment class and sampling strata

Sampling strata	Total acres	Load assessment class		
		Low	Medium	High
N, NE, E	2767	35%	57%	8%
S, SW, W	2234	39%	55%	6%
Ridgetop	2765	29%	60%	11%
NW, SE	1622	34%	60%	6%
All strata	9388	34%	58%	8%

The fuel loading averages were highest for five of the eight fuel load categories on N, NE, and E aspect strata grouping, perhaps attributable to higher site productivity and corresponding increased biomass on those topographic positions. Future sampling may indicate if faster decomposition rates on a more productive site equalizes the averages across aspects (Waldrop et al. 2013) and the rate at which that happens by fuel type category.

In areas with sloping terrain, fuel moisture content varied by aspect. South, west, and southwest aspects are exposed to increased solar radiation, resulting in decreased fuel moisture levels compared to other aspects with comparable canopy shading of fuels. Moisture levels are also influenced by topographic position with typically dry predispositions, such as ridgetops, within a limited elevation range (Lafon and Grissino-Mayer 2007). Because fuel moisture was not measured, a conservative fuel loading hazard rating would incorporate a fuel moisture condition significantly less than the specified moisture of extinction, defined as the dead fuel moisture at which a fire will not spread (Rothermel 1972).

DEVELOPMENT AND APPLICATION

The primary objective of this project was to establish a fuel load assessment from the sample dataset to augment current systems of fire risk analysis while maintaining enough flexibility for potential use in other capacities. In addition to modeling fire behavior and fire effects, this dataset can be used by management to develop fuel treatment prescriptions, estimate carbon storage, and assess wildlife habitat.

The data collected will be analyzed using FlamMap, a fire behavior mapping and analysis program that computes potential fire behavior characteristics over a landscape for constant weather and fuel moisture conditions. However, a sense of urgency to assess fire hazard before the spring 2013 window of West Virginia's bimodal fire season necessitated a preliminary model incorporating only fuel loading statistics to be reviewed in the interim. Subsequent developments to enhance this model will include additional inputs, facilitating comparisons with the results of the more robust FlamMap model.

Fuel moisture and climatic conditions including temperature, wind, precipitation, and humidity levels are integral in determining probability of ignition and fire behavior. However, because of their dynamic nature and temporal variability, they were not incorporated in this preliminary stage. Moisture levels available from weather data are a surrogate for the energy release component, which is an index of the potential heat release and fire intensity dependent on fuel moisture levels (Bradshaw 1984). West Virginia has 13 regional area weather stations (RAWS)² that can supplement fuel loading field data with climatic data to predict fire potential (Burgan 1998).

Two of the assessment tools the West Virginia Division of Forestry uses to identify wildfire danger levels statewide are the National Fire Danger Rating System and the Wildland Fire Assessment System (West Virginia Division of Forestry 2013a, 2013b). These types of products support analysis at a national and regional scale, but variability in fuels, weather, topography, and ignition sources

² Webster, B. 2010. Our role in the state. 16th Annual LEPC/SERC Conference; September 13-15; Pipestem State Park, Pipestem, WV.

contribute to the difficulty in applying the assessments at a local scale. Also, nationally mapped data layers used in these products often do not include local level resources or infrastructure, presenting management and planning challenges. Hindered by differing data standards, collection methods, and format, aggregation of local and national data is difficult (Calkin 2010).

Among the data gaps that the WVDOP identified in the 2010 forest assessment is the need for current and accurate fuel model maps. These have historically been produced by LANDFIRE, a nationwide interagency vegetation, fire, and fuel characteristics mapping program. In addition to LANDFIRE an increasing number of programs are available that model fire behavior, effects, fuels, smoke management, 75 of which are currently identified by the Missoula Fire Science Laboratory (U.S. Forest Service 2013). These models are invaluable to fire managers, however, most are complex and require familiarization with the functionality specific to each program and can require extensive training.

Complicating the use of the models is the lack of adequate fuel type descriptions of some eastern forests, particularly in the Appalachian region. Variable biophysical and topographical inputs dependent on specific conditions at the time and place of the risk assessment are another aspect that makes a general or national assessment difficult or even irrelevant. This is less of a concern in landscapes of relative homogeneity; however, unique characteristics of already inadequately described local fuel types necessitate a tailored model that can accommodate the site variability. Among unpredictable factors of fire intensity and severity in these variable conditions are residency time, spread rate and patterns, flame length, and their effects on fuel consumption and vegetation mortality (Rothermel 1972).

Several areas of future research and potential application include comparison between field gathered data and different national level datasets, and comparison of different national models using field collected data. Because field data is expensive and time consuming to collect, these comparisons can help to determine if there are significant differences between datasets when making risk assessments and can help to determine the level of accuracy that is lost by not collecting field data. Part of both prevention and preparedness on public lands is identifying areas of high wildfire risk that could benefit from mechanical fuels reduction and potentially the use of prescribed controlled burning in carefully considered areas. With field collected data from landscapes underrepresented by current fuel type models, the potential exists for the development of fuel models based on fire behavior in controlled experiments involving prescribed burns and documented wildland fire behavior. Collaboration opportunities have been identified with other state and federal agencies and nongovernment organizations that currently incorporate prescribed burns as part of their management plans. This includes participation in the Fire Learning Network which promotes the scientific basis of prescribed fire to restore and maintain fire-adapted ecosystems.

In addition to state and federal land, West Virginia has many individual and corporate forest landowners and interested parties that could benefit from fire risk assessments. Efforts to fulfill the preparedness aspect of WVDOP's fire plan also include outreach to private land and homeowners in the wildland-urban interface. Firewise communities and community wildfire protection plans are in place and are expanding to engage and provide guidance to homeowners on preparing and protecting their property from wildfire and minimizing potential damages. However, because of

the combination of factors making fire risk assessment difficult, many proactive prevention and protection measures are not being identified or taken.

This project demonstrated the potential utility of geographic information system (GIS) raster modeling using field-collected fuel loading data, local knowledge of ignition distribution and historical fire occurrence data, and climatic data from local RAWS to develop a framework for empirical modeling and a composite index of local scale fire risk.

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EFFECTS OF SIMULATED PRESCRIBED FIRE ON AMERICAN CHESTNUT AND NORTHERN RED OAK REGENERATION

Ethan P. Belair, Mike R. Saunders, and Stacy L. Clark¹

ABSTRACT

Introduction

American chestnut (*Castanea dentata* [Marsh.] Borkh.) was a dominant species in the forests of eastern North America prior to the importation of chestnut blight (*Cryphonectria parasitica* [Murr.] Barr) in the early 1900s and ink disease (*Phytophthora cinnamomi* Rands) in the 1800s (Anagnostakis 2012). Historical accounts and phylogeny (Manos et al. 2001) suggest chestnut may have certain disturbance adaptations similar to oak (*Quercus* spp.), including the ability to prolifically sprout following topkill (Paillet 2002). However, the response of chestnut to fire has only been observed in mature individuals, not in seedlings which would be used during restoration attempts. The objective of this study was to compare American chestnut's response to prescribed fire to that of the closely related red oak (*Q. rubra* L.). We hypothesized that available light and initial stem diameter would positively affect the vigor of sprouts produced (Dey 1991, Johnson et al. 2002).

Methods

We simulated the morphological effects of surface fire and associated topkill on 143 red oak and 94 chestnut seedlings using a burn chamber. Seedlings underplanted in April 2007 and May 2009 at three sites in north central Indiana were selected to provide a wide range (5.2–49.0 mm) of ground line diameters (GLD). Canopy photos were taken adjacent to each seedling to estimate canopy openness. Treatments were designed to simulate surface fire conditions optimal for releasing oak advanced regeneration (Brose and Van Lear 1998) while ensuring topkill of all seedlings (Peter et al. 2009) and occurred just prior to leaf out. Before ignition, we cleared the vegetation from the area around each seedling and placed a Kaowool™ (Thermal Ceramics, Inc., Augusta, GA) combustion chamber filled with dry leaf litter around the base of the stem. The litter was ignited and stems were heated using a plumber's torch, subjecting stems to temperatures of 200–300 °C for 60 seconds from each of two diametrically opposite directions. Individual sprout height (Ht), ground line diameter, and first order lateral branch (FOLB) count were monitored throughout the 2013 growing season. Ground line diameter was recorded 5 cm above ground line to allow for taper and curvature at the point of attachment, and only on sprouts ≥7.5 cm total height.

Results

Both species sprouted consistently in response to topkill with no difference in number of sprouts per stump ($p = 0.66$). Total sprout productivity (TSP) was calculated for both species as $TSP = BA \times Ht + (2 \times FOLB)$ where BA = basal area. Red oak responded with a greater TSP compared to American chestnut ($58.0 \pm 11.4 \text{ cm}^3$ compared to $117.6 \pm 17.7 \text{ cm}^3$; $p = 0.006$), though there was little difference between species in the final size of the tallest, most dominant sprouts.

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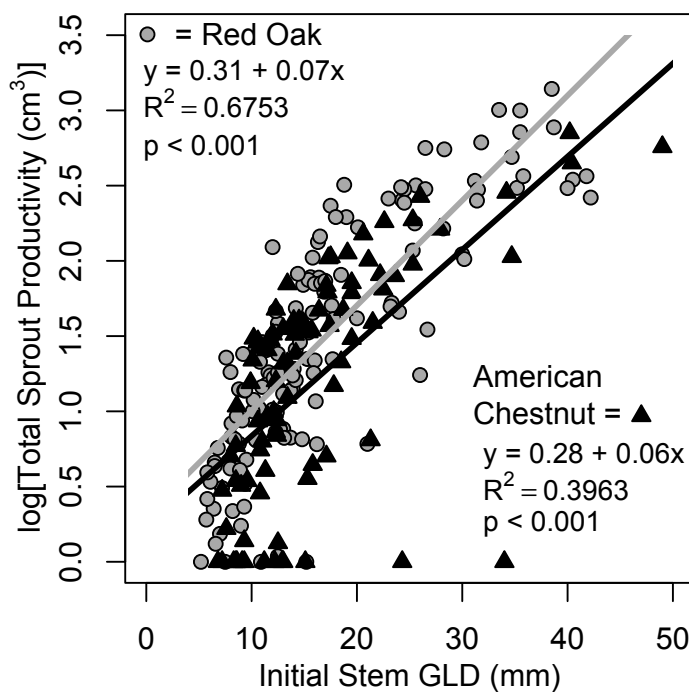


Figure 1.—Initial stem ground line diameter (GLD) regressed with log transformed total sprout productivity of American chestnut and northern red oak seedlings following heat induced topkill.

Initial stem GLD was positively related to the variation in TSP (Fig. 1) and both sprout height and sprout GLD after topkill for both species (Fig. 2). Initial GLD was also positively correlated with number of sprouts produced per stump ($r = 0.54$, $p < 0.001$). Correlations between canopy openness and both TSP and dominant sprout size for both species (all $r > 0.55$, $p < 0.001$) indicated that light availability positively influenced an individual's ability to recover following topkill. However, original stem size may have been confounded with light environment, as seedlings in high light conditions were larger after a given interval than those in low light environments. Additionally, our sample may be biased because the most open site contained only red oak but was included to provide seedlings of comparable size to the largest American chestnuts. When we restricted analysis to only the trees in comparable light environments, species differences in average TSP reversed, presumably due to red oak's considerably smaller average diameter at those sites.

Discussion

The tendency of larger seedlings to produce more and larger sprouts after topkill by fire is probably related to their more extensive root systems providing greater access to nutrient and water resources. The results suggested that both species had highly productive sprout responses following topkill by fire, producing sprouts up to 200 cm tall after one growing season. Both species also responded favorably to increased light availability and may, therefore, benefit from fire, harvesting, natural canopy mortality, or other disturbances that increase light availability at the forest floor (McCament and McCarthy 2004, Paillet 2002). Red oak had greater TSP, height, and GLD following topkill across initial GLD sizes compared to American chestnut, suggesting it might be slightly more productive than chestnut following topkill by fire. Our results implied that larger oak and chestnut seedlings can produce sprouts above browse line and can be considered advance regeneration in the growing season following a single prescribed fire. However, it is still unclear whether chestnut seedlings can tolerate multiple burns that are sometimes used for understory management in oak dominated forests (Dey and Hartman 2005).

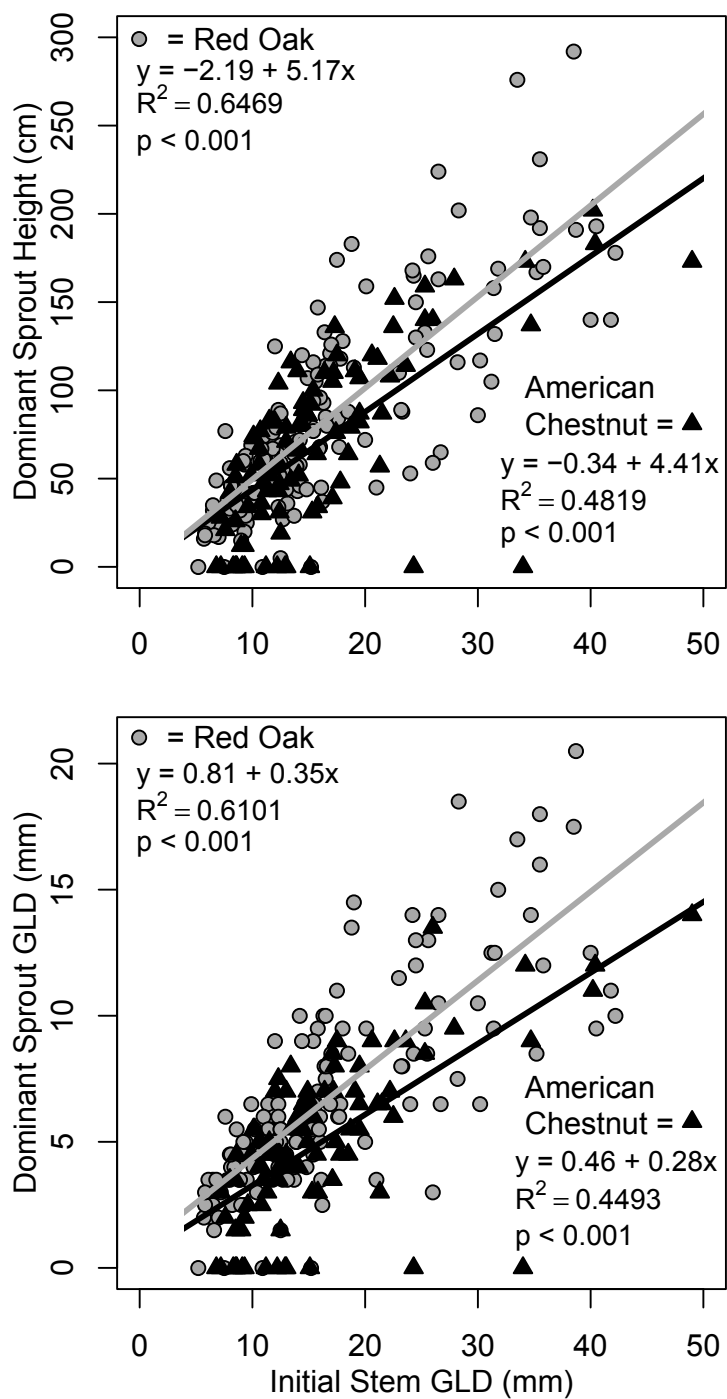


Figure 2.—Initial stem ground line diameter (GLD) regressed with height and GLD of dominant sprouts of American chestnut and northern red oak seedlings following heat induced topkill.

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THINNING AND PRESCRIBED FIRE ALTERS HARDWOOD SEEDLING SPROUTING IN THE WILLIAM B. BANKHEAD NATIONAL FOREST, ALABAMA

Callie Jo Schweitzer, Daniel C. Dey, and Yong Wang¹

ABSTRACT

Introduction

The William B. Bankhead National Forest is using active management to shift mixed *Quercus-Pinus* forests towards forests more dominated by upland hardwoods. We studied the use of three levels of thinning (none, light thin, and heavy thin) and three levels of prescribed fire (none, one burn, and two burns) and all combinations in a factorial experimental design to assess the level of juvenile sprouting in the hardwood reproduction cohort. Stands were either unthinned, thinned to 75 square feet of residual basal area (light thin), or thinned to 50 square feet residual basal area (heavy thin). The burn treatments included one burn, which was done on all burn treatments within 3 to 5 months post-thin, or two burns, where the second burn was conducted 3 years after the initial burn. All burns were conducted during the dormant season. In all 36 treatment stands, we surveyed five permanent vegetation plots before treatments were initiated, in the first growing season following treatment (thin and initial burn), following the third growing season, and in the fourth growing season (after the second burn). The density of reproduction with multiple sprouts (clumps), the number of sprouts per clump, and the density of all the sprouts were analyzed using all species, all *Quercus* species combined (seven different species, with the majority being *Q. alba* L., *Q. prinus* L., and *Q. coccinea* Muench.) and for *Acer rubrum* L., the primary competitor with oaks.

Results and Discussion

The density of clumps increased in all treatments over time. Pretreatment stands had approximately 1,400 clumps per acre and did not differ among stands. Four growing seasons following treatment, stands averaged 4,500 clumps per acre, with significant treatment differences. All stands that received two burns had more clumps per acre, with heavy thinned stands > light thinned > no thin (Table 1). Most *Quercus* clumps occurred in the two burn treatments. The light thin and two burn treatments had the highest density of clumps and were significantly greater than the no thin + no burn, no thin + one burn, heavy thin + no burn, heavy thin + one burn, and light thin + one burn treatments. *Acer rubrum* clumps were most dense on the thin (both light and heavy) and two burn treatments compared to all other treatments. The density of sprouts also changed with treatment and time, with higher densities on the two burn treatments, which increased to approximately 30,000 sprouts per acre. *Quercus* sprouts increased the most in the stands where two burns were done under light thinning (9,200 sprouts per acre) and heavy thinning (6,600 sprouts per acre). *Acer rubrum* also increased in both of these treatments by 11,000 sprouts per acre.

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Table 1.—Change in the number of clumps and sprouts per acre by treatment after four growing seasons

Treatment	All species		All <i>Quercus</i> spp.		<i>Acer rubrum</i>	
	Clumps	Sprouts	Clumps	Sprouts	Clumps	Sprouts
<i>Change in the number per acre</i>						
No thin + no burn [†]	720 c [‡]	2040 c	400 b	1300 c	160 b	280 b
No thin + 1 burn	1180 c	5520 bc	160 b	620 c	860 ab	3920 ab
No thin + 2 burns	3260 abc	13600 bc	900 ab	2980 bc	1420 ab	7020 ab
Light thin + no burn	2680 c	11640 bc	840 ab	2960 bc	1000 ab	5360 ab
Light thin + 1 burn	3020 bc	16080 b	320 b	1140 c	2040 a	12280 a
Light thin + 2 burns	5980 ab	29320 a	2360 a	9280 a	1520 ab	11420 a
Heavy thin + no burn	1800 c	7920 bc	480 b	1240 c	800 ab	4180 ab
Heavy thin + 1 burn	2500 c	11900 bc	280 b	700 c	1360 ab	7120 ab
Heavy thin + 2 burns	6320 a	30460 a	1340 ab	6600 ab	1860 ab	11940 a

[†]Thinning treatments: no thin; light thin = 75 square feet per acre residual basal area; heavy thin = 50 square feet per acre residual basal area. Prescribed burn frequencies: no burn = no prescribed fire; 1 burn = burned within 3 to 5 months after thinning; 2 burns = burned 3 to 5 months after thinning and again 3 years after the initial burn.

[‡]Within a column, values followed by the same letter do not differ at $\alpha \leq 0.001$.

Changes in the densities of the number of clumps and sprouts over time were also considered and somewhat mirrored the total densities (Table 1). For all species, the change in clumps after four growing seasons was greatest in those treatments receiving two burns and did not differ among the thinning treatments that received two burns. Changes in *Quercus* clumps among all treatments did not show any differences following the thin and first burn. Following the second burn, *Quercus* clumps were greatest for the two burn treatments, but those did not differ from the change incurred in the light thin + no burn after four growing seasons. *Acer rubrum* clumps had the greatest change in the light thin + one burn treatment. The greatest change in the densities of sprouts for all species (*Quercus* and *Acer rubrum*) was following the two burns under light and heavy thinning.

It is well established that a majority of oaks in new stands grow from seedling sprouts that have accumulated over time before regeneration harvesting. Both time and disturbance frequency influenced the regeneration cohort in this study. The density of clumps and sprouts increased following thinning and after each subsequent burn, with the greatest densities in the heavy thinned + two burn treatment. For *Quercus*, the density of clumps and sprouts was highest following thin and two burns; while thin and one burn treatments had lower densities than thin with no burn treatments. *Acer rubrum* clump and sprout densities appeared to be favored by thinning and both one burn and two burns.

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IS THERE EVIDENCE OF MESOPHICATION OF OAK FORESTS IN THE MISSOURI OZARKS?

Matthew G. Olson, Aaron P. Stevenson, Benjamin O. Knapp, John M. Kabrick,
and Randy G. Jensen¹

Abstract.—Many studies on oak-dominated forests of the Central Hardwood region (CHR) have reported increasing abundance of fire-sensitive species and poor recruitment of oak (*Quercus* spp.) in the absence of frequent fire. However, most of these studies were conducted in the eastern and central CHR, and the assumption that similar dynamics occur in the western CHR has not been fully substantiated. We investigated forest dynamics in relatively undisturbed, mature oak-hickory forests of the Missouri Ozarks during a 15-year period (1995-2010). Data for this study were from untreated sites (controls) of the Missouri Ozark Forest Ecosystem Project (MOFEP) that have not experienced wildfire or harvesting for over a half century. In order to evaluate the influence of site quality on compositional dynamics, we selected a subset of permanent plots found on four ecological land types (ELTs) spanning a range of site qualities. The density of maple (*Acer* spp.) seedlings (<1.5-inch diameter at breast height [d.b.h.]) increased on nearly all ELTs over the 15 year period. As of 2010, maples were the most abundant species in the seedling layer on higher quality ELTs, while oaks and hickories (*Carya* spp.) were a major component of the seedling layer of lower quality ELTs. However, oaks and hickories were a major component of the sapling (1.5- to 4.5-inch d.b.h.) layer of all ELTs, while maple was a minor component. In contrast to the understory dynamics, the oak-dominated overstories (≥ 4.5 -inch d.b.h.) of all ELTs remained largely unchanged from 1995 to 2010. These findings supported four working hypotheses: (1) upland forests of the Missouri Ozarks are in early stages of mesophication where fire has been excluded for at least 50 years; (2) mesophication in the western CHR is occurring at a slower rate than in eastern portions of the CHR; (3) mesophication is slowest on xeric, south-facing slopes; and (4) the predominance of low quality soils and frequent drought in the Ozarks will limit these forests from reaching late stages of mesophication, particularly on xeric sites.

INTRODUCTION

There is mounting evidence that oak dominance in the Central Hardwood region (CHR) is unsustainable under prevailing disturbance regimes. Research on oak-dominated forests since the mid-20th century has indicated a potential shift in composition to more shade-tolerant tree species (Abrams et al. 1997, DeSantis et al. 2010, Christensen 1977, Glitzenstein et al. 1990, Lorimer 1984, Monk 1961, Nowacki et al. 1990, Richards et al. 1995). In particular, many studies have noted increasing abundance of fire-sensitive species coincident with diminishing or no recruitment of oak. These findings along with historical records, evidence from paleoecological and dendrochronological investigations, and knowledge of oak ecology have led to the widely held position that loss of oak dominance is linked to fire suppression (Abrams 1992).

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The loss of oak dominance has implications for other ecosystem components. Aside from direct and indirect effects on wildlife species through trophic interactions, successional replacement of oak species will likely also cause changes in understory microenvironment (light, temperature, humidity, soil moisture). In turn, alteration in the understory microenvironment may affect future forest vegetation. The mesophication hypothesis states that long-term fire suppression in communities of fire-tolerant species leads not only to successional replacement by fire-sensitive species, but that the microenvironment continually changes to favor fire-sensitive species and deteriorates for fire-tolerant species during the replacement process (Nowacki and Abrams 2008).

There is growing consensus that widespread mesophication is occurring throughout the CHR (Arthur et al. 2012, Nowacki and Abrams 2008). However, most of these studies have come from the eastern and central portions of the CHR, and the assumption that similar dynamics occur in the western CHR has not been fully substantiated. Recent research indicates that oak-dominated ecosystems at the extreme western edge of CHR are transitioning to opportunistic species in the absence of fire (Burton et al. 2010, DeSantis et al. 2010, Thomas and Hoagland 2011). Past research on forest succession in Missouri has revealed inconsistencies when generalizing trends at the state level. Pallardy et al. (1988) observed reductions in oak sapling densities accompanied by increases in sugar maple (*Acer saccharum*) saplings over 14 years. In another study, sugar maple regeneration was found to be abundant wherever a seed source was present, while oak regeneration was largely relegated to xeric sites despite a ubiquitous seed source (Nigh et al. 1985). However, these studies were focused on succession in the mesic Missouri River Hills where maple is prolific (Nigh et al. 1985, Pallardy et al. 1988) or in stands intentionally selected based on the presence of sugar maple (Nigh et al. 1985). There is evidence that mesophication may not proceed as quickly in the xeric Ozarks Highlands of Missouri as in other portions of the state or the CHR, a pattern consistent with how mesophication is predicted to proceed on xeric sites (Nowacki and Abrams 2008). Shifley et al. (1995) observed low abundance of maple in xeric forests of the Missouri Ozarks, which they contrasted with findings of increasing maple further east. However, a more recent study of long-term, large-scale compositional changes in the Missouri Ozarks detected both an increase in density and expansion of fire-sensitive species onto xeric, fire-prone sites currently dominated by oak species (Hanberry et al. 2012, Hanberry et al. 2014).

The objective of this study was to determine if oak-dominated upland forests of the Missouri Ozark Highlands are shifting to fire-sensitive species where fire has been excluded. Although the process of mesophication involves changes in multiple, inter-related factors that lead to reduced flammability of historically fire-maintained plant communities, compositional shifts to fire-sensitive species provide an early indication of mesophication. In the Missouri Ozarks, the major fire-sensitive species capable of replacing oak as a canopy component are red maple (*Acer rubrum*) and sugar maple. Therefore, we were mainly interested in assessing whether the abundance of maple was increasing relative to oak across a site quality gradient. Since the probability of oak self-replacement is inversely proportional to site quality and maples are typically associated with mesic sites (Johnson et al. 2009), we also postulated that the understory abundance of maple would be greater and that of oaks would be lower as site quality increased.

METHODS

Study Sites

This study used data collected as part of the Missouri Ozark Forest Ecosystem Project (MOFEP). MOFEP was initiated in 1989 by the Missouri Department of Conservation (MDC) as a long-term, large-scale experiment investigating the ecological impacts of even-aged, uneven-aged, and no-harvest management on Missouri Ozark forests. MOFEP sites are located on MDC land and occur mostly in the Current River Oak Forest Breaks and Current River Oak-Pine Woodland Hills land type associations (Kabrick et al. 2000). MOFEP's nine study sites are operational compartments that range in size from 776 to 1,275 acres and are representative of the scale of MDC forest management on state land in the Ozarks. At the start of the experiment, these sites were dominated by mature, relatively undisturbed forest and were largely free of manipulation for at least 40 years. See Brookshire and Shifley (1997) for more details on the MOFEP experiment.

Data for this study came from three untreated control sites of the MOFEP experiment. These sites have not experienced wildfire or harvesting for over a half century and, therefore, are well suited for assessing the consequences of fire exclusion on succession in oak-dominated forests of the Missouri Ozarks.

Analytical Approach

Data from permanent sample plots were used to assess changes in tree species abundance over the 15-year period from 1995 to 2010. Woody vegetation on MOFEP was monitored using nested fixed-area plots: trees ≥ 4.5 -inch d.b.h. in 0.5-acre plots (henceforth referred to as overstory); stems 1.5- to 4.4-inch d.b.h. in four 0.05-acre plots (saplings); stems 3.3 feet tall and up to 1.4-inch d.b.h. in four 0.01-acre plots (large seedling); and stems < 3.3 feet tall in sixteen 0.00025-acre plots (small seedling). Stems counts per plot of overstory, sapling, large seedling, and small seedling size classes were converted to trees per acre (TPA) prior to the analysis. Basal area (BA) measured in square feet per acre was included for assessing overstory change

The following nine taxonomic groups were included in this study: (1) red oak (*Quercus coccinea*, *Q. marilandica*, *Q. rubra*, *Q. shumardii*, and *Q. velutina*); (2) white oak (*Quercus alba*, *Q. muehlenbergii*, and *Q. stellata*); (3) hickory (*Carya glabra*, *C. texana*, and *C. tomentosa*); (4) shortleaf pine (*Pinus echinata*); (5) blackgum (*Nyssa sylvatica*); (6) sassafras (*Sassafras albidum*); (7) flowering dogwood (*Cornus florida*); (8) maple (*Acer rubrum* and *A. saccharum*); and (9) other species (*Celtis* spp., *Diospyros virginiana*, *Juglans nigra*, *Morus* spp., *Prunus serotina*, *Ulmus* spp., and others).

We selected permanent sample plots spanning a site quality gradient to assess the influence of site quality on compositional dynamics. MOFEP permanent sample plots were originally stratified within each site according to ecological land types (ELT) as delineated based on variation in slope position and aspect (Brookshire and Shifley 1997). We selected plots occurring on four ELTs common to MOFEP sites (Table 1). ELTs 3 and 5 occur at upper and lower positions, respectively, on south-facing slopes (i.e., exposed slopes), while ELTs 4 and 6 are found at upper and lower positions on north-facing slopes (i.e., protected slopes). According to site index, these four ELTs can be arranged in order of increasing site quality as follows: ELT 3 < ELT 5 < ELT 4 < ELT 6 with site index (SI_{50}) = 65, 69, 72, and 75, respectively, based on black oak at base age 50. Although these

Table 1.—Description of ecological land types (ELT) used in this study as part of the Missouri Ozark forest ecosystem project (MOFEP) experiment

ELT	Slope aspect	Slope position	Base saturation	Site index ₅₀ (feet)	Extent in MOFEP study area (%)
3	135-314°	Upper backslope	Low	65 ^a	21
5	135-314°	Lower backslope	Moderate	69	10
4	315-134°	Upper backslope	Low	72	18
6	315-134°	Lower backslope	Moderate	75	9

^a Site index is based on black oak at base age 50, and ELTs are arranged in order of increasing site quality. Table modified from Kabrick et al. (2008b).

ELTs are more common to MOFEP sites relative to others, ELTs 5 and 6 each represent about 10 percent of MOFEP study sites.

In order to assess compositional changes between two sampling years, 1995 and 2010 densities of each taxonomic group by size class were analyzed by analysis of variance (ANOVA) using a randomized complete block design with repeated measures. For ANOVA models, ELT was included as a fixed effect and site was used as a blocking factor (random effect). Since our main interest was in assessing changes in abundance in relation to site quality, we used contrasts to compare densities between years by species and size class individually for each ELT. ANOVA models and contrasts were run using PROC MIXED (SAS 9.2, SAS Institute Inc., Cary, NC). Statistical significance was assessed at $\alpha=0.05$.

RESULTS

Overstory

On upper, south-facing slopes (ELT 3), red oak overstory density was significantly lower in 2010 (50 TPA) than 1995 (68 TPA) (Fig. 1A). The density of overstory white oak, on the other hand, was significantly greater in 2010, increasing from 41 to 52 TPA. The densities of blackgum and dogwood were also significantly greater in 2010, but each made up only a fraction of total overstory density in both years. White oak overstory BA was significantly greater in 2010, and red oak BA was substantially larger than that of other species (Fig. 2A).

The density of overstory red oak on lower, south-facing slopes (ELT 5) was significantly lower in 2010 (Fig. 3A), dropping from 65 to 41 TPA. As a result of this decrease, white oak was the most abundant taxon in the overstory in 2010 (52 TPA). There were no differences detected in overstory BA between years (Fig. 2B). Based on BA, red oak was the most dominant overstory taxa in both years (>43 square feet per acre).

The density of red oak overstory trees on upper, north-facing slopes (ELT 4) was significantly lower in 2010, whereas maple and blackgum densities were significantly greater after 15 years (Fig. 4A). However, both maple and blackgum comprised only a minor component of total overstory density. White oak replaced red oak as the most abundant overstory tree by 2010. No significant changes in BA were detected over the 15 years where the overstory remained dominated by oak and hickory species (Fig. 2C).

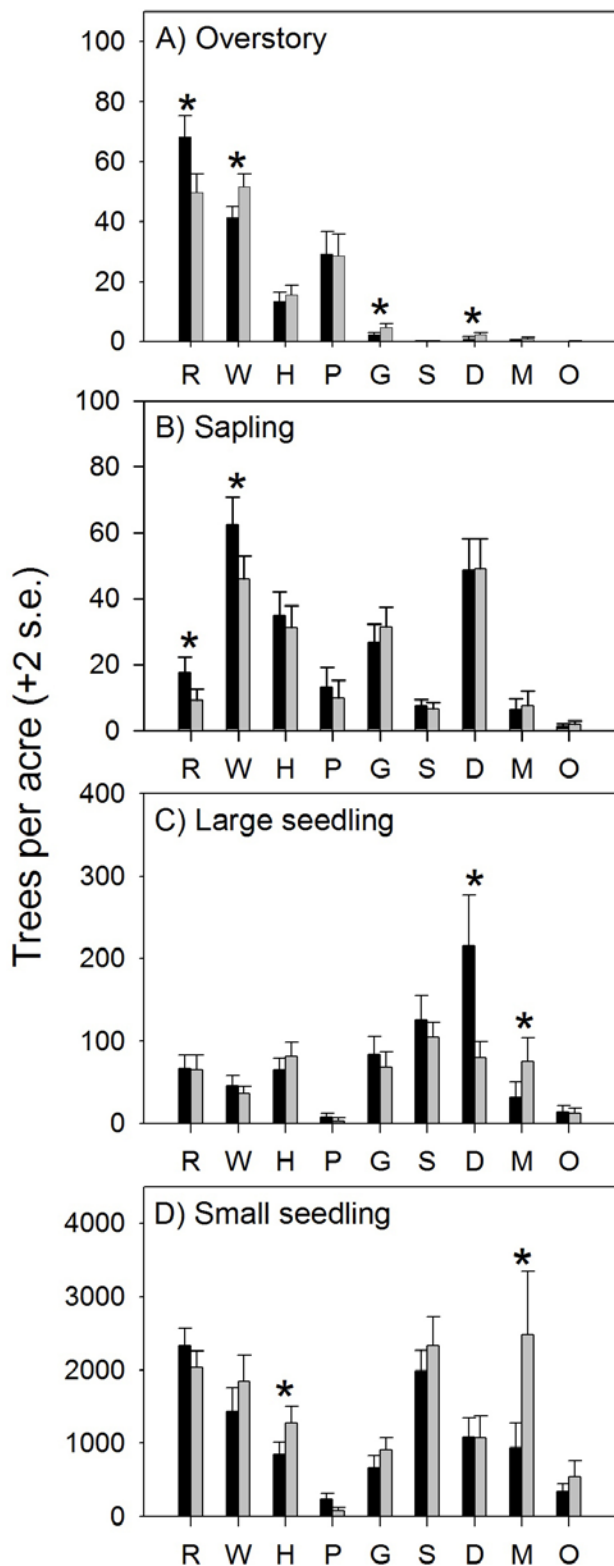


Figure 1.—Mean number of trees per acre (± 2 standard errors) in 1995 (black bars) and 2010 (gray bars) for (A) overstory, (B) sapling, (C) large seedling, and (D) small seedling size classes on ELT 3 in untreated control sites of the Missouri Ozark Forest Ecosystem Project. Abbreviations are: R=red oak, W=white oak, H=hickory, P=shortleaf pine, G=blackgum, S=sassafras, D=dogwood, M=maple and O=other species. Asterisk indicates a significant difference between years for a taxonomic group within size class ($p < 0.05$).

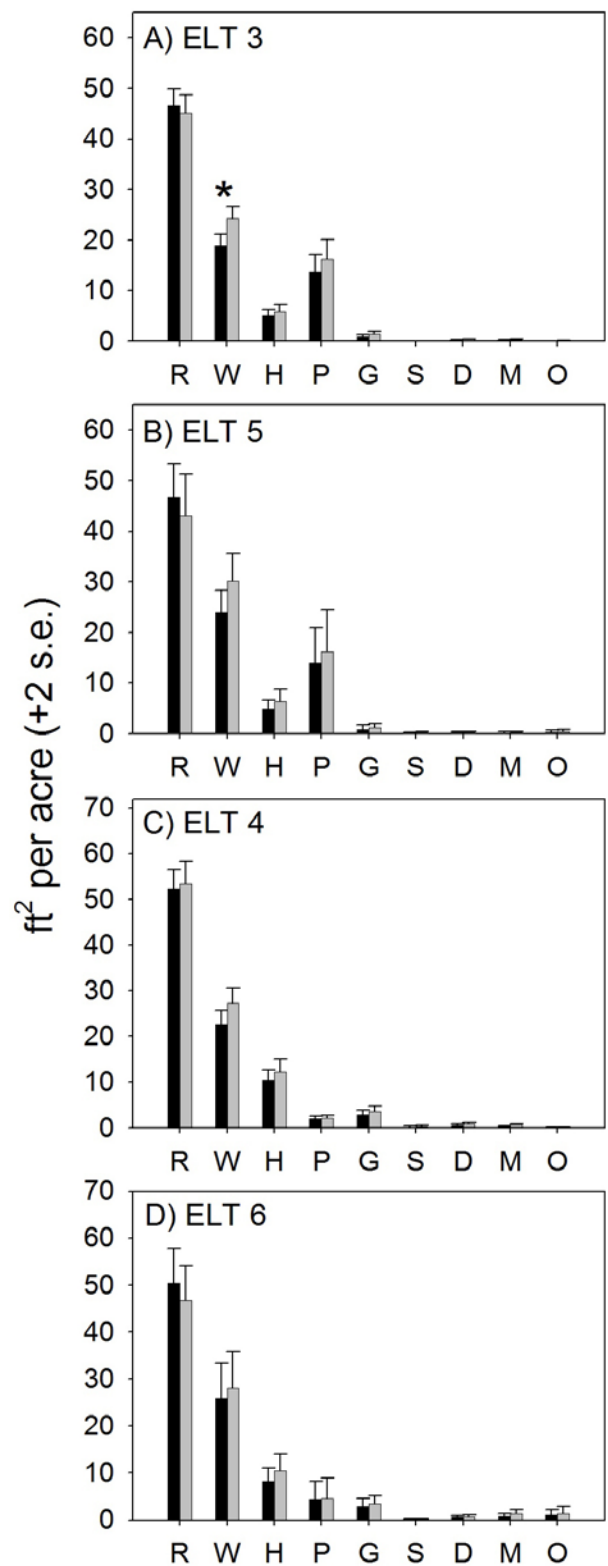


Figure 2.—Mean basal area (± 2 standard errors) in 1995 (black bars) and 2010 (gray bars) for overstory taxa on different ecological land types (ELT) in the untreated control sites of the Missouri Ozark Forest Ecosystem Project: (A) ELT 3, (B) ELT 5, (C) ELT 4, and (D) ELT 6. Abbreviations are: R=red oak, W=white oak, H=hickory, P=shortleaf pine, G=blackgum, S=sassafras, D=dogwood, M=maple and O=other. Asterisk indicates a significant difference between years for a taxonomic group within size class ($p < 0.05$).

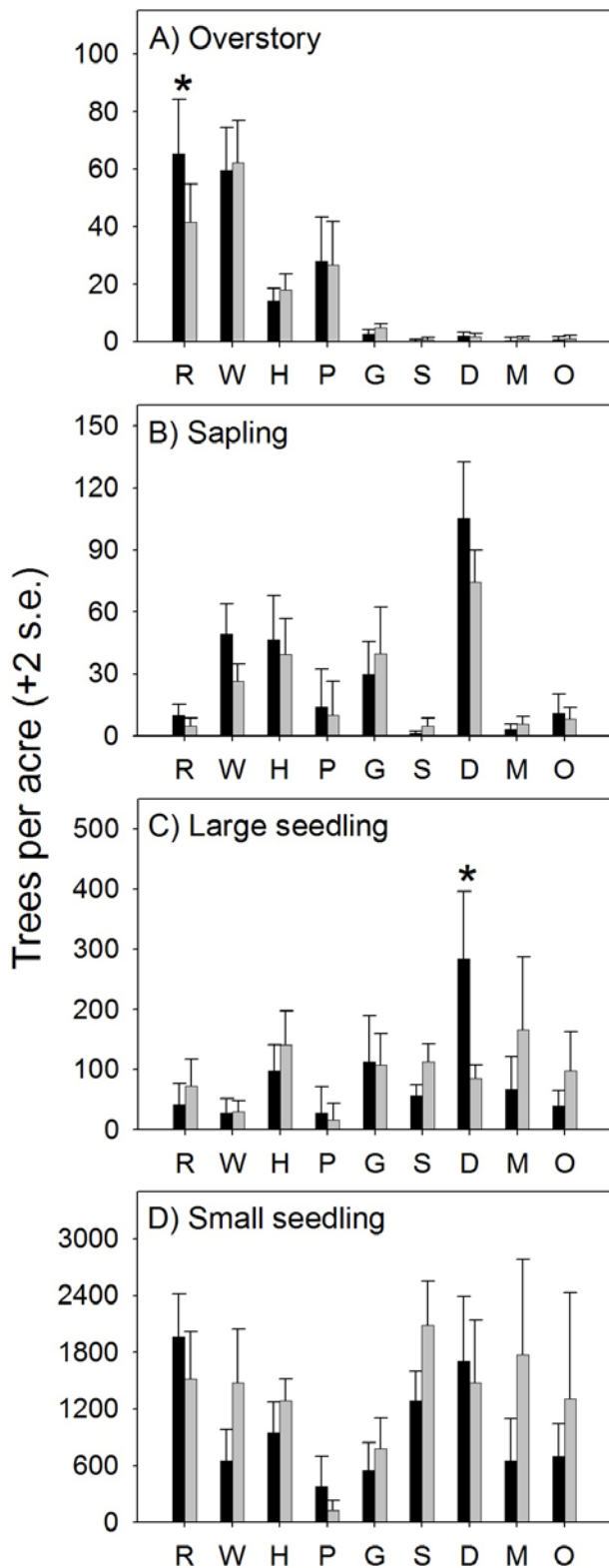


Figure 3.— Mean number of trees per acre in 1995 (black bars) and 2010 (gray bars) for (A) overstory, (B) sapling, (C) large seedling, and (D) small seedling size classes on ELT 5 in untreated control sites of Missouri Ozark Forest Ecosystem Project. Abbreviations are: R=red oak, W=white oak, H=hickory, P=shortleaf pine, G=blackgum, S=sassafras, D=dogwood, M=maple and O=other species. Asterisk indicates a significant difference between years for a taxonomic group within size class ($p < 0.05$).

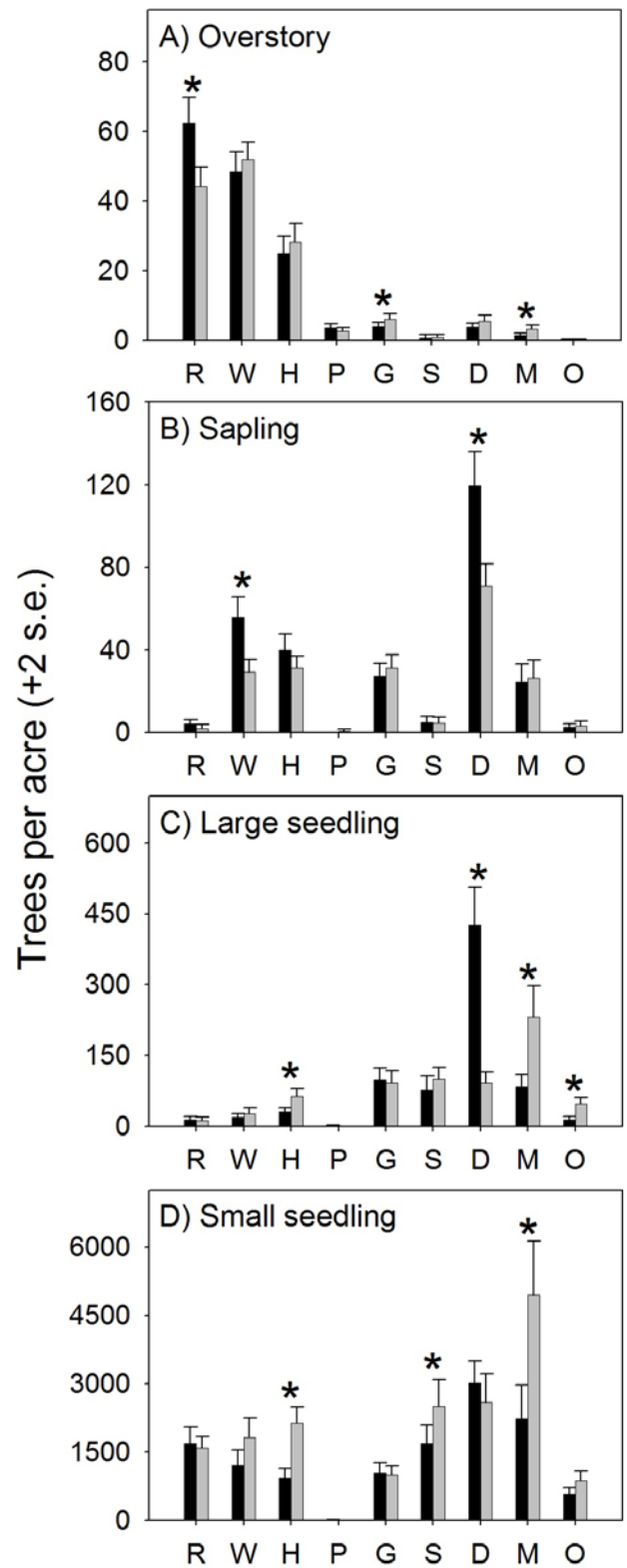


Figure 4.— Mean number of trees per acre in 1995 (black bars) and 2010 (gray bars) for (A) overstory, (B) sapling, (C) large seedling, and (D) small seedling size classes on ecological ELT 4 in untreated control sites of the Missouri Ozark Forest Ecosystem Project. Abbreviations are: R=red oak, W=white oak, H=hickory, P=shortleaf pine, G=blackgum, S=sassafras, D=dogwood, M=maple and O=other species. Asterisk indicates a significant difference between years for a taxonomic group within size class ($p < 0.05$).

On lower, north-facing slopes (ELT 6), red oak overstory density was significantly lower in 2010 (Fig. 5A). Although not statistically significant, the density of white oak overstory trees were nominally lower and hickory was nominally greater in 2010. White oak was the most abundant overstory taxon in both years. Red oak was the second most abundant in 1995, whereas hickory density was slightly greater than red oak by 2010. Red oak and white oak were the dominant taxa according to BA (Fig. 2D).

Sapling

White oak was the most abundant species group of the sapling layer on ELT 3 in 1995 (Fig. 1B). Red oak and white oak sapling densities were significantly lower in 2010 than in 1995, dropping from a mean of 18 to 9 TPA and 62 to 46 TPA, respectively. Despite this significant decrease, the relative density of white oak saplings was still high in 2010. Dogwood was the most abundant species in the sapling size class in 2010. Maple sapling density was less than 10 TPA in both periods.

No differences were detected between years for any taxa in the sapling class on ELT 5 (Fig. 3B). However, there were notable numerical decreases in density over the 15 years. White oak and dogwood sapling densities were nominally lower in 2010. Dogwood remained the most abundant taxon in the sapling layer despite the large decrease in mean density.

Dogwood was the most abundant sapling on ELT 4 in 1995 (119 TPA; Fig. 4B) and maintained this status in 2010 despite a large reduction in density (71 TPA). Sapling densities of white oak were significantly lower in 2010. On ELT 4, white oak sapling density declined by nearly 50 percent over the 15-year period (56 vs. 29 TPA). Red oak was a minor component of the sapling layer. Maple sapling density changed little over the 15 years but was comparable to that of oak saplings by 2010.

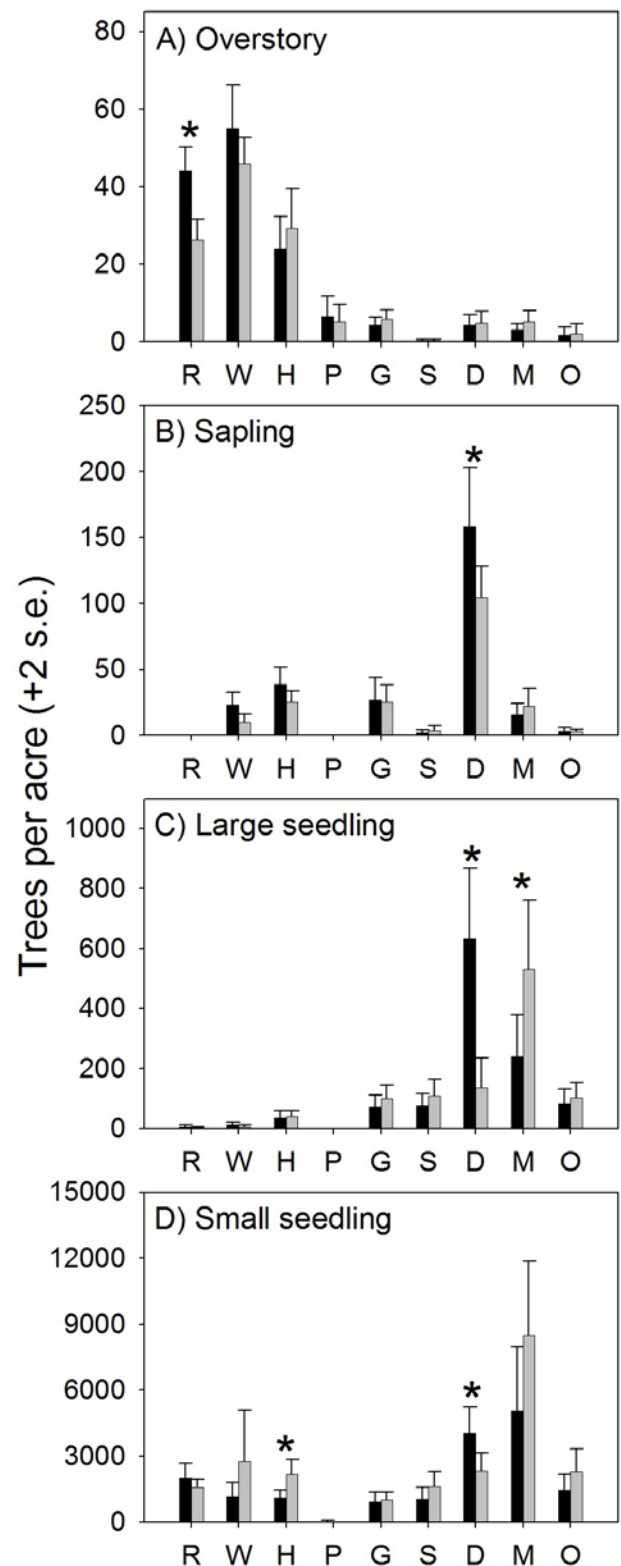


Figure 5.— Mean number of trees per acre in 1995 (black bars) and 2010 (gray bars) for (A) overstory, (B) sapling, (C) large seedling, and (D) small seedling size classes on ELT 6 in untreated control sites of Missouri Ozark Forest Ecosystem Project. Abbreviations are: R=red oak, W=white oak, H=hickory, P=shortleaf pine, G=blackgum, S=sassafras, D=dogwood, M=maple and O=other species. Asterisk indicates a significant difference between years for a taxonomic group within size class ($p < 0.05$).

Dogwood was the most abundant sapling at both sampling periods on ELT 6, although it was significantly lower in 2010 (158 and 104 TPA; Fig. 5B). The mean densities of white oak and hickory saplings were nominally lower in 2010, whereas mean density of maple saplings increased slightly. No red oak saplings occurred in plots in either sampling period.

Large Seedlings

On ELT 3, dogwood was the most abundant taxon of large seedlings in 1995 (Fig. 1C). However, dogwood density was significantly lower in 2010 than 1995, dropping from a mean of 215 TPA to 80 TPA. In contrast, maple density was significantly greater in 2010, doubling over the 15-year period (31 vs. 75 TPA). Mean densities of red and white oaks showed very little change, making up only a small proportion of large seedlings. By 2010, sassafras was the most abundant taxon.

Dogwood was the most abundant taxon of large seedlings on ELT 5 in 1995 (283 TPA; Fig. 3C). However, the density of large dogwood seedlings was significantly lower by 2010 (85 TPA). Although not significant, mean densities of several taxa in the large seedling class were nominally greater in 2010 than 1995; most notable were increases in maple and other species. Red oak density also showed a sizeable increase over 15 years (42 vs. 73 TPA), whereas white oak density remained largely unchanged. Both oak species groups were a minor component of large seedlings in both years. By 2010, maple was the most abundant large seedling (167 TPA).

The number of large dogwood seedlings decreased significantly over the 15-year period on ELT 4 (427 vs. 93 TPA; Fig. 4C). Large seedling densities of maple, hickory, and other species were significantly greater in 2010 than in 1995, with maple being the most abundant taxon after 15 years (231 TPA). The oaks only made up a small fraction of large seedlings (<30 TPA).

There was a significant decrease in large dogwood seedlings on ELT 6 (Fig. 5C). The large decline in dogwood was offset by an increase in maple density, which was significantly greater in 2010 (239 vs. 532 TPA). As of 2010, maple was the most abundant large seedling. Both oak species groups made up a minor fraction of large seedlings (<11 TPA).

Small Seedlings

On ELT 3, small maple and hickory seedling densities were significantly greater in 2010 than 1995 (Fig. 1D). The density of small maple seedlings was 2.5 times greater after 15 years (2,482 vs. 939 TPA). Maple went from the fifth most abundant in 1995 to most abundant species group of the small seedling size class in 2010. Although no difference was detected between years, red oak density was nominally lower in 2010 while white oak density was greater after 15 years. Small sassafras seedling density remained high over the 15-year period (1,985 and 2,336 TPA in 1995 and 2010, respectively).

There were no significant differences in small seedling density detected between years on ELT 5 (Fig. 3D). Red oak was the most abundant species group in the small seedling class in 1995 (1,960 TPA). Although not significant, mean densities of white oak, sassafras, maple, and other species were considerably greater in 2010. Sassafras was the most abundant taxon by 2010 (2,087 TPA).

On ELT 4, densities of small hickory, sassafras, and maple seedlings were significantly greater in 2010 (Fig. 4D). Dogwood was the most abundant taxon in the small seedling class in 1995 (3,005 TPA). By 2010, maple was the most abundant (4,944 TPA) and had substantially greater density than other taxa. Although not significant, mean density of small white oak seedlings was nominally greater in 2010.

The density of small hickory seedlings was significantly greater in 2010 on ELT 6, whereas the density of dogwood was significantly lower (Fig. 5D). Mean densities of maple and white oak were nominally greater in 2010. Mean density of small maple seedlings increased from 5,058 to 8,484 TPA over 15 years, whereas mean density of small white oak (1,150 to 2,759 TPA) and hickory (1,104 to 2,284 TPA) more than doubled.

DISCUSSION

Mesophication is a process of shifting forest composition from fire-adapted to fire-sensitive species under fire exclusion (Nowacki and Abrams 2008). Maples are the most common mesophytic, fire-sensitive species in the Missouri Ozarks, and therefore we postulated that increasing abundance of maple species would provide early evidence of mesophication in this region. Our study showed that maple seedling densities (small and large seedlings) consistently increased across a gradient of site quality. Furthermore, maple was a major component of the seedling layer on all ELTs by 2010 and was among the most common taxa on better quality sites. Sites used for this study have not been burned in at least a half century suggesting that the observed increase in maple regeneration densities could be an early indicator of mesophication in Ozark upland forests.

The process of mesophication starts with the establishment of shade-tolerant, fire-sensitive species followed by their recruitment into larger size classes. Maple seedlings were relatively abundant by 2010, whereas maple sapling density still remained low. This pattern suggests that mesophication may be in an early stage at these sites. Historically, maple was relegated to less fire-prone sites in the Ozarks, such as mesic toe slopes, upland waterways, or rocky sites with shallow soil, but has expanded into adjacent uplands over the last two centuries along with other fire-sensitive species (Hanberry et al. 2012). Frequent, intense drought experienced in the Ozarks could also limit recruitment of non-oak species, such as maple (Johnson et al. 2009, Kabrick et al. 2008b, Nigh et al. 1985). Maple sapling densities were up to nine times greater on protected slope ELTs compared to those on exposed slopes suggesting stronger recruitment limitation on drier, south-facing slopes in this study.

Another potential outcome of mesophication is displacement of fire-tolerant species by fire-sensitive species in the understory. Oak, hickory, and shortleaf pine are the primary fire-tolerant tree species groups of the Missouri Ozarks. Although there were no significant differences detected for oak seedling density, numerical differences in mean density suggested that white oak tended to increase and red oak tended to decline over the study period. This difference between oak species groups is likely related to the greater shade tolerance of white oak (Johnson et al. 2009). Hickories, which are slightly more shade tolerant than are many of the oaks (Burns and Honkala 1990), increased in all seedling layers over the 15-year period, and increases were significant on nearly all ELTs. Shortleaf pine comprised only a small fraction of the seedling layers and, when present, densities declined. The low abundance of shortleaf pine regeneration in these mature, relatively undisturbed forests is mainly due to this species' low tolerance of shade and requirement of exposed mineral soil for seedling

establishment (Lawson 1990). These findings suggest that not all fire-tolerant species regeneration is decreasing because of mesophication, but rather some decreases in regeneration abundance are simply due to limited availability of light and suitable seedbeds.

A low relative density of large oak advance regeneration compared to shade-tolerant species has been cited as an indicator of an oak-dominated forest undergoing successional replacement by fire-sensitive species (Abrams and Downs 1990, Abrams et al. 1997). Nowacki and Abrams (1992) refer to this as an oak sapling bottleneck. In this study, large seedlings and saplings were considered large advance regeneration. Relative to oak, large maple seedlings were more abundant on all ELTs except ELT 3 where oaks had a numerical advantage in both the large seedling and the sapling layers. ELT 3 is the most xeric of the ELTs considered in this study. This finding is consistent with our current knowledge of Ozark forests; specifically, that oaks are more resilient against non-oak displacement on drier sites compared to mesic sites (Johnson et al. 2009). However, sapling densities of oak and hickory decreased on all ELTs over 15 years, whereas maple sapling density was nominally greater by 2010.

The overstories of all ELTs remained dominated by oak and hickory species over the 15-year period, with pine a major component on south-facing sites (ELT 3 and ELT 5). However, red oak density significantly decreased on all ELTs, while BA nominally decreased. This reduction in red oak density was likely the result of oak decline, a widespread disease complex impacting mainly species of the red oak group, particularly black and scarlet oak (Kabrick et al. 2008a, Voelker et al. 2008). Along with this ubiquitous decrease in red oak were increases in the abundance of white oak in the overstories on all ELTs. White oak is less susceptible to decline than the red oak group. Using the same study sites, Shifley et al. (2006) observed that red oaks experienced three and a half times the mortality rate of white oaks. The high abundance of mature red oak species at these sites suggests that oak decline will continue to reduce the abundance of red oak and provide recruitment opportunities for white oak. Red oak decline could also release shade-tolerant, fire-sensitive species and facilitate their recruitment, thereby accelerating succession (Abrams and Nowacki 1992), and possibly mesophication, of these and similar Ozark forests.

There were several other notable trends that came out of this study. Perhaps the most conspicuous were the large declines in dogwood abundance, which was most evident on north-facing slopes (ELT 4 and ELT 6). Oswalt et al. (2012) observed significant range-wide declines in dogwood populations estimated from U.S. Forest Service Forest Inventory and Analysis data. The authors largely attributed the decline to mortality caused by the nonnative fungus *Discula destructiva* (dogwood anthracnose) but also cited increasing forest density, drought, and competition with shade-tolerant tree species, such as sugar maple, as other possible causes. Dogwood anthracnose is not considered a major issue in Missouri at this time², so the declines we observed were likely related to stress associated with drought and competition. Since dogwood is a major understory competitor in the Ozarks (Dey et al. 1996), dogwood decline would create recruitment opportunities for oak and non-oak species alike. Over the 15-year period of our study, sassafras developed into a major component of the regeneration layers on several ELTs, particularly on ELT 3 where it was the most abundant species in the small and large

² Wright, S. 2014. Personal communication. Forest Pathologist, Missouri Department of Conservation, 3500 E. Gans Road, Columbia, MO 65201.

regeneration classes. Although sassafras is considered shade intolerant (Griggs 1990), sassafras is often a part of the advance regeneration layer of oak-hickory forests in the Missouri Ozarks (Grabner 2000, Hartman and Heumann 2003, Shifley et al. 2000). The density of the other species group, composed of an ecologically diverse collection of species (including mesophytes), also tended to increase in regeneration layers. Several of these species are capable of developing into large overstory trees and, therefore, could potentially compete with oak and hickory for overstory positions. In particular, black cherry (*Prinus serotina*) is often locally dominant during early stand development following clearcutting in the Ozarks, yet it is conspicuously absent from mature stands. Frequent and intense drought likely limits the development of black cherry in Ozark upland forests.

CONCLUSIONS

The Ozarks are known for supporting a higher proportion of oak and hickory species than other forested areas of the CHR, even in the midst of fire suppression (Johnson et al. 2009). The unique combination of climate, soil quality, site conditions, and forest composition has led some to postulate that the oak-hickory forest type is successional stable in the Ozarks (Pallardy 1995). However, there is evidence that mesophytic species, particularly maples, are not only capable of replacing oak overstory trees in the Ozarks but that this process is already underway (Hanberry et al. 2012, Hanberry et al. 2014, Nigh et al. 1985). We documented increases in maple abundance in the understories of upland Ozark forests over a 15-year period. During this same period, the densities of oak and hickory regeneration tended to increase, but their relative abundance was lower than that of maples by 2010 on all but the driest sites. Maple saplings and overstory trees, on the other hand, were a minor component of these forests, while oak and hickory dominated large size classes. The findings presented here along with findings of other recent studies (Hanberry et al. 2012, Hanberry et al. 2014) suggest these and similar upland forests in the Missouri Ozarks could be in early stages of mesophication where fire has been excluded for at least 50 years, that this process may be occurring at a slower rate than in eastern and central parts of the CHR, and that the rate of mesophication is slowest on xeric, south-facing slopes. It also could be plausible that the predominance of low quality soils and frequent drought in the Ozarks will hinder these forests from experiencing much mesophication. This conclusion is consistent with research out of Oklahoma on sites that were more xeric than the ones considered in this study (Burton et al. 2010, DeSantis et al. 2010, Thomas and Hoagland 2011). However, since our findings are based on only 15 years of stand dynamics, which represents just 15-20 percent of a typical even-aged forest rotation (80-100 years), our conclusions are preliminary at this time and are best treated as working hypotheses. Continued monitoring and future analysis will be necessary for testing and refining our working hypotheses regarding successional status of Ozark forests under fire exclusion.

ACKNOWLEDGMENTS

We thank the Missouri Department of Conservation (MDC) for continued support of the MOFEP experiment and access to long-term data. We also thank Sherry Gao, MDC Biometrician, for statistical advice.

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SILVICULTURE

TEN-YEAR RESULTS OF USING OAK CLEANINGS TO MAINTAIN OAK SPECIES DOMINANCE ON THE ALLEGHENY NATIONAL FOREST

Kurt W. Gottschalk, Gary W. Miller, Robert White, Andrea Hille, and Thomas M. Schuler¹

Abstract.—The Allegheny National Forest (ANF) in northwestern Pennsylvania implemented precommercial thinning in young stands to maintain oak (*Quercus* spp.) stems in a competitive position. This administrative study was developed to test ANF standards for precommercial thinning for success in maintaining oak composition. An additional objective was to examine stand development and competitive patterns of these young mixed stands. Two component studies were installed. One study area was in a 25- to 30-year-old stand where we released intermediate and suppressed oak species, in particular white oak (*Q. alba*), to see if they would survive and improve in crown class. The other study was installed in 16-year-old stands where crop tree selection and release treatments using the ANF standards were done, with untreated control plots reserved. Eight treated and eight control plots in two stands were treated during the winter of 2000-2001 and followed for 10 years.

In the first study, mortality of intermediate and suppressed white oak stems was 84 and 38 percent in control versus treated areas. The crop tree release treatments were successful in keeping trees alive, but crown vigor declined and no trees increased in crown class. In the second study, the mortality of codominant crop trees was 13 and 2 percent in control versus treated stands. The proportion of oak stems remaining at least codominant was 59 and 78 percent for control versus treated trees. Stand-level oak composition in control stands was 11.3 percent after 10 years (11.7 percent initially). In treated stands, it was 17.7 percent after 10 years (10.4 percent initially). The crop tree release treatments successfully increased survival, growth, and composition of oak.

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

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TEN-YEAR RESPONSE OF COMPETING VEGETATION AFTER OAK SHELTERWOOD TREATMENTS IN WEST VIRGINIA

Gary W. Miller, James N. Kochenderfer, Jeffrey D. Kochenderfer, and Kurt W. Gottschalk¹

Abstract.—Successful oak regeneration depends on the relative status of advanced oak reproduction and associated competing woody vegetation present when harvests or other stand-replacing disturbances occur. This study was installed to quantify the effect of microsite light availability and deer browsing on the development of advanced northern red oak (*Quercus rubra*) seedlings and competing vegetation in 80-year-old, mixed mesophytic Appalachian hardwood stands dominated by northern red oak. Advanced oak seedlings and competing woody species were monitored in forty-eight 0.4-acre permanent plots for 10 years. Microsite light was manipulated with herbicide injection and cut-stump treatments to stems in the intermediate and suppressed crown classes. Twelve plots were randomly assigned to four microsite light levels: Control, Low, Medium, and High. Eight plots in each treatment were randomly assigned to receive protection from deer browsing by a woven wire fence. The major competing vegetation included black cherry (*Prunus serotina*), American beech (*Fagus grandifolia*), sweet birch (*Betula lenta*), red maple (*Acer rubrum*), striped maple (*A. pensylvanicum*), and yellow-poplar (*Liriodendron tulipifera*). The development of competing vegetation in each treatment combination was compared 1, 3, 5, 7, and 10 years after treatment. Both microsite light level, as measured by photosynthetically active radiation (PAR), and fencing had a significant effect on the abundance and height class of competing vegetation. Competing vegetation increased at higher PAR levels and within fenced plots at a moderate pace for the first 5 years after treatment, followed by a surge in height growth in the second 5 years after treatment. After 10 years, sweet birch was the most aggressive competitor in the treated plots, with nearly 10,000 stems per acre ≥ 3 feet tall in fenced plots with the highest PAR levels. Guidelines for prescribing similar preparatory treatments and a discussion of management implications for long-term oak regeneration success are provided.

INTRODUCTION

Regenerating northern red oak (*Quercus rubra*) on high-quality growing sites is a continuing problem in the central Appalachian region. New stands that develop after overstory harvests often contain fewer oaks than the preceding stand. The basic problem is that overstory harvests are applied when an insufficient number of competitive advanced oak seedlings are present to compete with other hardwood species after the harvest. As a result, few oaks ascend into the canopy of the new stand to replace the parent trees that were removed. Instead, competing species occupy a greater proportion of the new stand and the proportion of oaks in the overstory falls short of management objectives.

Successful oak regeneration is related to the size and number of advanced seedlings present when harvests occur (Loftis 1990a, Sander et al. 1984). For example, on northern red oak site index 80 (base age 50 years), the probability that an advanced oak seedling with a 0.1-inch basal diameter will

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become dominant or codominant 20 years after a harvest is essentially zero (Loftis 1990a). Although numerous small seedlings may be present before a harvest, very few will compete successfully after the harvest because of their small initial size. This probability increases to 1 percent for a 0.2-inch basal diameter, and to 8 percent for a 0.75-inch basal diameter. As the seedling size and probability of success increase, fewer seedlings are needed to obtain adequate regeneration after a harvest. A preharvest inventory of advanced oak seedlings is recommended to determine if there will be a sufficient oak component in the new stand (Loftis 1990a). If projected oak regeneration is insufficient, silvicultural treatments may be needed to increase the growth and survival of advanced seedlings before the overstory is removed (Loftis 1990b).

The species composition of a new stand is determined by competition among the species present in various forms at the time of the disturbance, following the “initial floristics composition” model suggested by Egler (1954). After the overstory harvest, numerous woody and herbaceous species compete for the available sunlight, water, and nutrients. The sources of regeneration include: (1) new seedlings from seed stored in the forest floor; (2) sprouts from cut stumps, wounded roots, and broken shoots; and (3) advanced seedlings that developed before the disturbance. Species that compete with oaks on mesic sites often exhibit faster initial height growth than new oak seedlings and small advanced oak seedlings. If the oaks are not able to keep pace with competing species in the early stages of development, they usually die as the new overstory canopy closes above them (Trimble 1973). Oak stump sprouts are usually competitive with other species, but they contribute relatively few new stems on mesic sites (Loftis 1983b, Sander 1988). As a result, successful oak reproduction on mesic sites comes primarily from relatively large advanced oak seedlings. If large advanced oak seedlings are lacking before the harvest, then competing species usually dominate the composition of new stands (Beck and Hooper 1986).

In undisturbed mature oak stands, advanced oak seedlings usually exhibit both poor survival and slow growth. In one study, the survival of a cohort of northern red oak seedlings that germinated after a good acorn crop steadily declined from 60 percent after 1 year to only 10 percent after 10 years (Beck 1970). Similarly, the average total height of survivors was less than 1 foot after 1 year and generally did not increase over the next 10 years. As mixed oak stands in the central Appalachians approach maturity, adequate advanced oak reproduction usually does not develop due to several factors: (1) acorns are consumed or damaged by deer, insects, rodents, and birds; (2) advanced oak seedlings and sprouts are browsed by deer; and (3) cohorts of new seedlings are suppressed and killed by excessive shade from dense interfering vegetation in the midstory and understory strata. These conditions call for preparatory treatments that reduce acorn predation, reduce deer browsing of established seedlings, and reduce interfering plants so that advanced oak seedlings can grow to sufficient sizes before the parent trees are removed (Lorimer 1992, Marquis 1981, Marquis et al. 1976, Tilghman 1989).

Forest managers can increase the probability of successful oak regeneration by prescribing preparatory treatments that enhance the size and competitiveness of advanced oak seedlings several years before a planned overstory harvest (Beck 1988, Carvell and Tryon 1961, Gottschalk 1983, Hannah 1987, Leak et al. 1987, Loftis 1990b, Marquis et al. 1992, Sander and Clark 1971). Shelterwood treatments reduce stand density and increase the amount of sunlight and other site resources available to advanced oak seedlings. The added sunlight increases both survival and growth of advanced oak

seedlings, thus increasing the abundance and competitiveness of advanced oaks in the next stand. In the southern Appalachians, shelterwood treatments that removed more than 50 percent of the stand basal area stimulated the growth of advanced oak seedlings, but also stimulated the development of competing species such as sweet birch (*Betula lenta*) and yellow-poplar (*Liriodendron tulipifera*), particularly where canopy gaps were created (Loftis 1983a). Alternative shelterwood treatments that removed only 30 percent of stand basal area from below the overstory canopy, with no canopy gaps, increased survival and growth of advanced oak seedlings without stimulating the development of competing species (Loftis 1988). Although similar treatments have not been tested in the central Appalachians, advanced oak reproduction was found to be more abundant in mature stands where the overstory canopy is closed and the subcanopy density is relatively sparse (Miller 1997, Schuler and Miller).

An important factor to consider in prescribing a shelterwood treatment is the amount of sunlight needed to enhance the survival and growth of advanced oak without overstimulating their competitors. At very low levels of microsite light, the oak seedlings will not respond. At very high levels of microsite light, the response of competing species may surpass that of the oaks. If prescribed fire or other remedial treatments are not available to control competing vegetation several years after a heavy shelterwood removal cut (Brose et al. 1999), a gentler approach may be necessary. Forest managers in the central Appalachian region need a reliable and efficient treatment for developing adequate advanced oak reproduction before harvest operations that does not require a followup prescribed fire. In many cases, prescribed fire may be inconsistent with landowner objectives or policy constraints. A relatively light-handed shelterwood treatment is needed to enhance the survival and growth of advanced oak seedlings while limiting the response of their competitors.

This study examined the effect of various light levels and deer fencing on the development of competing woody vegetation after noncommercial shelterwood treatments that reduced only the midstory density as described by Loftis (1990b). The overstory canopy was left intact, thereby increasing sunlight on the forest floor by relatively small increments.

STUDY SITES

The study was installed in 80-year-old second-growth central Appalachian hardwood stands on the Monongahela National Forest in northern Randolph County, West Virginia. Overstory trees in the study area regenerated after landscape-scale logging operations that were conducted between 1915 and 1920. In 1998, northern red oak accounted for 59 percent of the basal area. Yellow-poplar, black cherry (*Prunus serotina*), American beech (*Fagus grandifolia*), red maple (*Acer rubrum*), sugar maple (*A. saccharum*), and cucumbertree (*Magnolia acuminata*) also occupied significant proportions of the overstory. Annual precipitation in the study area averages 59 inches and is evenly distributed throughout the year. Soils are described as Dekalb channery loam (loamy-skeletal, mixed, mesic Typic Dystrochrept) (Soil Conservation Service 1967). The study area is located on site index 80 for northern red oak (base age 50). Several layers of dense subcanopy vegetation (trees ≤ 10 inches diameter at breast height [d.b.h.]) were present in the suppressed and intermediate crown classes before treatments were applied. This vegetation included striped maple (*A. pensylvanicum*), American beech, red maple, and sugar maple. There were approximately 20 to 25 white-tailed deer (*Odocoileus virginianus*) per square mile in the study area.

METHODS

In 1998, a total of 48 square treatment plots were installed on the study site. Each plot was 0.4 acre in size and included a 0.1-acre measurement plot surrounded by a similarly treated buffer. Within each measurement plot, the initial stand inventory included species, d.b.h., and crown class of all trees ≥ 1 inch d.b.h.

Within each 0.1-acre measurement plot, competing woody vegetation was tallied in each of nine circular 0.001-acre subplots, whose center points were marked with a steel rod. The tally included the number of stems by species and the following five height classes: height < 0.5 feet, 0.5 to 0.9 feet, 1 to 2.9 feet, 3 to 4.9 feet, and ≥ 5 feet. Competing woody vegetation was tallied before treatment and again 1, 3, 5, 7, and 10 years after treatment.

Photosynthetically active radiation (PAR) was measured within each plot in late July before treatment and each year after treatment to quantify changes in microsite light. It was measured with synchronized Accupar[®] Ceptometers (Decagon Devices, Inc., Pullman, WA) 3 feet above the ground at a fixed location in a nearby open field and at 10 designated points within each measurement plot. Measurements in the open were compared to mean measurements within the plots at synchronized times to determine percent PAR associated with each plot (Gendron et al. 1998, Parent and Messier 1996).

Treatments

The treatments included three levels of the shelterwood method described by Loftis (1990b). This method removes trees from below the overstory canopy, starting with the smallest trees and including trees in progressively larger d.b.h. classes until a desired threshold for removal is reached. Trees were removed by using an approved herbicide applied to the cut stumps of trees < 1 inch d.b.h. or injected by using a hack-and-squirt method for larger trees. The herbicide solution used in all treatments was 41 percent glyphosate active ingredient diluted to 50 percent of full strength in water (20.5 percent glyphosate active ingredient). Each injected tree received one incision 1.75 inches long and 1.5 ml of herbicide solution per inch of d.b.h. Twelve plots were randomly assigned to each of the three treatments described below for a total of 48 plots. Oaks were not removed in any treatment as they serve as a desired source of reproduction for the future. The herbicide treatments were applied in late July 1999.

Control

No vegetation was cut or treated with herbicide.

Low

All woody stems ≥ 2 feet tall and < 1 inch d.b.h. were severed near the ground and their stumps were sprayed with herbicide. In addition, woody stems ≤ 2 inches d.b.h. were injected with herbicide.

Moderate

This treatment included all stems removed in the Low treatment. In addition, all stems ≤ 7 inches d.b.h. were injected with herbicide.

High

This treatment included all stems in both the Low and Moderate treatments plus all remaining stems in the suppressed and intermediate crown classes.

Fenced or Unfenced

Eight plots in each of the Control, Low, Medium, and High treatments were protected from deer by a 6.5-foot-tall woven wire fence, and four plots in each treatment were not protected. The fences were erected in August 1998 and maintained until the conclusion of the study in 2009.

Data Analysis

Statistical analyses were completed to provide insight into two important relationships: (1) the effect of the herbicide treatments on microsite light as measured by percent PAR and (2) the effect of the herbicide treatments and fencing on the development of competing woody vegetation as measured by the number of stems present in three height classes.

A one-factor repeated measures analysis of variance (ANOVA) was used to examine the effect of the herbicide treatments on microsite light. The fixed effect model has the form:

$$Y_{ij} = \mu + \alpha_i + \beta_j + (\alpha\beta)_{ij} + \varepsilon_{ij}$$

where

Y = percent PAR,

μ = the overall mean,

α = the effect of the herbicide treatment,

β = the effect of time, and

ε = the random error.

A two-factor repeated measures ANOVA was used to examine the effect of the herbicide treatments (factor 1) and fencing (factor 2) on the number of stems of competing woody vegetation in each height class. The fixed effect model has the form:

$$Y_{ijk} = \mu + \alpha_i + \beta_j + \theta_k + (\alpha\beta)_{ij} + (\alpha\theta)_{ik} + (\beta\theta)_{jk} + (\alpha\beta\theta)_{ijk} + \varepsilon_{ijk}$$

where

Y = the number of stems in a given height class,

μ = the overall mean,

α = the effect of the herbicide treatment,

β = the effect of fencing,

θ = the effect of time, and

ε = the random error.

The remaining terms represent the interaction of factors in the full model. The general linear models procedure in SYSTAT 13 (Systat Software Inc., Chicago, IL) was used for all statistical analyses. The Tukey-Kramer HSD mean separation test was used for all multiple comparisons. Treatment effects were considered to be significant when $p < 0.05$. For each analysis, the residuals were tested for normality by using the Shapiro-Wilk test and for homogeneity of variance by using the Levene test.

Table 1.—Average number of stems removed in each treatment and the resulting effects on basal area and photosynthetically active radiation (PAR), Monongahela National Forest

Treatment	Plots no.	Stems removed		Reduction in basal area		Residual stand PAR
		----- number/acre -----		ft ² /acre	percent	percent
		≥1.0 in. d.b.h.	<1.0 in. d.b.h.			
Control	11	—	—	—	—	2.3
Low	11	270	1,154	5	2.7	4.4
Medium	10	400	1,172	23	13.5	8.7
High	10	440	1,139	46	22.8	12.4

RESULTS

A severe wind storm in the 2003-04 dormant season caused several overstory trees to fall over in one section of the study area. As a result, 6 of the original 48 plots were dropped from the study because there was a significant increase in percent PAR in those plots the following growing season. The number of plots retained in the study is presented in Table 1.

The average basal area on all study plots including stems ≥1.0 inch d.b.h. was 185 square feet per acre before treatment. The herbicide treatments reduced stand basal area by 3, 14, and 23 percent in the Low, Medium, and High treatments, respectively (Table 1). In addition, each treatment removed approximately 1,150 stems per acre <1.0 inch d.b.h. The basal area reductions in this study were equal to or less than those recommended for stands of similar site index in the southern Appalachians (Loftis 1990b).

Microsite Light Measurements

Before treatment, the low levels of microsite light beneath the dense subcanopy vegetation were not conducive to oak seedling survival and growth. The average percent PAR was 1.9 percent before treatment, and none of the plots were receiving the minimum amount of light needed for oak seedling survival. When seedlings do not receive enough light, as is common in stands with a dense subcanopy layer, photosynthesis produces less carbohydrates than are used in respiration; thus, the seedlings eventually die (Hodges and Gardiner 1993). Levels of PAR in all plots before treatment were below the threshold level needed for oak seedlings to achieve a positive carbon balance (Hanson et al. 1987). It was clear that low microsite light levels on the forest floor had prevented the development of any large advanced oak seedlings for many years.

One year after the herbicide treatments were applied, there was a significant increase in percent PAR in all treatments ($p < 0.01$). The Low, Medium, and High treatments increased microsite light to 4.4, 7.7, and 12.4 percent PAR, respectively (Fig. 1). The repeated measures ANOVA indicated a significant effect of treatment ($p < 0.01$), time ($p < 0.01$), and the interaction of treatment and time ($p < 0.01$). The differences among the treated plots compared to control plots remained intact throughout the 10-year study period. In the fenced plots, there were significant differences in percent PAR among the treatments for all years except the fourth and sixth years. In those years, percent PAR in the High and Medium treatments appeared to be similar, whereas differences among the other treatments remained intact. In the unfenced plots, there were significant differences in percent PAR

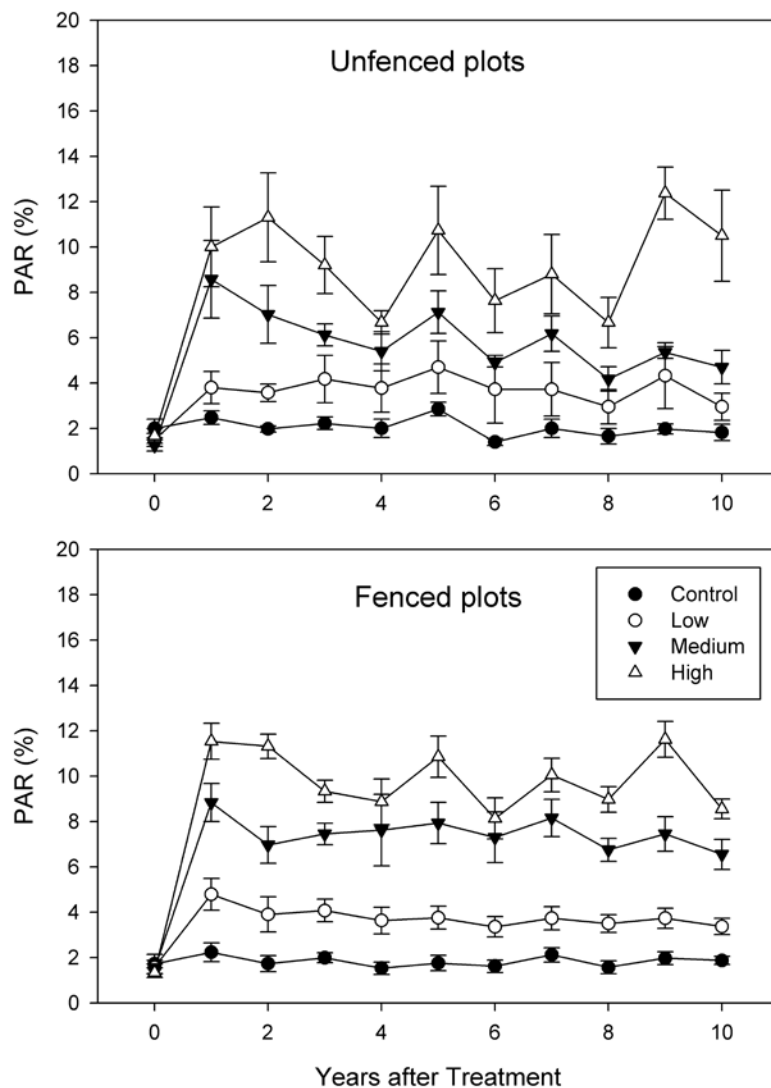


Figure 1.—Average percent photosynthetically active radiation (PAR) before and 10 years after herbicide treatments on Monongahela National Forest sites with one standard error shown above each point.

among the treatments in the 2nd, 3rd, 5th, 7th, and 10th years, again indicating that increases in percent PAR remained intact throughout the study period. There were no apparent differences in percent PAR between fenced and unfenced plots.

Changes in Competing Woody Vegetation

The repeated measures ANOVA indicated a significant effect of treatment ($p < 0.01$), fencing ($p < 0.01$), time ($p < 0.01$), and the interaction of treatment, fencing, and time ($p < 0.01$). As expected, there was a significant difference between the treated and control plots as early as the first year after treatments were applied. Later, the number of stems of competing woody vegetation in the Medium and High treatment plots was significantly greater than that observed in the Control and Low plots. In general, the number of stems of competing woody vegetation increased as both sunlight and time increased. In addition, this relationship was more pronounced in fenced plots, where the effect of deer species preference did not influence the composition of the competing woody vegetation. A brief comparison among the treatment and fencing combinations is presented in the following sections to chronicle the response of the competing woody vegetation over the 10-year study period.

Table 2.—Pretreatment average number of stems per acre <1 inch d.b.h. by species and height class, Monongahela National Forest

Species	Height class					Total
	<6 in.	6 in. to 1 ft	1 ft to 3 ft	3 ft to 5 ft	≥5 ft	
	----- number/acre -----					
Black cherry	18,556	0	0	0	0	18,556
American beech	222	333	444	111	222	1,333
Sweet birch	333	0	0	0	0	333
Red maple	16,889	0	0	0	0	16,889
Red oak	2,889	667	0	0	0	3,556
Striped maple	1,222	556	556	111	333	2,778
Yellow-poplar	0	0	0	0	0	0
Other commercial	222	0	0	0	111	333
Noncommercial	889	0	111	0	0	1,000
Total	41,222	1,556	1,111	222	667	44,778

Pretreatment

Before treatments were applied, American beech and striped maple made up virtually all of the competing woody vegetation ≥1 foot tall in the understory (Table 2). Both species are classified as very shade tolerant in the central Appalachians (Trimble 1973). American beech stems were primarily root sprouts from poletimber and sawtimber parent trees distributed throughout the study area. The striped maple competition was a combination of seedling and sprout origin stems, and all of it was <2 inches d.b.h. It was apparent as the study was installed that the American beech and striped maple competition had developed over several decades as the stand approached maturity. These species had developed into low, dense interference, and it was assumed that numerous cohorts of oak seedlings had come and gone under the dense shade over many years.

Black cherry and red maple seedlings <1 foot tall were common and abundant throughout the study area before treatments were applied. Seed sources for both species were present in the overstory canopy. Although black cherry seedlings are somewhat shade tolerant when they are very small, there was not enough sunlight for them to grow and develop into larger size classes (Marquis et al. 1992). Moreover, black cherry is not a preferred deer browse when other foods are present. Red maple is considered shade tolerant, yet no seedlings ≥6 inches tall were found in the pretreatment inventory of competing species (Table 2). Although red maple was present in the sapling, poletimber, and sawtimber size-classes, low levels of sunlight and deer browsing may have prevented red maple from growing into larger advanced reproduction.

The pretreatment distribution of competing woody vegetation by height class and treatment is presented in Figure 2. Most stems were found in the 1- to 3-foot height class, followed by the ≥5-foot height class. The number of stems in the 3- to 5-foot height class was significantly lower than in the other two height classes before treatment ($p < 0.01$), and the number of stems within each height class was not significantly different by the assigned treatment.

First Year after Treatment

Almost all stems in the height classes ≥3 feet tall and many stems in the 1- to 3-foot height class were removed in the treated plots. By design, oaks were not removed in the treated plots to provide

a future source of reproduction; thus, some plots had a few oak seedlings in the 3- to 5-foot height class 1 year after treatment. At this relatively early stage of development, the number of stems of competing woody vegetation was not significantly different among the treatments. The control plots remained unchanged, and the effect of increased sunlight and protection from deer browsing in the treated plots was not yet apparent (Fig. 3).

Third Year after Treatment

After three growing seasons, there was a significant increase in competing woody vegetation in the 3- to 5-foot height class within the fenced plots ($p < 0.01$) that received the Low and High treatments, and in the unfenced plots that received the Low treatment ($p < 0.01$) (Fig. 4). In the fenced plots, the increases included red maple seedlings responding to greater levels of sunlight and protection from deer browsing. In the unfenced plots, the increases included only striped maple seedlings responding to greater levels of sunlight. It was assumed that deer browsing continued to hold back the development of red maple seedlings for several years in the unfenced plots.

Fifth Year after Treatment

There was a surge in the development of competing woody vegetation in the 1- to 3-foot height class by the fifth year after treatment in all treated plots, particularly those in the fenced plots (Fig. 5). Within the fenced plots, the number of sweet birch and red maple stems in the Medium and High treatment plots was significantly greater than in all other treatments ($p < 0.01$), averaging >10,000 stems per acre in the High fenced plots. Within the unfenced plots, the surge in the number of stems in the 1- to 3-foot height class was primarily red maple. Deer prefer to browse sweet birch over red maple where both species are present (Tilghman 1989); thus, red maple was the primary beneficiary of the increase in sunlight in the unfenced plots. Note that few sweet birch seedlings were present before treatments were applied (Table 2). By the fifth year after treatment, thousands of birch seedlings had emerged from seed and grown into the 1- to 3-foot height class inside the fenced plots.

Seventh Year after Treatment

After the seventh year, the numbers of stems in the 1- to 3-foot and 3- to 5-foot height classes in the Medium fenced and High fenced treatments were significantly greater than those in the Control plots ($p < 0.01$). Sweet birch was the most abundant species in the 1- to 3-foot height class, although red maple and yellow-poplar were also important components. Sweet birch was the dominant species in the 3- to 5-foot height class. In the unfenced plots, the number of seedlings in the Medium and High treatments was significantly greater than in the Low treatment (Fig. 6), with sweet birch as the dominant species. It was also clear that the number of stems in the 3- to 5-foot and ≥ 5 -foot height classes had not recovered to pretreatment conditions after 7 years, indicating that the herbicide treatments provided a somewhat sustained reduction in competition overtopping the desired advanced oak seedlings.

Tenth Year after Treatment

After the 10th year, there were more than 7,000 stems per acre in both the 3- to 5-foot and ≥ 5 -foot height classes in the High fenced treatment (Fig. 7). There was also a slight decrease in the number of stems in the 1- to 3-foot height class, likely due to the effect of overtopping vegetation. The number of stems in the ≥ 5 -foot height class in Medium fenced and High fenced plots exceeded that found in

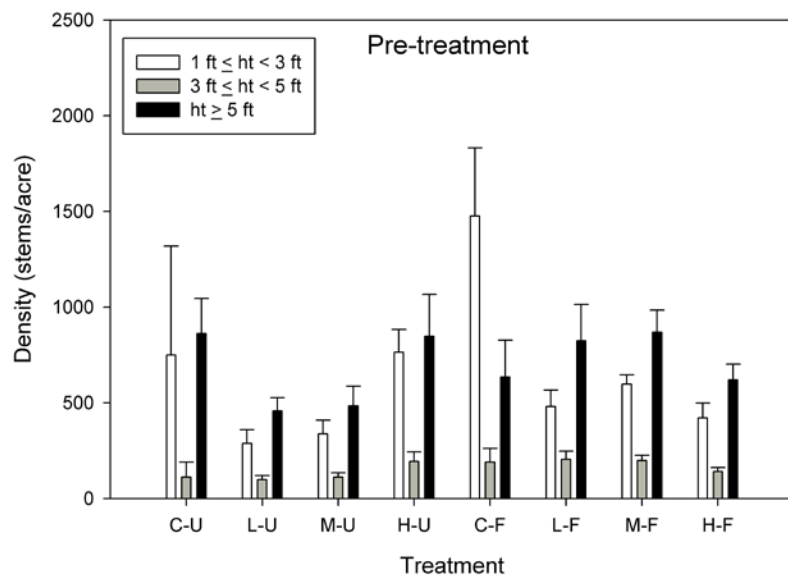


Figure 2.—Pretreatment distribution of competing woody vegetation by height class and treatment, where C = Control, L = Low, M = Medium, H = High, U = Unfenced, and F = Fenced, on Monongahela National Forest sites. One standard error is shown above each bar.

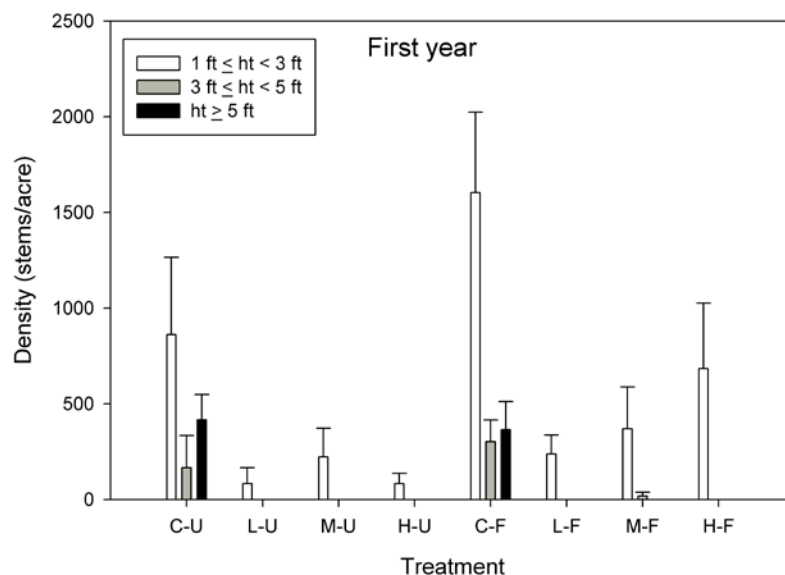


Figure 3.—First-year distribution of competing woody vegetation by height class and treatment, where C = Control, L = Low, M = Medium, H = High, U = Unfenced, and F = Fenced, on Monongahela National Forest sites. One standard error is shown above each bar.

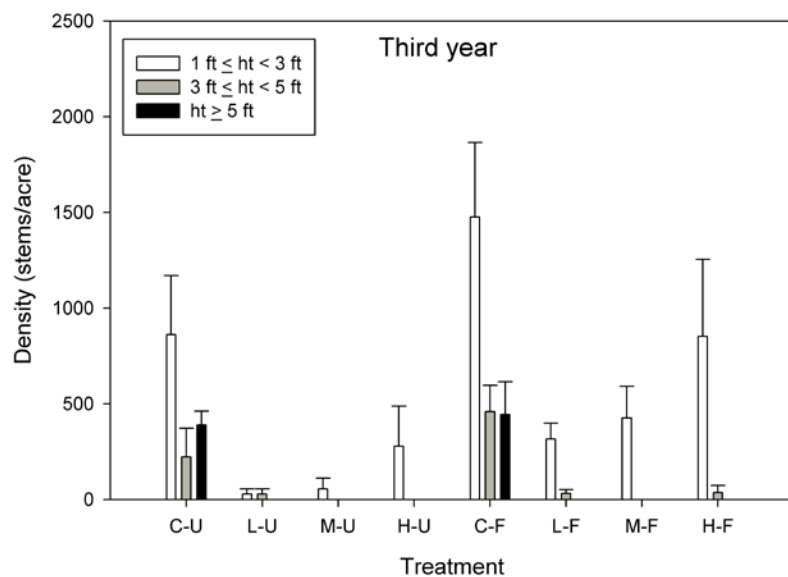


Figure 4.—Third-year distribution of competing woody vegetation by height class and treatment, where C = Control, L = Low, M = Medium, H = High, U = Unfenced, and F = Fenced, on Monongahela National Forest sites. One standard error is shown above each bar.

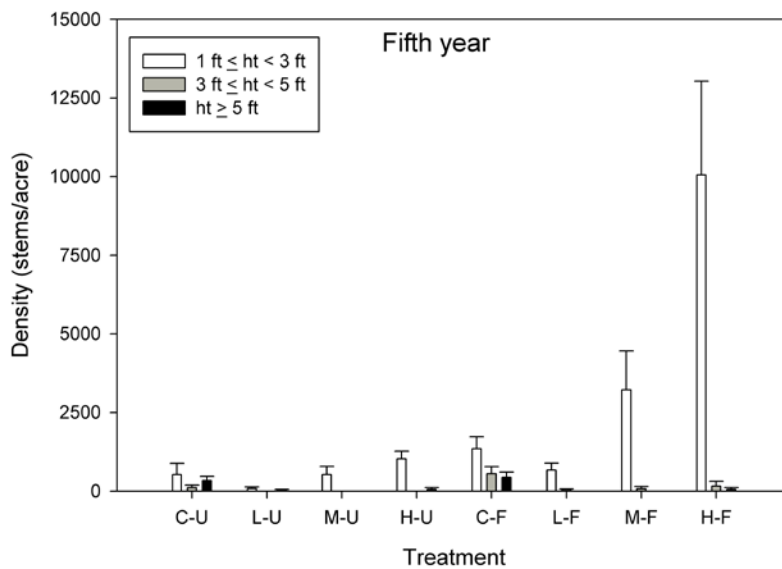


Figure 5.—Fifth-year distribution of competing woody vegetation by height class and treatment, where C = Control, L = Low, M = Medium, H = High, U = Unfenced, and F = Fenced, on Monongahela National Forest sites. One standard error is shown above each bar.

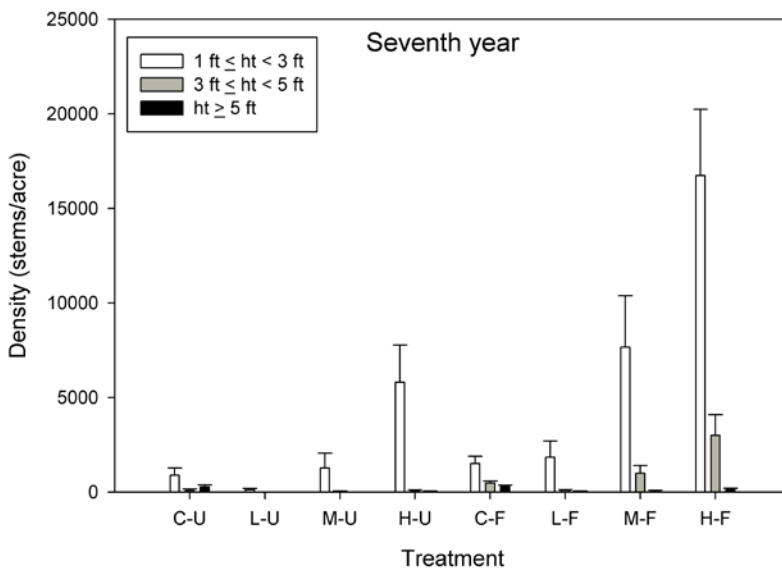


Figure 6.—Seventh-year distribution of competing woody vegetation by height class and treatment, where C = Control, L = Low, M = Medium, H = High, U = Unfenced, and F = Fenced, on Monongahela National Forest sites. One standard error is shown above each bar.

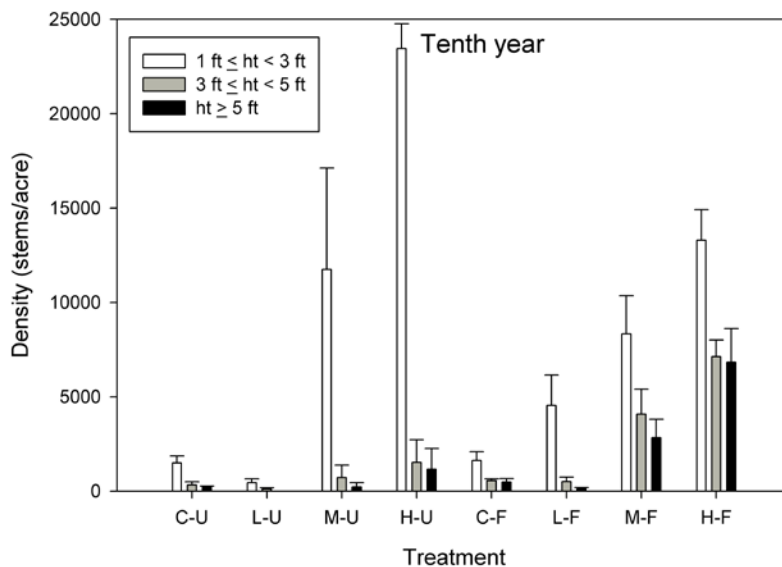


Figure 7.—Tenth-year distribution of competing woody vegetation by height class and treatment, where C = Control, L = Low, M = Medium, H = High, U = Unfenced, and F = Fenced, on Monongahela National Forest sites. One standard error is shown above each bar.

all other treatments ($p < 0.1$). In the fenced plots, the competing woody vegetation was dominated by sweet birch. Black cherry, red maple, and yellow-poplar were also important components. In the unfenced plots, sweet birch was the dominant species in all height classes in the Medium treatment. In the High unfenced treatment, sweet birch was also dominant, with some red maple and black cherry stems present in the 1- to 3-foot height class. By the 10th year, the number of stems in the 3- to 5-foot and ≥ 5 -foot height classes had surpassed pretreatment conditions in the Medium and High treatments in both fenced and unfenced plots ($p < 0.1$).

DISCUSSION

The response of advanced oak seedlings to the shelterwood treatments applied in this study will be analyzed in a separate report. A brief summary is provided here to add context to information presented on the development of competing woody vegetation. Based on more than 2,300 tagged seedlings, the initial average height of advanced oak seedlings was 0.4 foot (Miller et al. 2004). After 10 years, average height increased to 0.5 foot in the Control unfenced plots and 1.5 feet in the High fenced plots. Oak seedling survival ranged from 2 percent in the Control unfenced plots to 46 percent in the High fenced plots. Basal diameter in the Control unfenced and High fenced plots averaged 0.08 inches and 0.24 inches, respectively. In general, dominance probabilities applied to the oak seedlings present after 10 years indicated that both the Medium and High treatments were predicted to yield an acceptable number of codominant oaks in the next stand to meet management objectives.

This analysis focused on the development of competing woody vegetation in height classes ≥ 1 foot tall because regeneration in smaller height classes can be ephemeral, varying with factors such as periodic seed crops, weather conditions, and subtle changes in the availability of alternative deer food from year to year. The three height classes recognized in this study represent increasing degrees of competition that influence the long-term survival of a cohort of advanced oak seedlings. Although much of the competing woody vegetation reached heights that exceeded that of the advanced oak seedlings, previous research on dominance probabilities indicated that oak seedlings of sufficient size will reach codominant status after an overstory harvest (Loftis 1990a). This study showed that competing woody vegetation did not become abundant in the ≥ 5 -foot height class until 7 to 10 years after treatment (Figs. 6 and 7). This result implies that advanced oak seedlings had at least 10 years to develop into larger size classes in preparation for competing in the next stand.

A key decision for land managers is whether to incur the expense of the deer fence in preparation for more successful oak regeneration in the next stand. In this study, fencing produced both positive and negative effects. Oak seedling survival and growth were greater within fenced plots, but the absence of browsing also allowed the rapid development of competing woody vegetation, particularly sweet birch in the Medium and High treatments (Figs. 7 and 8). Advanced reproduction of other desirable species such as black cherry, yellow-poplar, and red maple became established inside the fenced plots; red maple and sweet birch dominated the unfenced plots. Deer impact in the study area was deemed to be moderate based on the observed response of competing woody vegetation and the appearance of certain herb species inside the fenced plots in the latter years of the study (Brose et al. 2008). At higher levels of deer impact, protection from deer browsing may be necessary to promote a competitive cohort of oak seedlings. At lower levels of deer impact, the deer fence may not be needed. The decision to install a deer fence is a function of the desired proportion of oak in the future stand

and the ambient level of deer impact when the oak regeneration process is initiated. The results of this study indicated that a moderate level of oak regeneration success was possible in the unfenced plots that received the Medium and High treatments.

Although percent PAR remained elevated in the 10th year (Fig. 1), it is expected that the sunlight available to desired advanced oak seedlings will diminish rapidly in future years, particularly in the fenced plots that received the High treatment. Measurements were taken at 3 feet above the ground, and there was a surge in competing woody vegetation ≥ 3 feet tall in the latter years of the study. It is expected that a similar response—although delayed several more years—will occur in the Medium-fenced plots and in the unfenced plots that received the Medium and High treatments.

The time interval between applying the treatment and the surge in the development of competing woody vegetation has implications for scheduling the eventual removal of the overstory to initiate the regeneration of a new stand. The results of this study indicated that overstory removal before the seventh year would likely yield unsatisfactory oak regeneration success in the next stand. Within the High treatment plots, percent PAR averaged 12.4 percent (Table 1), which is adequate for survival and growth of oak seedlings. Still, previous research has shown that oak seedlings require at least 6 years to attain a root-shoot ratio equal to 1.0 under a range of sunlight conditions (Brose 2011). Considering the rapid development of competing woody vegetation in the fenced plots between the 7th and 10th years, it would be necessary to schedule the overstory harvest during that period to release the advanced oak seedlings. In the unfenced plots, the overstory harvest could be delayed a few more years. The precise timing of the overstory harvest in both situations would depend on the size and number of advanced oak seedlings present, dominance probabilities associated with those seedlings, and the desired proportion of oak in the next stand (Loftis 1990a).

Periodic control of undesirable vegetation can be a valuable long-term practice in forest management. Zedaker (1986) reasoned that applying herbicide treatments at opportune times in the life cycle of hardwood stands is an effective means of allocating site resources to desirable species. In this case, preharvest herbicide treatments allowed advanced northern oak seedlings to acquire the site resources necessary to become competitive with other species and enhance the probability of successful oak regeneration. Oaks are notorious for slow height growth in the early stages of development (Hodges and Gardiner 1993). Small seedlings need at least 8 to 10 years of desirable growing conditions before overstory removal to develop into competitive advanced seedlings. Preharvest herbicide treatments provide such conditions, in that interfering plants are eliminated quickly and do not become reestablished for many years. Forest managers should consider maintaining relatively low levels of undesirable subcanopy vegetation in hardwood stands, even many years before a planned harvest, to keep interfering species in check and continually allocate resources to preferred species.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

COMPARISON OF OAK AND SUGAR MAPLE DISTRIBUTION AND REGENERATION IN CENTRAL ILLINOIS UPLAND OAK FORESTS

Peter J. Frey and Scott J. Meiners¹

Abstract.—Changes in disturbance frequencies, habitat fragmentation, and other biotic pressures are allowing sugar maple (*Acer saccharum*) to displace oak (*Quercus* spp.) in the upland forest understory. The displacement of oaks by sugar maples represents a major management concern throughout the region. We collected seedling microhabitat data from five upland oak forest sites in central Illinois, each differing in age class or silvicultural treatment to determine whether oaks and maples differed in their microhabitat responses to environmental changes. Maples were overall more prevalent in mesic slope and aspect positions. Oaks were associated with lower stand basal area. Both oaks and maples showed significant habitat partitioning, and environmental relationships were consistent across sites. Results suggest that management intensity for oak in upland forests could be based on landscape position. Maple expansion may be reduced by concentrating mechanical treatments in expected areas of maple colonization, while using prescribed fire throughout stands to promote oak regeneration.

INTRODUCTION

Historically, white oak (*Quercus alba*) dominated much of the midwestern and eastern U.S. hardwood forests (Abrams and Nowacki 1992, Franklin et al. 1993). Oak is classified as an early successional forest species, and many researchers agree that oak populations were maintained by Native American or lightning-initiated fires (Abrams 2003, Abrams and Nowacki 1992, Hutchinson et al. 2008, Moser et al. 2006, Nowacki and Abrams 2008, Ruffner and Groninger 2006, Shumway et al. 2001). These periodic low to moderate surface fires favored the ecophysiological attributes of oak over those of fire-sensitive, shade-tolerant tree species, thereby continually resetting succession and allowing oaks and other shade-intolerant species to persist in both the canopy and understory (Abrams 2003, Abrams and Nowacki 1992, Crow 1988, Franklin et al. 1993, Fralish 2004, Hutchinson et al. 2012). Initially, European settlers adopted the practice of burning hardwood forests (Nowacki and Abrams 2008). While these fires favored oak regeneration, logging by European settlers also removed the existing oak overstory, transforming an uneven-aged oak forest into an even-aged stand dominated by secondary growth white oak (Fralish et al. 1991, 1994; Franklin et al. 1993; Hutchinson et al. 2008).

By the early 20th century, nearly all of the remaining upland oak forest had endured some degree of compositional and structural transformation. Influenced by changes imposed by fire suppression policies and reductions in harvesting operations, understory environmental conditions shifted towards regeneration of shade-tolerant species (Ozier et al. 2006). The nearly continuous forested landscape became a series of smaller fragmented forest stands. After clearing for agriculture and urban expansion, eastern and Midwestern hardwood remnants were heavily dissected by roads, railways, cable cuts, and pedestrian pathways, further decreasing the frequency and magnitude of disturbance in these ecosystems (Nowacki and Abrams 2008).

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The present composition and health of upland oak-hickory forests are the result of biological and structural changes over the last four centuries (Oak 2006, Ruffner and Groninger 2006). Oak regeneration has nearly ceased due to 100 years of disturbance suppression, intensive deer browsing, and competition with native and exotic invasive plant species (Abrams 2003, Bowles et al. 2005, Hutchinson et al. 2008). Maple (*Acer* spp.), ash (*Fraxinus* spp.), elm (*Ulmus* spp.), and other mesic species are placing strong competitive pressures on oak and hickory (*Carya* spp.) seedlings, and often have more success reaching sapling size (Fralish 2004). Even in oak-dominated stands where there are a substantial number of oak seedlings and no regeneration of late successional species, the current overstory and understory environments are still preventing oak survival into the next size class (Abrams and Nowacki 1992, Bowles et al. 2005, Haas and Heske 2005).

Historically restricted to mesic, nutrient-rich locations, late successional species are increasingly regenerating in the upland dry-mesic and xeric nutrient-poor sites traditionally dominated by white oak (Abrams 2003, Fan et al. 2012). Sugar maple (*A. saccharum*) and other fire-intolerant species have reached tree size in mesic stands and serve as a major seed source for recruitment into adjacent dry-mesic and xeric sites (Fralish 2004, Franklin et al. 2003). Understory shading caused by canopy closure, increased plant density in the subcanopy and understory, and the accumulation of a denser litter layer, may form moist, nutrient-rich microsites able to support sugar maple and similar species in formerly xeric or dry-mesic stands (Collins and Good 1987, Crow 1988, Franklin et al. 1993).

Forest managers fear that without silvicultural treatments, the white oak component of these forests will be replaced by sugar maple and other shade-tolerant species (Franklin et al. 1997, Moser et al. 2006). Some studies predict white oak may nearly disappear in many of the remaining upland forest tracts within the next 50 to 100 years (Fralish 2004, Franklin et al. 1993, Haas and Heske 2005). However, there is no clear consensus on the best management practices to use. Selecting one method or set of management practices to rescue the oak component in all upland forests may not be possible. It may be more practical to identify a set of environmental indicators that could be used to quantify the current health of the oak component in a stand. Linking an individual silvicultural practice or several with preidentified ranges of tolerance for these environmental indicators, could allow forest managers to more effectively choose an appropriate course of action to counter sugar maple expansion while restoring oak regeneration.

To identify how silvicultural treatments alter seedling-environment links, we surveyed five forests with different site histories and documented the microhabitat characteristics of individual seedlings. These data were used to determine (1) whether species of tree seedlings are selectively recruiting into specific microhabitats (habitat partitioning), (2) whether silvicultural treatment results in changes in microhabitat at the stand scale, and (3) whether silvicultural treatments alter seedling-environment relationships. The goal of this work was to determine the best stand locations for silvicultural treatment that will alter seedling-environment links towards those that favor oak regeneration and minimize maple regeneration.

METHODS

Field Sampling

From May to August 2007, study sites were established in forest stands at the U.S. Army Corps of Engineers' Lake Shelbyville Recreation Area in east central Illinois (39°32'17" N, 88°42'36" W). Data were collected from five upland oak forest sites differing in age class (old-growth vs. secondary growth) or management application (burned, thinned, or no treatment) (Table 1). Stands with trees 100 to 200 years old and free of signs of disturbance (e.g., tree stumps, canopy gaps, fire scars) were considered old-growth, and those with canopy trees 40 to 100 years old with evidence of disturbance were designated secondary-growth (Fralish et al. 1991). Forest overstory and understory measurements were taken at random points established along transects in each stand. Data were collected at least 30 m from a forest edge to minimize edge effects (Abrams and Nowacki 1992, Crow 1988, Parrott et al. 2012). A 10-factor prism gage was used at each random point to estimate the basal area (BA) per acre of standing trees in each stand and then converted to BA per ha (Table 1). Diameter at breast height (d.b.h.) was measured and species identified for each tallied tree. Understory data were collected in 50-m² circular plots and sapling data in 200-m² circular plots fixed with the random point at center.

Tree seedling species densities were measured by using the point-center quarter method at each random point (Abrams and Nowacki 1992, Collins and Good 1987, Hartman and McCarthy 2007). All individual seedlings <100 cm tall (15,592 stems) were identified to species, except for ash and hickory, which were identified only to genus. Red oak (*Q. rubra*) and black oak (*Q. velutina*) were difficult to distinguish when small, so these were grouped together. The presence, size, and position of maple and oak saplings taller than 100 cm and <6 cm d.b.h. were also recorded for each quadrant at the random points. Measurements were taken with standard forestry tools for percent canopy cover, litter depth (average of two ruler measurements), soil moisture, slope steepness, slope aspect, slope position, herbaceous cover, and shrub cover present at each random point and at the location of the nearest oak and maple seedlings (within 4 m in each quadrant). Point positions were recorded by using a hand-held global positioning system, and the presence of the exotic shrubs autumn olive (*Elaeagnus umbellata*) and honeysuckle (*Lonicera maackii*) was also noted.

Data Analysis

Standing tree data were used to calculate BA per ha for each species by random point and stand. Seedling BA and density per ha were determined for each stand and then used to calculate the relative density of each species in the understory. Slope position data recorded for each random point and

Table 1.—Description and history of the upland oak sites sampled at the Lake Shelbyville Recreation Area in Illinois, 2007

Stand	Area (ha)	Age class	Prescribed burn	Thinning	BA per ha
1	21.37	Secondary growth	No	Yes	23.9
2	16.80	Old-growth	Yes	No	24.4
3	20.26	Secondary growth	Yes	No	22.8
4	5.60	Secondary growth	Yes	No	27.9
5	17.99	Secondary growth	No	No	31.8

the associated closest oak and maple seedlings were separated into four categories based on landscape location: plateau, high, mid, and low. Slope aspects were grouped into three categories based on expected solar inputs: mesic, xeric, and plateau. We designated north, east, northeast, and northwest aspects as mesic, and south-, west-, southwest-, and southeast-facing positions as xeric at random points and at sampled oak and maple seedlings. Locations where no dominant aspect was observed were designated as plateau, the flat hilltop portion in each stand.

Statistical differences between tree distribution patterns and the effects of landscape position in the forest canopy and understory were analyzed by using a multivariate analysis of variance (MANOVA; SAS Version 9.1.3, SAS Institute Inc., Cary, NC) using the Wilk's Lambda test statistic. To test our hypothesis of habitat partitioning, the relative abundance of maples, white oak, and all oak species was related to slope aspect and position for both the understory and canopy layers. Relative densities of oak and sugar maple seedlings were compared among stands and the relative density of oak seedlings was correlated with BA per ha of overstory trees in each plot. A MANOVA was used to determine whether oak and sugar maple seedlings exhibited environmental selectivity. Environmental conditions recorded for each oak and maple seedling were compared to the measurements taken at each random point. A principal component analysis (PCA) ordination of the same environmental variables was used to visualize differences among sample points and stands in the MANOVA.

RESULTS

According to upland hardwood forest stocking guides by Roach and Gingrich, the five white oak-dominated stands sampled were fully stocked to nearly overstocked (U.S. Forest Service 2001). As would be expected, the stand that did not receive a thinning or prescribed burning treatment had the largest BA per ha (Table 1). Three of the four sampled stands receiving either a thinning or prescribed burn had noticeably smaller BA per ha (Table 1).

Slope position had a significant effect on tree distribution in both the understory and canopy layers in the sampled sites (Table 2). Although slope aspect did not have a significant effect on tree distribution in either the understory or canopy, the interactive effects of slope position and aspect did have a significant effect in the understory, showing finer environmental selectivity in the seedling stage (Table 2). Patterns of relative abundance in the canopy followed trends commonly associated with upland hardwood forests. Sugar maple was more abundant on lower slope positions and mesic

Table 2.—Effects of landscape position on tree distribution in the understory and canopy forest layers on sites at Lake Shelbyville Recreation Area, Illinois, with significant P-values in bold (results from a MANOVA test using Wilks' Lambda statistic)

Effect	F-Value	P > F
<i>Understory</i>		
Slope position	4.26	<0.0001
Slope aspect	1.65	0.1318
Slope position × aspect	2.05	0.0189
<i>Canopy</i>		
Slope position	2.46	0.0097
Slope aspect	1.29	0.2549
Slope position × aspect	0.48	0.9289

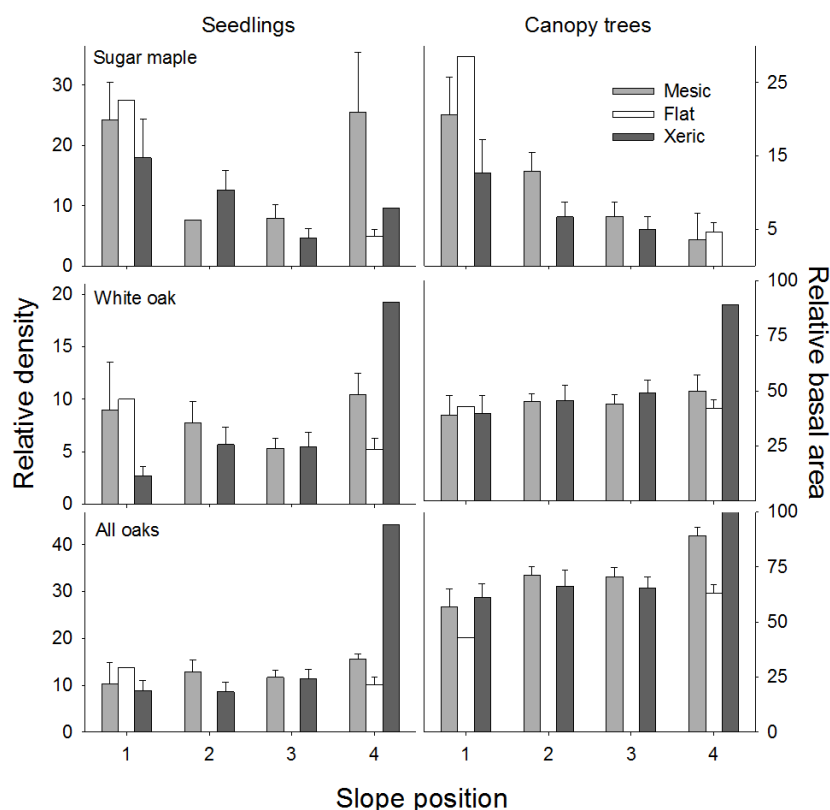


Figure 1.—Effect of slope position and aspect on the distribution of oak and maple in the forest understory and canopy. Slope position – 1: low, 2: mid, 3: high, 4: plateau.

aspects (Fig. 1). In contrast, the distribution of white oak and all oak species combined was fairly constant regardless of slope position or aspect, with only a slight increase in abundance on higher slope positions.

Patterns of overstory tree distribution differed from those in the midstory and understory.

Occurrences of sugar maple saplings were higher than all oak species in each stand, suggesting that maple seedling survival rates into the next age class were much higher in our study areas. The number of sugar maple saplings exceeded that of combined oak species by ≥ 81 percent in stands 1, 3, and 5 whereas sugar maple outnumbered oaks in stand 2 by only 35 percent and in stand 4 by only 12 percent.

We observed 332 sugar maple seedlings and 276 white oak seedlings across the five studied stands, with sugar maple and white oak relative densities reaching 8.65 and 5.97 individuals per 50 m², respectively. The observation of sugar maple and white oak in the understory was nearly equal in stands 1, 2, and 4. However, we recorded the presence of about twice as many sugar maples as white oaks in stands 3 and 5. We expected to find more sugar maple in the understory of stand 5 because it did not receive either a thinning or prescribed burning treatment. However, in stand 3 we observed only six sugar maples in the overstory, which were 76 to 89 percent fewer observations than in the other four stands. Sugar maple seedling densities were highest in the lower slope positions regardless of slope aspect, but also showed a remarkable increase on mesic aspects in the plateau portions of the sampled stands (Fig. 1). White oak and all oak seedlings combined were fairly uniform across all slope positions and aspects in the understory, similar to the overstory pattern. In contrast with sugar maple, the highest densities of oak seedlings were observed on the xeric aspects of the plateau in each stand (Fig. 1).

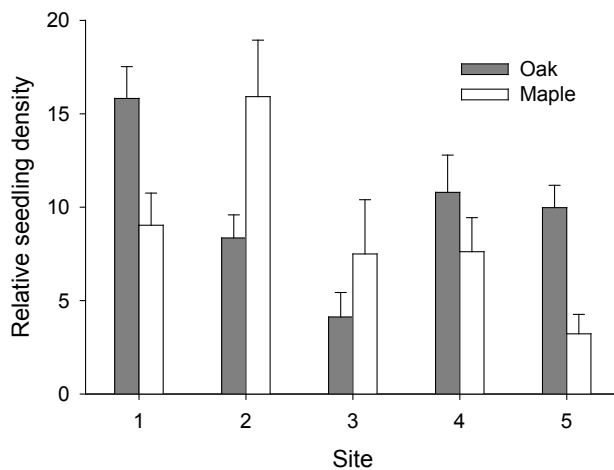


Figure 2.—Oak and maple seedling densities across study sites. 1: thinning, 2-4: prescribed burn, 5: no treatment.

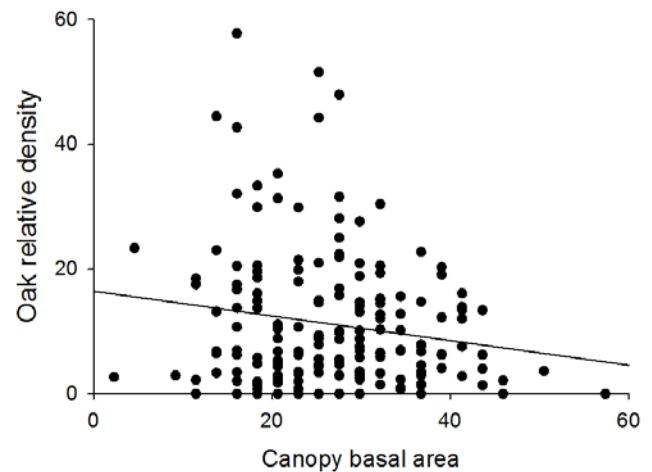


Figure 3.—Relative density of oak seedlings as a function of basal area of overstory trees.

There were marked differences in oak and maple seedling densities across forest stands. Oak relative densities were the highest in stand 1, where a thinning treatment occurred about 13 years before our study (Fig. 2). Oak seedling densities also remained higher than sugar maple in stands 4 and 5. Even though stands 2 and 3 received a prescribed burn, sugar maple seedling densities were nearly double those of oak seedlings in stand 3 (Fig. 2). Ten of the 12 sampling locations in stand 3 were located on either high or plateau slope locations, suggesting that this area was susceptible to adjacent sources of wind-blown maple seeds. We expected oak seedling densities to increase as the BA per ha of overstory trees decreased (Fig. 3). There was overall a significant negative correlation ($R = -0.16$; $P = 0.028$) between canopy BA and oak seedling relative density. When we analyzed the effect separately for each stand, however, there was a significant correlation only in stand 4.

The results of the PCA showed that microclimatic conditions differed appreciably among stands. The PCA of seedling and random microhabitats resulted in two informative axes (eigenvalues >1) and explained 25.0 and 17.7 percent of the variation in the data set, respectively. However, oak and maple seedlings were consistently separated from each other and from random data points in each stand (Fig. 4, Table 3). Contrasts in the MANOVA revealed significance in random versus *Acer* points ($P < 0.0001$) and *Acer* versus *Quercus* points ($P < 0.0001$) along the first PCA axis. Sugar maple and oak did not differ from each other on the second PCA axis ($P = 0.9792$), but both differed from random points ($P = 0.0099$ and $P = 0.0075$, respectively). The stand $\times$ species interaction was not significant, suggesting the direction

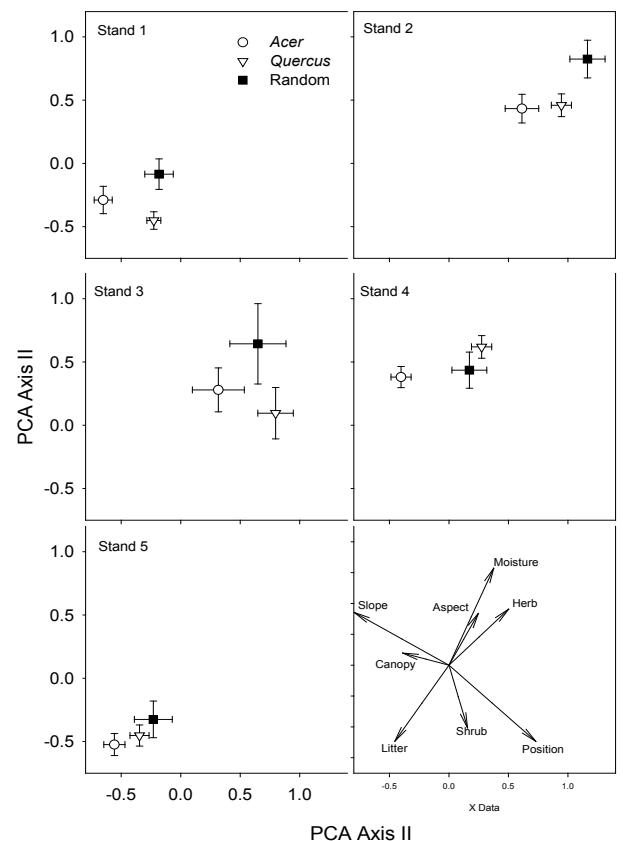


Figure 4.—Environmental selectivity of oak and maple seedlings in relation to random points. Data come from a PCA ordination of environmental variables. Lower right panel portrays the loading of each environmental variable on the PCA.

Table 3.—Habitat specificity in maple and oak seedlings sampled on sites at Lake Shelbyville Recreation Area, Illinois, relative to random points, with significant P-values in bold (results from a MANOVA test using Wilks' Lambda statistic)^a

Effect	F-Value	P > F
Stand	61.95	<0.0001
Species	14.42	<0.0001
Stand × species	1.46	0.1058

^a Visualization of these data is presented in Figure 4.

of habitat selection was consistent. Environmental shifts from random points were similar in stands 1 and 5, but with slightly greater effects from canopy and shrub coverage in stand 1. In stands 2 and 3, maple appeared influenced most by aspect and moisture, whereas oak displayed additional effects from slope position. Oak and maple separated along environmental gradients most notably in stand 4. As in stands 2 and 3, aspect and moisture appeared the most influential on oak site selection. In contrast, maple appeared most affected by slope and canopy cover conditions.

DISCUSSION

Although BA per ha was noticeably smaller in three of the stands receiving silvicultural treatments, the fact that they remained at fully stocked levels suggests they did not have the canopy openness needed to promote competitive oak regeneration (Hutchinson et al. 2008, Moser et al. 2006). The higher BA per ha in stand 4 could be explained by the stand's size (5.6 ha), shape (rectangular), and surroundings (row crop agriculture fields). This stand configuration could promote colonization from wind-dispersed seeds and intensify seed predation rates on acorns (Haas and Heske 2005, Hutchinson et al. 2008), even though the narrowness of the tract would have increased light penetration into the forest interior (Shotola et al. 1992).

The responses of maples and oaks to slope position and aspect were comparable to other studies (Abrams 2003, Fralish 1994, Fralish et al. 1991, Ozier et al. 2006) and may explain differences among stands in relative abundance. The higher numbers of sugar maple in stand 3 could be related to the abundance of favorable mesic conditions facilitated by dense canopy layers in the overstory and midstory. Growth and survival patterns of saplings in the midstory have been documented to cause shifts in species composition of tree seedlings during succession (Delucia et al. 1998, Fan et al. 2012). Dominance of maple in the subcanopy restricts solar inputs to the understory, causing an increase in competition intensity that favors shade-tolerant species (Bowles et al. 2005, Franklin et al. 1993). This effect appears prominent as sugar maple slowly colonizes upslope and across moisture gradients into areas traditionally less suitable for this species. Light restrictions imposed by adjacent maple canopy and subcanopy trees, along with a denser maple leaf litter resistant to disturbance (e.g., fire, insect herbivory), facilitate an increase in mesic germination sites that favors further maple seedling establishment (Franklin et al. 2003). Maple expansion patterns across both moisture and slope gradients increase direct competition with oak, reducing the ability of oak seedlings to reach sapling size (Adams and Anderson 1980, Ozier et al. 2006).

Successful oak regeneration has been linked to forest disturbance and lower stocking rates (Abrams 2003, Bowles et al. 2005, Crow 1988), so we assumed oak seedling densities would be highest in

stands with disturbance and smaller BA per ha of canopy trees. Although this pattern remained true in two of the four stands receiving a silvicultural treatment, the lack of oak recruitment into the sapling class across all sites suggests that a future species shift in overstory dominance will occur. Numerous studies suggest stand thinning by individual tree selection, as opposed to group selection, provides maple with a lifelong advantage over seedlings and saplings of oak, which needs direct sunlight throughout its life (Fralish 2004, Nowacki and Abrams 2008, Ozier et al. 2006). In addition, the positive effects of fire without canopy disturbance on oak regeneration may be short lived and not sufficient to disrupt the successional trends toward shade-tolerant species (Abrams 2003, Franklin et al. 2003, Nuttle et al. 2013).

Microsite habitat segregation between oaks and maples at the stand level was consistent with other studies that have documented similar shifts along environmental gradients between tree species (Collins and Good 1987). Although oak is tolerant of a wide range of environmental conditions, limited mostly by light levels and disturbance, maple requires mesic habitats driven by an intact canopy and subcanopy with a denser developed understory (Horsley et al. 2002, Nigh et al. 1985). Most importantly, the environmental shifts generated by maple dominance in the canopy will specifically shift the understory environment towards microenvironments that favor maple seedlings. In these forest stands, oak may be simply surviving in the microsites available as sugar maple alters environmental conditions; these changes reduce disturbance and facilitate moister, cooler microsites—further reducing the ability of oak to recruit into the next size class (Nowacki and Abrams 2008). In addition, the consistency of environmental preferences among stands suggests that silvicultural treatments have not altered the underlying physiological constraints on tree regeneration.

Seedling responses in this system suggest that there are different physiological mechanisms involved in regulating oak and maple regeneration. Maple species appear to be selectively recruiting into specific microhabitats that are subsequently altered towards environmental conditions that further promote maple regeneration. In contrast, oak regeneration seems to be more responsive to changes in microhabitat imposed by silvicultural disturbances. Silvicultural treatments can alter understory environmental conditions, and can provide increased opportunities for oak regeneration. However, results of this study indicate that single thinning or fire prescriptions are insufficient to restrict maple to preferred environments. Several studies suggest using prescribed burns before and after group selection thinning, which would promote oak seedling root development while culling current sugar maple regeneration (Crow 1988, Delucia et al. 1998, Franklin et al. 1993, Haas and Heske 2005, Hutchinson et al. 2008, Moser et al. 2006, Ruffner and Groninger 2006). The effects of slope and aspect on patterns of maple and oak regeneration in our study support the argument that land managers could use landscape position to pinpoint silvicultural treatments at the stand level. Because maples and other shade-tolerant species have shown the ability to stump sprout (Hutchinson et al. 2008, Tift and Fajvan 1999) and achieve sapling sizes resistant to fire, maples in the subcanopy should be both mechanically and chemically culled (Hutchinson et al. 2012). Maple expansion may be reduced by concentrating mechanical treatments in expected areas of maple colonization, while continually using fire throughout stands to promote oak regeneration. By effectively targeting management efforts, land managers may focus on those sites most likely to result in increased oak regeneration.

ACKNOWLEDGMENTS

This study was supported by a Research and Creative Activity Award from the Graduate School of Eastern Illinois University. We thank Lexi Phillips, Zachary Beck, and Patrick Schreiber for providing technical assistance throughout the field sampling portion of this study. Special thanks go to Lee R. Mitchell and the U.S. Army Corps of Engineers for allowing us access to sites surrounding Lake Shelbyville.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

FOREST REGENERATION

ESTABLISHING PERENNIAL SEED-BASED ENERGY CROPS ON RECLAIMED SURFACE MINE SOILS IN THE CENTRAL APPALACHIANS

Jamie L. Schuler, Shawn Grushecky, and Jingxin Wang¹

Abstract.—Renewable energy has been at the forefront of the United States' energy policies. Cellulosic feedstocks have received considerable interest in the Appalachian region because of their abundance and availability, but cost competition from other energy sectors has limited their use in the region. Some other bioenergy feedstocks, such as corn and soybeans, are not a viable alternative for most of the region. Though not considered suitable for traditional agricultural crops, a large portion of disturbed mine land in West Virginia, Pennsylvania, and eastern Ohio has been reclaimed and planted with perennial grasses. To find an alternative to the more traditional feedstocks, we are exploring the use of perennial seed-based energy crops harvested from trees established on these reclaimed lands. Dunstan hybrid chestnuts (*Castanea dentata* × *Castanea mollissima*) and hybrid hazelnuts (*Corylus* sp.) were planted on a reclaimed surface mine in north-central West Virginia. Individual seedlings were planted with or without composted manure. Soil from the same site was used to study the effects of various combinations of poultry-based biochar, wood-based biochar, and two water sources (rainwater and mine drainage) on the survival and growth of Dunstan chestnut, hybrid hazelnut, and Allegheny chinkapin (*Castanea pumila* var. *pumila*) in a greenhouse. We will describe the first-year results of field and greenhouse tests of Dunstan chestnut and Allegheny chinkapin; hybrid hazelnut survival was so low that this species was not further analyzed. Growth and development of chestnut and chinkapin seedlings in the field and greenhouse during the first growing season did not benefit from amendments. Creating a sustainable bioenergy industry based on perennial seed-based crops in the central Appalachian region will depend on the rapid establishment and growth of tree crops over large areas. Results from these studies will help to inform decisions about establishing and maintaining these crops.

INTRODUCTION

Energy production has been a major driver for the central Appalachian region for more than a century. Coal and gas/oil extraction have traditionally supported the largely rural economies and community social programs. Although coal remains a locally abundant and low-cost fuel, its high carbon emissions coupled with the low price of natural gas have reduced its use for electric power generation. With reduced coal output and mounting public concern over the environmental impacts of nonrenewable energy development including recent shale gas extraction, new opportunities are needed for the region. It is expected that renewable energy will fill the energy void being left by fossil fuels. More than 200 large-scale ethanol and 100 biodiesel operations are currently converting corn, soybeans, and other crops into liquid fuels. To date new opportunities for the development and adaptation of sustainable energy technologies have gone unrealized in central Appalachia, in part due to the lack of bioenergy production facilities and feedstocks appropriate for the region.

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Corn and soybeans are currently the seed crops most often utilized for biofuel production in the United States. The central Appalachian region, however, is constrained in agricultural production because it lacks suitable site conditions. Despite having one of the most productive growing climates in the eastern United States, with moderate temperatures and abundant rainfall (exceeding 200 cm/yr in areas), the region is limited by terrain and soil conditions. Its sloped and rocky soils are less compatible with intensive and mechanized farming practices associated with modern annual crop production systems.

One new potential pathway for creating a sustainable bioenergy industry for the region involves the use of perennial seed energy crops harvested from tree plantations. Currently some tree species are grown specifically for their seeds. Pecans (*Carya illinoensis*), walnuts (*Juglans* spp.), hazelnuts (*Corylus* spp.), and pistachios (*Pistacia* spp.) are some of the more commonly grown trees for “nut” production in the United States. Select perennial oilseed crops have high oil contents ranging from 560 to 1,400 L/ha (Molnar 2012), $\geq 5,600$ L/ha for Chinese tallow (*Triadica sebifera*) (Breitenbeck 2008), and 2,150 L/ha (Hill et al. 2010) for sweet pecan. In contrast, soybeans yield approximately 517 L/ha (Hill et al. 2010). Similarly, some species have seeds low in oil but high in carbohydrates, such as chestnuts (*Castanea dentata*), which can be processed by using technologies similar to those used by corn-to-ethanol platforms.

Perennial seed energy crops are desirable in central Appalachia as in other regions for many reasons: one-time establishment cost, reduced site disturbance, fewer cultural inputs (e.g., fertilizers, irrigation, pesticides), and reduced energy inputs. Although the Appalachian region lacks agricultural land capable of supporting significant production of annual seed energy crops, it has a tremendous amount of idle land that could potentially be converted into production areas for other energy feedstocks, such as perennial seed crops. There are an estimated 0.5 million ha of surface mine sites in central Appalachia, and an additional 3.6 million ha of marginal crop land potentially available. Some of the disturbed mine land in West Virginia, Pennsylvania, and eastern Ohio has been reclaimed. Much of the reclaimed mine lands have been “restored” by grading the blasted overburden material, replacing native topsoil salvaged from the site, and seeding aggressive perennial cool-season grasses for erosion control.

The extreme disturbance to the normal structure, chemistry, and biology of these soils makes them challenging as substrates to support productive crop growth. They are characterized by low or high soil pH, high salinity, high heavy metal content, variable drainage, high rock fragment content, and absence of typical soil microbes. Therefore, some recent effort has looked into amending soil properties to improve conditions to better support tree growth. For example, waste products such as fly ash, animal manures, and sewage have been added to the surface to improve the soil properties (Skousen et al. 2013). Additionally, biochar, a carbon-rich substance produced when organic matter is combusted under low oxygen, is gaining interest as a potential soil amendment, specifically because of its potential to increase crop productivity and improve soil physical properties and long-term carbon storage (Lehmann and Joseph 2009).

As an alternative to the more traditional energy feedstocks, perennial seed-based energy crops harvested from trees established on these reclaimed sites are being explored to create a sustainable bioenergy industry for the Appalachian region. The objectives of our study are to examine growth

rates and establishment practices of perennial seed crops on one of the many reclaimed mine sites in Appalachia. One field study was initiated to compare seedling establishment on a former surface mine site using Dunstan chestnut (*Castanea dentata* × *Castanea mollissima*) and hybrid hazelnut with and without soil amendment. A companion study was initiated to further examine opportunities to improve growing conditions for several perennial seed crop species using wood- and poultry-based biochar in a greenhouse trial.

METHODS

Field Study

A 2-ha portion of a former surface mine in north-central West Virginia was used for this study. Twelve single-species blocks containing 20 trees each were planted in April 2013 using two species and two soil amendment treatments. Hybrid hazelnut and Dunstan chestnut were each hand planted into augered holes on six planting blocks. Three blocks per species were amended with composted cow manure. Cow manure (7,275 cm³) was incorporated into each hole during augering. The hazelnuts were planted at 3.0 m by 3.0 m spacing; the chestnuts were spaced at 6.1 m by 6.1 m.

Immediately after planting, 1.2-m-tall tree shelters (Jump Start® “Full Sun,” Plantra, Inc., Eagan, MN) were installed around each seedling to reduce deer browse. Directed herbicide applications (2 percent glyphosate solution) were also performed around each seedling in the spring and midsummer during the first growing season to control competition.

Initial basal diameter and total height were measured 2 weeks after planting, which was before budbreak. Foliage samples were collected from five seedlings per plot in August 2013. End-of-year survival, basal diameter, and total height were determined in September 2013.

Greenhouse Study

The main experiment evaluated growth and survival of Allegheny chinkapin (*Castanea pumila* var. *pumila*) grown in mine soil subjected to two amendment levels and two types of irrigation:

1. No amendment + rainwater irrigation
2. Poultry-based biochar (mixed 2.5 percent volume/volume) + rainwater irrigation
3. No amendment + mine drainage water
4. Poultry-based biochar + mine drainage water

Each treatment was repeated six times and randomly assigned a position on a greenhouse bench. Pots were periodically rearranged to ensure similar growing conditions.

The irrigation treatments were initiated because of the availability of mine drainage water for irrigation at the mine site used in the field study. Rainwater was collected as runoff from a plastic trap and stored in a plastic barrel. Mine water was collected at the reclamation site directly from the pipe before it was applied to the surface outside the area of the field study. The mine water was recirculated daily over limestone to increase the pH of the water. Rain water pH averaged 5.85 and ranged from 5.64 to 6.07. Mine water pH averaged 7.59 and ranged from 7.25 to 8.25.

Table 1.—Chemical analysis[†] of biochar samples used in greenhouse study, West Virginia

Sample	N	P ₂ O ₅	K ₂ O	S	B	Zn	Mn	Fe	Cu	Ca	Mg	Na	Al
	<i>percent</i>												
Poultry (n=2)	3.33	9.61	6.245	0.99	0.01	0.075	0.095	0.53	0.215	6.415	2.05	2.13	0.615
Wood (n=1)	0.79	0.08	0.23	0.01	0.001	0.002	0.01	0.02	0.01	0.37	0.03	0.02	0.01

[†] N: nitrogen; P₂O₅: phosphate; K₂O: potash; Na: sodium; Al: aluminum.

The greenhouse study used surface soil from the same area used for the field study. Three mine soil samples were collected and analyzed by an independent lab (Waters Agricultural Laboratories, Inc., Camilla, GA). The average results were 6.7, 250, 722, 6674, 162, 1.39, 9.74, 310, 408, 7.3, and 2.02 kg/ha for phosphorus (P), potassium (K), magnesium (Mg), calcium (Ca), sulfur (S), boron (B), zinc (Zn), manganese (Mn), iron (Fe), and copper (Cu), respectively. Soil pH was 7.8 and cation exchange capacity was 18.7 meq/100 g. A chemical analysis of the poultry-based biochar samples used in the greenhouse study is given in Table 1.

Dunstan chestnut and hybrid hazelnuts were included as a comparison to the chinkapin. However, only the untreated soil versus poultry-based biochar treatments were included as treatments. Each combination of species and amendment was repeated five times.

A final set of treatments was included to explore the response of Allegheny chinkapin to a different type of biochar. Seedlings growing on unamended mine soil were compared to ones growing with soil amended with wood-based biochar (2.5 percent volume/volume, Table 1). These seedlings were not directly compared to those grown as part of the main greenhouse study because seedling quality differed between the experiments.

All pots were well watered with rain water (except those designated to receive mine drainage water) 3 days per week beginning at establishment in April through September 2013. Foliage samples were randomly collected from each seedling in August and composited by treatment for analysis (no statistical comparison was performed). Height and basal diameter were measured for each seedling just before harvest in September after one growing season. Following harvest, individual plants were separated into their biomass components (root, stem plus branches, and leaves), dried to a constant temperature at 65 °C, and weighed.

DATA ANALYSIS

All data were evaluated using SAS Proc GLM (SAS Institute Inc., Cary, NC) with a P = 0.10 level of significance. Hybrid hazelnut survival was poor in both experiments (<10 percent for the field study and 0 percent for the greenhouse study) and was not analyzed further.

RESULTS

Field Study

First-year survival of Dunstan chestnut was 78 and 90 percent for manure and control seedlings, respectively. Manure-treated seedlings were shorter and had smaller diameters than control seedlings, although differences were not significant (Table 2). By the end of the growing season, control seedlings increased in height approximately 20 cm compared to only 10 cm for treated seedlings. Similarly, control seedlings grew in diameter 1.3 mm on average compared to 0.8 mm for treated seedlings. Foliage testing revealed manure-treated seedlings had significantly higher nutrient concentrations for nitrogen (N), P, and Mg, whereas control seedlings had higher foliar Ca concentrations (Table 3).

Greenhouse Studies

Greenhouse experiment 1 sought to examine the effects of two types of irrigation water and biochar on Allegheny chinkapin growth and survival. The addition of 2.5-percent biochar was detrimental to both growth and survival (Table 4). The addition of biochar significantly increased mortality in chinkapin, regardless of irrigation type (main effect, $P = 0.0001$). Although individual seedling biomass was reduced, a significant difference between rainwater + biochar and rainwater-treated seedlings was limited to the leaf tissue, which was reduced by more than 50 percent. Mine water and rainwater treatments were not significantly different. Mean biomass of the three components differed by 1 to 12 percent between the two types of irrigation, and total biomass differed by 8.5 percent.

Chemical concentrations in the foliage based on one composite sample for each treatment are listed in Table 5. The poultry-based biochar treatment resulted in higher elemental concentrations in chinkapin compared to nonamended seedlings in greenhouse study 1. Compared to treatment with rainwater, seedlings irrigated with mine water had lower elemental concentrations for all elements except for Ca, S, and Al.

Table 2.— First-year mean growth and survival (+ standard error) of Dunstan chestnut planted on a reclaimed mine site in West Virginia

Treatment	Height (cm)		Groundline diameter (mm)		Survival (%)
	Year 0	Year 1	Year 0	Year 1	
Control	81.3 (1.2)	98.3 (3.0)	8.1 (0.15)	9.4 (0.19)	90 (0.11)
Manure	82.2 (1.2)	91.1 (3.2)	7.9 (0.15)	8.7 (0.20)	78.3 (0.11)
P-value	0.59	0.65	0.09	0.26	0.46

Table 3.—Foliar concentrations of various elements for Dunstan chestnut growing on a reclaimed mine site in West Virginia

Treatment	N	P	K	Mg	Ca	S	B	Zn	Mn	Fe	Cu	Al
	-----%						-----mg/kg-----					
Control	2.13	0.12	0.64	0.31	1.69	0.19	46.7	28.7	165.0	84.3	4.7	26.3
Manure	2.26	0.13	0.71	0.35	1.25	0.19	46.0	29.0	121.7	76.7	5.0	23.0
P-value	0.10	0.02	0.15	0.02	0.06	1.00	0.95	0.89	0.40	0.40	0.64	0.63

Table 4.—Mean biomass accumulation (+ standard error)[†] for the greenhouse experiments after one growing season, West Virginia

Species	Treatment	Leaf	Root	Stem	Total	Survival
Greenhouse study 1						%
Chinkapin [‡]	Rainwater	11.4 (0.9)a	22.0 (2.7)	9.1 (1.2)	42.5 (4.0)	100
Chinkapin	Rainwater+PBC [§]	4.5 (1.6)b	14.4 (4.7)	12.6 (2.1)	31.5 (6.9)	33.3
Chinkapin	Mine Water	10.1 (0.9)a	19.8 (2.7)	9.0 (1.2)	38.9 (4.0)	100
Chinkapin	Mine Water+PBC	na	na	na	na	0
	P-value	0.01	0.41	0.35	0.41	
Greenhouse study 2						
Chinkapin	Rainwater	6.1 (1.2)	18.6 (2.8)	4.3 (0.7)	29.0 (3.8)	100
Chinkapin	Rainwater+WBC [§]	7.2 (1.2)	12.6 (3.1)	3.7 (0.7)	23.4 (4.2)	80
	P-value	0.53	0.19	0.56	0.36	0.96
Greenhouse study 3						
Chestnut [‡]	Rainwater	16.6 (1.6)a	43.9 (7.9)a	28.8 (3.9)	89.4 (12.6)a	100
Chestnut	Rainwater+PBC	7.5 (2.1)b	16.2 (10.1)b	20.6 (5.1)	44.4 (16.2)b	60
	P-value	0.01	0.07	0.25	0.07	0.94

[†] Values with the same letter within a biomass component and study are not statistically different.

[‡] Chinkapin = Allegheny chinkapin. Chestnut = Dunstan chestnut.

[§] PBC= poultry-based biochar. WBC = wood-based biochar.

The second study compared Allegheny chinkapin seedlings grown in mine soil only to ones with mine soil amended with wood biochar (Table 4). Again, no statistically significant difference existed between the treatments. The largest difference between treatments was for root biomass, which differed by 32 percent. Stem and leaf tissue biomass differed by 16 and 18 percent, respectively.

Foliar elemental concentrations for seedlings growing in wood-based biochar amended soil were higher for N, K, Ca, and Al compared to unamended seedlings (Table 5). The greatest difference between treatments was for Al concentrations, which differed by a factor of three.

Foliar concentrations for rainwater-irrigated treatments between studies 1 and 2 were quite different for most elements. However, these treatments are not directly comparable because the initial seedling quality of the second study was relatively poor (much smaller seedlings) when compared to the rainwater-treated seedlings of the first greenhouse study.

The third greenhouse study investigated the growth and survival of Dunstan chestnut in soil amended with poultry biochar. All biomass values were significantly smaller (except for stem) for seedlings under the biochar treatment (Table 4). The most sizable difference between treatments occurred for root biomass, which differed by 270 percent.

Foliage testing on the composite samples suggests the poultry biochar treatment greatly increased the macronutrients in Dunstan chestnut seedlings (Table 5). Although not statistically comparable, foliage analyses between the field and greenhouse studies for Dunstan chestnut suggest similar concentrations for unamended seedlings. Biochar-treated seedlings had greater concentrations than manure-treated ones.

Table 5.—Foliar concentrations for seedlings grown in a greenhouse on soils from a reclaimed mine site in West Virginia

Species	Treatment	N	P	K	Mg	Ca	S	B	Zn	Mn	Fe	Cu	Al
-----%-----								-----mg/kg-----					
Greenhouse study 1													
Chinkapin [†]	Rainwater	2.93	0.16	0.80	0.53	1.88	0.29	27	34	377	93	11	13
Chinkapin	Rainwater+PBC [‡]	3.21	0.28	1.40	0.76	2.77	0.45	40	72	472	144	18	16
Chinkapin	Mine water	2.62	0.12	0.64	0.52	1.99	0.29	20	23	254	73	6	23
Greenhouse study 2													
Chinkapin	Rainwater	2.09	0.09	0.53	0.43	1.78	0.22	23	23	210	70	4	28
Chinkapin	Rainwater+WBC [‡]	2.24	0.09	0.54	0.40	1.87	0.20	23	23	182	59	4	86
Greenhouse study 3													
Chestnut [†]	Rainwater	2.29	0.11	0.62	0.37	1.29	0.22	29	32	157	70	5	24
Chestnut	Rainwater +PBC	2.66	0.13	0.76	0.52	1.96	0.28	28	24	171	67	6	22

[†] Chinkapin = Allegheny chinkapin. Chestnut = Dunstan chestnut.

[‡] PBC = poultry-based biochar. WBC = wood-based biochar.

DISCUSSION

If perennial seed-based energy crops (e.g., tree plantations established for seed production) are to be a viable feedstock for energy production in the future, large areas will need reforestation. Critical to this endeavor will be rapid establishment and development of these tree crops.

The amendments used in these field and greenhouse studies were not beneficial to seedling growth and development through the first growing season. Though it is possible that the treatment effects will manifest themselves in coming years, a major goal for any planting is to maintain high survival and promote early growth. Rapid growth is especially desirable in reclaimed mine sites in Appalachia to overcome intense weed competition and high deer populations. Weeds and deer were managed in the field study via herbicides and tree tubes; however, these treatments were expensive. Eliminating the need for tubes and reducing subsequent herbicide applications will more than offset the expense of applying fertilizer or manure (Barlow et al 2009, Texas Forest Service 2013).

Composted cow manure as soil amendment has been beneficial for many crops. Beneficial changes in the soil environment include increases in pH, cation exchange capacity, organic matter, and nutrient concentrations (Gil et al. 2008, Raviv 2005). The lack of response to the manure was not supported by the foliar analysis, which indicated increased nutrient concentrations for many major elements (Table 3) relative to the control. These tests did suggest N, K, S, and B are still somewhat low even for seedlings growing on manure-treated soils. Typical composted manure contains 1.38 percent N, 0.042 percent P, 0.054 percent K, 0.003 percent Ca, and 0.003 percent Mg, and has a pH of 7.5 (Miller et al. 2012). It is possible that a greater amount of manure (or more time to become available) will be required to overcome soil deficiencies (especially N, P, K).

The poultry biochar treatment was initially toxic to many of the seedlings. Though not quantified, many seedlings treated with poultry biochar lost leaves during the early summer, likely due to high soluble salt concentrations. Towards the end of the growing season, however, the surviving poultry biochar-treated seedlings looked much healthier (less chlorotic) than the untreated seedlings, presumably because some of the salts were leached out. Some agricultural crops have been shown to have reduced germination and

yields with poultry biochar applications greater than 2.5 percent (Revell et al. 2012). Further research is needed to determine appropriate poultry biochar application rates.

Similar to our results, other studies have shown wood-based biochar has no effect on tree seedling growth. McElligott (2011) demonstrated that *Populus trichocarpa* did not respond to 25- and 50-percent hardwood biochar additions to native Andisols. Heiskanen et al. (2013) failed to show a growth response after applying up to 60 percent conifer-based wood biochar to Norway spruce. Both studies suggest the lack of response was partly due to low N levels in the biochar (the principle limiting element) and possibly immobilization, and indicate that biochar plus inorganic fertilizers may be required to see beneficial effects related to biochar.

ACKNOWLEDGMENTS

We thank Jeff Slahor and Larry Osborn of the Appalachian Hardwood Center for their assistance in the field. Funding was provided by the Dr. Samuel Foundation Grant through West Virginia University Davis, College of Agriculture, Natural Resources, and Design.

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FIRST-YEAR GROWTH FOR TWO OAK SPECIES AND THREE PLANTING STOCKS PLANTED ON AREAS DISTURBED BY HURRICANE KATRINA

Andrew Dowdy, Andrew W. Ezell, Emily B. Schultz, John D. Hodges, and Andrew B. Self¹

ABSTRACT

Introduction

Bottomland hardwood forests were damaged by Hurricane Katrina in 2005 when it made landfall along the Gulf Coast. Regenerating these areas, which can be difficult without planning and artificial regeneration, has often been problematic when using 1-0 bare-root seedlings because of inconsistencies with the seedling quality. Some growers have begun producing containerized and large-container potted seedlings to provide a more consistent seedling. This study evaluated the first-year height and ground line diameter (GLD) growth of three different planting stocks (1-0 bare-root, conventional containerized, and EKOgrown® [EKOsystems Partners, Ithaca, NY] seedlings) of water oak (*Quercus nigra*) and swamp chestnut oak (*Q. michauxii*).

Methods

Research was conducted on two privately owned tracts located in southeast Mississippi, near Hattiesburg and Lucedale. Soil series represented in this study were Freest-Susquehanna-Prentiss and Lenoir silt loam. Rainfall recorded for the sites from April through September 2005 was 826 mm for Hattiesburg and 1133 mm for Lucedale.

The Malone tract (31°23'47.93" N, -89°28'33.24" W) near Hattiesburg was a mixed stand of loblolly pine (*Pinus taeda*), sweetgum (*Liquidambar styraciflua*), and water oak prior to Katrina. After the hurricane, a salvage operation was performed and the tract has been kept open by periodic mowing with a tractor. The few remaining stems on the area were injected with a 20 percent aqueous solution of Arsenal® (Imazapyr; BASF Specialty Chemicals, Research Triangle Park, NC) to prepare for planting. A hardpan about 25 cm below ground was reported by the planting crews.

The second tract, the Welford tract (30°49'27.27" N, -88°27'13.86" W) near Lucedale, was also a mixed stand of loblolly pine and hardwoods prior to Katrina. A small drainage in the center of the tract contained several stems of pond cypress (*Taxodium ascendens*). According to the landowner, this tract floods during moist winters and springs because of its close proximity to the Escatawapa River. After Katrina, a salvage operation was performed and the remaining debris was piled. The tract has been mowed and plowed every year for a wildlife food plot. Remaining stems were injected with a 20 percent aqueous solution of Arsenal to prepare for planting.

The Malone tract has a study area established to accommodate 1,800 seedlings planted on a 3.05 m by 3.05 m spacing. The Welford tract has a study area to accommodate 1,800 seedlings planted on

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a 2.74 m by 2.74 m spacing. Spacing was altered for the Welford tract because of its limited size. A compass and two 92-m surveyor's tapes were used to ensure row straightness and uniform tree spacing. Each study area was divided into three replicates. Six plots containing 100 planting locations were randomly assigned to groups of adjacent rows within each replicate to represent each of the six species and planting stock combinations. Pin flags of different colors were used to distinguish planting locations for species and planting stock combinations. A 1.2-m-long piece of rebar was placed at the beginning and end of each planting row and an aluminum tag was attached denoting the replicate, row number, species, and planting stock. The corners of each study area were marked with 3.0-m-long pieces of polyvinyl chloride pipe placed over a 1.2-m-long piece of rebar to prevent disturbance to the study area.

Seedling GLD was measured with a digital caliper to the nearest mm. Total height was measured with a meter stick to the nearest cm. Initial measurements of the seedlings were taken shortly after planting in late February 2013. First-year measurements were taken on November 2-3, 2013.

Results

Both sites had similar results. The conventional containerized seedlings had the most significant growth for GLD and total height for all three planting stocks. Ground line diameter growth and total height growth were significantly greater for bare-root seedlings than for EKOgrown seedlings but significantly less than for conventional containerized seedlings. The following tabulation shows GLD results for water oak and swamp chestnut oak seedlings; growth values followed by different letters are significantly different at the level of $\alpha = 0.05$.

EKOgrown seedlings exhibited a negative height growth on both sites for both species. We observed and recorded moderate to extreme dieback in these seedlings. Water oak bare-root seedlings exhibited browse damage from deer. Deer browse was observed on 50 percent of the water oak bare-root seedlings on both sites, which accounted for the overall lack of height growth in that species. The tabulation below shows height growth results; growth values followed by different letters differ significantly at the level of $\alpha = 0.05$.

Species/planting stock	GLD (mm)	
	Initial	Growth
Water oak/bare-root	6.52	1.15 B
Water oak/containerized	4.60	3.55 A
Water oak/EKOgrown	10.83	1.90 C
Swamp chestnut oak/bare-root	7.88	1.75 B
Swamp chestnut oak/containerized	6.20	1.95 A
Swamp chestnut oak/EKOgrown	10.98	0.65 C

Species/planting stock	Height (cm)	
	Initial	Growth
Water oak/bare-root	54.6	-2.7 B
Water oak/containerized	52.6	5.4 A
Water oak/EKOgrown	130.2	-11.4 C
Swamp chestnut oak/bare-root	45.8	2.9 B
Swamp chestnut oak/containerized	48.3	6.2 A
Swamp chestnut oak/EKOgrown	100.4	-14.9 C

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REGENERATION DYNAMICS DURING OAK DECLINE IN ARKANSAS

Martin A. Spetich¹

ABSTRACT

Methods

From 2000 to 2008 hardwood regeneration <5 cm diameter at breast height were individually tagged and monitored in the Boston Mountains of northern Arkansas. The study site is a 32-ha area in an upland oak-hickory stand that was approximately 70 years old in 2000. Mean basal area for all standing trees in 2000 was 25.9 m²/ha, and there were 417 standing trees/ha. Stocking was 88 percent. In 2000 we established 480 permanent circular regeneration plots, each with a 1.31-m radius. In each quarter of each regeneration plot, we measured species, distance, azimuth, height, ground diameter, stem age, and origin of the two tallest trees of northern red oak (*Quercus rubra*) and white oak (*Q. alba*). By mid-2001 this stand began to exhibit symptoms of severe oak decline. In 2004, a prescribed fire was applied to one-fourth of the study area.

Results and Discussion

In this preliminary analysis, survival probability of red and white oak regeneration in the burned and unburned areas was examined by using logistic regression. Red oak survival increased with increasing initial stem diameter. For red oak with initial basal stem diameters <7 mm, regeneration had a lower probability of survival in the burned area than in the unburned area. However, for stem diameters of ≥7 mm that relationship was reversed (Fig. 1). Maximum probability of survival for the largest stem diameters for red oak was 0.78 in the burned areas and 0.70 in the unburned area. The highest

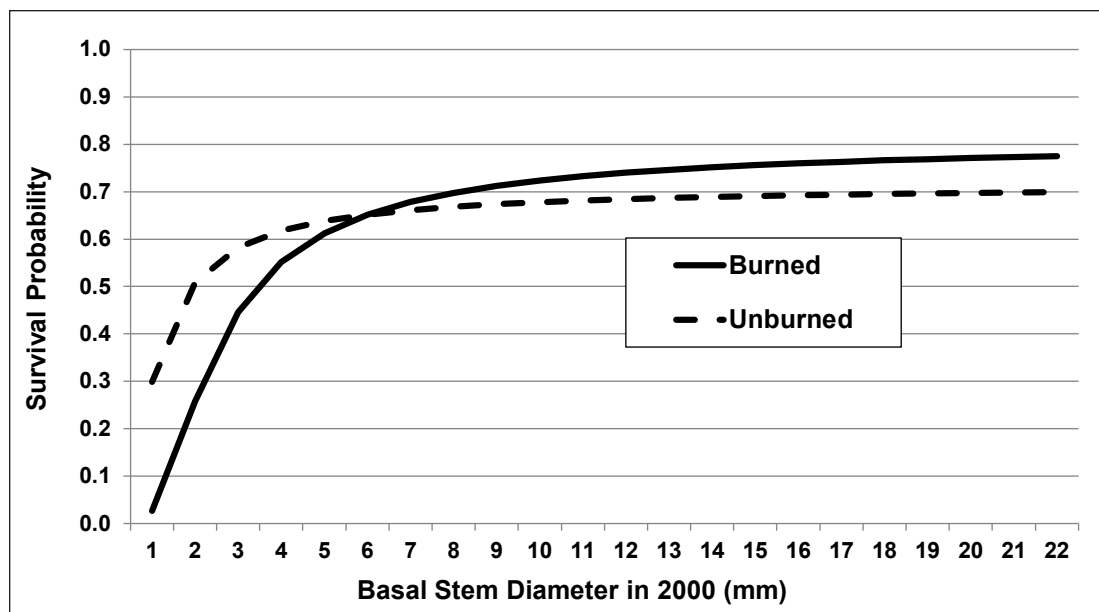


Figure 1.—Probability of 2008 red oak regeneration survival, Boston Mountains, Arkansas.

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probability of survival occurred in stems >6 mm basal stem diameter for both burned and unburned areas. The logistic regression model for red oak regeneration survival in the area treated with prescribed fire in 2004 is as follows:

$$PSROb = \frac{1}{1 + EXP \left\{ - \left[1.466 - \left(5.046 * \frac{1}{BSD} \right) \right] \right\}}$$

where

PSROb = probability of survival of red oak regeneration in burned area, and
BSD = basal stem diameter in mm in the year 2000.

The predictor 1/BSD *p*-value was <0.001. The Hosmer-Lemeshow goodness of fit (*p*-value) of the model was 0.317. Small *p*-values designate a poor fit of the equation to the data whereas large values (>0.05) indicate a good fit. Based on the Hosmer-Lemeshow goodness-of-fit statistic, differences between estimated survival probabilities and observed responses were not significant.

The logistic regression model for red oak regeneration survival in the unburned area is:

$$PSROub = \frac{1}{1 + EXP \left\{ - \left[0.922 - \left(1.774 * \frac{1}{BSD} \right) \right] \right\}}$$

where

PSROub = probability of survival of red oak regeneration in the unburned area, and
BSD = basal stem diameter in mm in the year 2000.

The predictor 1/BSD *p*-value was 0.007. The goodness-of-fit (*p*-value) of the model was 0.748. As with the results for the model of red oak regeneration survival in the burned area, the goodness-of-fit statistic indicates that the estimated survival probabilities and observed responses were not significantly different.

In the white oak model, stem diameter by itself was not a significant factor. White oak survival increased with increasing initial stem age. However, there was very little difference in white oak survival between the burned and unburned areas with maximum survival probabilities of 0.81 in the burned area and 0.83 in the unburned area for the oldest initial stem ages (Fig. 2). High survival probabilities for white oak began to be reached at a stem age of 5 years. The logistic regression model for white oak regeneration survival in the area treated with prescribed fire in 2004 is below:

$$PSWOB = \frac{1}{1 + EXP \left\{ - \left[1.671 - \left(3.209 * \frac{1}{AGE} \right) \right] \right\}}$$

where

PSWOB = probability of survival of white oak regeneration in burned area, and
AGE = stem age in years of seedling in the year 2000.

The predictor 1/AGE *p*-value was 0.001. The Hosmer-Lemeshow goodness of fit (*p*-value) was = 0.287, indicating a good fit of this model to the observed data.

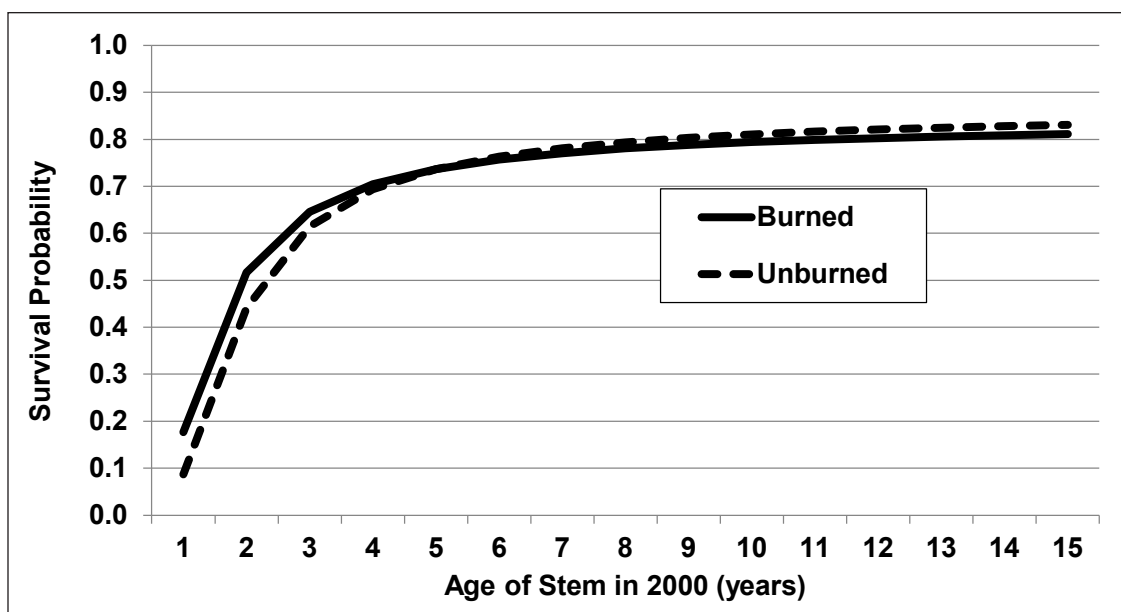


Figure 2.—Probability of 2008 white oak regeneration survival, Boston Mountains, Arkansas.

The logistic regression model for white oak regeneration survival in the unburned area is as follows:

$$PSWOub = \frac{1}{1 + EXP \left\{ - \left[1.875 - \left(4.224 * \frac{1}{AGE} \right) \right] \right\}}$$

where

PSWOub = probability of survival of white oak regeneration in the unburned area, and

AGE = stem age in years of seedling stem in the year 2000.

The predictor 1/AGE *p*-value was <0.001. The Hosmer-Lemeshow goodness of fit (*p*-value) was 0.731, which indicates there is no significant difference between the model-predicted and observed survival values.

Conclusions

Based on preliminary results of this study, a reasonable management target for red oak regeneration of ≥6 mm basal stem diameter regardless of fire treatment is recommended. For white oak a 5-year stem age maximized the probability of survival regardless of treatment and may be a useful predictor of survival.

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FOREST ASSESSMENT

FOLLOWING THE FATE OF HARVEST-DAMAGED TREES 13 YEARS AFTER HARVESTS

Randy G. Jensen and John M. Kabrick¹

ABSTRACT

Introduction

Logging damage to residual trees during harvest operations can reduce the future volume, quality, and value of wood products. Timber harvests in 1996 on the Missouri Ozark Forest Ecosystem Project (MOFEP) provided a rare opportunity to follow the fate of trees wounded by felling or by skidding with rubber-tired skidders.

Methods

Study sites on MOFEP are in the western Central Hardwood region in oak-hickory and oak-pine forests. The predominant species on MOFEP are white oak (*Quercus alba*), black oak (*Q. velutina*), scarlet oak (*Q. coccinea*), shortleaf pine (*Pinus echinata*), and hickories (*Carya* spp.), which account for 86 percent of the trees and 87 percent of the basal area of trees ≥ 4.5 inches diameter at breast height (d.b.h.). Tree survival rates on MOFEP were determined in the dormant season of 1997-98 on 9,749 live trees, of which 938 were bark wounded in 1996, and monitored again in 2009-10 during the MOFEP overstory inventory conducted on 298 permanent 0.5-acre vegetation plots on 6 study sites ranging in size from 772 to 1,271 acres. In summer 1997, the following bark wound dimensions were measured on wounded trees: wound width, length, height above ground line, and azimuth the wound was facing. In summer 2009, bark wound dimensions were remeasured on 745 live trees, 13 years after harvest, to determine bark closure rates. Analysis of variance was used to examine the interaction of wounding with species, management method, and crown or size class on tree survival and on wound closure. For significant effects, the Scheffe method of multiple comparisons was used to compare individual means.

Results

The survival of all trees was greatest for white oak (96 percent), followed by hickories (93 percent), shortleaf pine (85 percent), scarlet oak (76 percent), and black oak (73 percent) over a 12-year period ($P = 0.0008$). The survival of all bark-wounded trees (87 percent) was slightly higher than for nonwounded trees (82 percent) on the same vegetation plots ($P = 0.007$), although there is no apparent explanation. Survival in wounded trees ($P = 0.0031$) was 99 percent for white oak, 92 percent for hickories, 84 percent for black oak, and 73 percent for scarlet oak. Survival on sites under even-age management was 86 percent compared to 83 percent survival on uneven-aged sites, but the difference was not significant. Wounded trees within the dominant crown class had 93-percent survival; survival rates decreased with decreasing crown class to suppressed trees with 74-percent survival ($p = 0.0557$).

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After 13 growing seasons, the wounds of 76 percent of the trees that had been damaged during harvesting had completely closed, including the 23 percent of wounded trees that had multiple wounds. White oak was the most effective at closing all bark wounds ($P = 0.0161$) at 92 percent, followed by scarlet oak (82 percent), black oak (58 percent), hickories (36 percent), and shortleaf pine (17 percent). Crown position was not a significant effect on the ability of trees to close bark wounds ($P = 0.1492$), nor was size class ($P = 0.2082$) when evaluating large saplings (4.5 to 6 inches d.b.h.), pole timber (6 to 11 inches d.b.h.), and saw timber (≥ 11 inches d.b.h.).

Discussion

Dwyer et al. (2004) attributed the low incidence of tree damage after 1996 MOFEP timber sales in part to close communication between timber sale administrators and timber buyers and logging crews. Topics of discussion should include careful location of skid trails, landings, and haul roads to avoid future crop trees when possible. Overall, our analysis showed that wounding during felling and skidding operations would not increase tree mortality, and the wounds of the damaged trees are likely to close or heal within 13 years. When possible, damaged trees should be culled during the current operation as the wounds might not be noticeable during timber marking for the next harvest entry.

Acknowledgments

Funding for this research was provided by the Missouri Department of Conservation (MDC). We would like to thank the many technicians who collected MOFEP overstory and tree wound data and Sherry Gao, MDC, for biometric support.

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A POTENTIAL QUANTITATIVE METHOD FOR ASSESSING INDIVIDUAL TREE PERFORMANCE

Lance A. Vickers, David R. Larsen, Daniel C. Dey, John M. Kabrick,
and Benjamin O. Knapp¹

Abstract.—By what standard should a tree be judged? This question, perhaps unknowingly, is posed almost daily by practicing foresters. Unfortunately, there are few cases in which clearly defined quantitative (i.e., directly measurable) references have been established in forestry. A lack of common references may be an unnecessary source of error in silvicultural application and potentially confounds efforts to understand the biology and ecology of forest processes. The utility of the few references that have been established is immense. For example, foresters can assess site productivity for an area by calculating site index, which is a standardized, quantitative reference for site productivity. Moreover, foresters can compare site productivity across multiple areas (say, the Missouri Ozarks versus southern Indiana) because site index is a common reference that is widely accepted. Similarly, foresters can evaluate and compare stand density by calculating stocking percent.

Perhaps no field of forestry could better benefit from greater quantification than regeneration and early stand dynamics. Regeneration is arguably the most critical component of sustainable forest management, yet there has been little effort to develop quantitative methods for assessing the performance of individual trees throughout the regeneration period. There are qualitative metrics that are certainly useful (e.g., crown class), but subjectivity and other constraints complicate and limit their application. Furthermore, crown classes typically are not applicable for very young trees before crown closure. As a result, regeneration models and evaluations that use crown class to define success typically make predictions late in the post-harvest regeneration period.

In an effort to improve upon quantitative methods, we have developed growth references for a limited number of sites, scenarios, and species. Growth references have long been used in the medical field for quantitative clinical assessment of juvenile development. These growth references typically are in the form of charts with selected quantiles of a reference distribution for a variable of interest plotted against age (e.g., height-age). Our objective is to demonstrate the potential utility of this quantitative method for assessing the performance of individuals by using young trees in the Missouri Ozarks as an example. We describe how growth references based on quantiles from a height distribution of trees at the same age on similar sites could be useful tools to quantitatively compare trees that are too young to be classified by traditional crown classes. We also describe how growth reference charts may extend the inference window of stand reconstruction techniques (i.e., stem analysis). Potential for additional applications in modeling regeneration and early stand dynamics, as well as improving precommercial silvicultural prescriptions, is discussed.

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

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EFFECTS OF PRESCRIBED FIRE ON THE WOOD QUALITY AND MARKETABILITY OF FOUR HARDWOOD SPECIES IN THE CENTRAL APPALACHIAN REGION

Janice K. Wiedenbeck and Thomas M. Schuler¹

Abstract.—A series of research studies addressing the effect of prescribed fire on oak (*Quercus* spp.) regeneration has been ongoing on the Fernow Experimental Forest in northeastern West Virginia for 10 years. The study site is a mesic, mixed oak forest. Two prescribed fires were conducted in spring 2002 and 2005. In 2010, a shelterwood harvest was conducted. A complementary study was undertaken to evaluate the effects of the prescribed fires on the quality and marketability of the wood removed in this harvest. Seventy-four logs from the four most populous commercial species located on the study site were tracked from forest through milling. Before harvest, trained timber graders visually evaluated the residual effects of the prescribed fires on tree grade and merchantable volume. At the sawmill's log yard, pictures were taken of the logs and paint marks were placed on the log ends to indicate the side of the log most affected by the fire. During sawing, the first two boards recovered from the marked side were marked for examination. The percentage of these most "at-risk" boards showing indications of defect potentially attributable to the heat of the fire ranged from 10 percent for yellow-poplar (*Liriodendron tulipifera*) to 65 percent for red maple (*Acer rubrum*), which translates to between 2 and 16 percent of all boards sawn from these butt logs. Fire-associated defects included mineral stain, decay and incipient decay, shakes, and checks.

INTRODUCTION

The use of prescribed fire in eastern hardwood forest management has gained the acceptance of many forestry, wildlife, and ecology professionals. Resource managers use fire as a tool to develop desired regeneration, habitat, and ecological restoration outcomes and to reduce fuel buildup to lower the risk of wildfires. Numerous national forests in the U.S. Forest Service's Eastern Region have revised their planning documents to include fire as a restoration tool (Nowacki et al. 2009). In Ohio, use of prescribed fire by the state's Division of Forestry increased from less than 100 acres and three or fewer fires per year in the 1990s to an average of more than 1,000 acres and seven fires per year from 2000 through 2008 (Bowden 2009) and to an average of about 2,500 acres burned today (Ohio Department of Natural Resources 2013). The "2012 National Prescribed Fire Survey Report" (Melvin 2012) indicates that the acres to which prescribed fire has been applied in the central Appalachian region has been trending up in Pennsylvania, Virginia, Tennessee, Kentucky, and Ohio. The Monongahela National Forest in West Virginia sets annual targets for prescribed burning at about 3,000 acres. Restoration goals are being realized, but the effects on wood quality are not fully known.

Eastern hardwood tree mortality caused by fire has been evaluated in multiple studies. Factors assessed in these studies included fuel types and loadings (Brose and Van Lear 1999, Wendel and Smith 1986, Yaussy and Waldrop 2010), bark thickness (Harmon 1984, Yaussy and Waldrop 2010), tree diameter (Harmon 1984, Hutchinson et al. 2005, McCarthy and Sims 1935), season in which

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fire occurred (Brose and Van Lear 1999), fire severity (Regelbrugge and Smith 1994, Yaussy and Waldrop 2010), and tree vigor before fire exposure (Yaussy and Waldrop 2010). Fire damage severity based on degree of bole damage and crown condition also has been widely studied (Brose and Van Lear 1999, Pomp et al. 2008, Wendel and Smith 1986). The process of wound formation after fire injury has been studied by Sutherland and Smith (2000) and Smith and Sutherland (2001, 2006).

Only two studies appear in the literature related to the timber value impacts of fire. (Both studies were based on wildfire.) In a study of multiple stands in Kentucky, the average timber volume loss per acre due to wildfires was found to be 2,298 board feet and the average value loss per acre was \$404 (Reeves and Stringer 2011). A study conducted in a region of southern West Virginia that has had a long history of wildfires found that net volume and dollar value per acre declined with increasing fire frequency; value decreased as much as \$619 per acre, representing a 54-percent decline in stumpage value for stands subjected to six wildfires over 36 years (Wood 2010). Information about the impact of fires on hardwood timber quality, volume, and value is of interest to land owners and managers but is not well substantiated or understood across a range of timber types, landscapes, and fire conditions. Quality and value information would be useful for decisionmaking related to fire prevention efforts and prescribed fire use as well as stumpage pricing. Extending the information to include insights into the impact of fire on wood product quality, volume, and value potential would be a further step toward informing land owners and managers as well as timber buyers and producers so that decisions on timber management and procurement can be optimized.

OBJECTIVES

The preponderance of research on the effects of fire, both wildfire and prescribed fire, on hardwood species has focused on mortality rates and tree quality impacts. There is a dearth of research that extends this analysis to look at product recovery and quality impacts. This information has been sought out by the West Virginia Division of Forestry, among others. The overall objective of this study was to begin to develop information on the impact of fire on the volume and value yields of lumber of affected trees. The principal hypothesis under investigation was: There is no relationship between species and the occurrence of six types of lumber defects judged to have been caused by exposure to two prescribed fires.

An important, nonmeasurable objective of this research was to develop insight from this pilot study that will allow us to design a larger scale, longer term study on the effects of fire on lumber volume and value recovery that will be used by resource managers in making decisions about the use of prescribed fire and the value of trees that have been compromised by wildfire.

METHODS

The overstory trees from the prescribed fire and oak regeneration study conducted on the Fernow Experimental Forest in West Virginia (Schuler et al. 2013) were scheduled for removal during the dormant season of 2009-10. This was the first stage of the shelterwood removal process. The prescribed fires were conducted in the Canoe Run watershed of the Fernow Experimental Forest (39.03° N, 79.67° W). The elevation of the study site ranged from 1,920 to 2,200 feet with a western aspect and a mean slope of 39 percent. Overstory species composition of much of the Fernow is described as mixed mesophytic; this study site is dominated by northern red oak (*Quercus rubra*),

chestnut oak (*Q. prinus*), and white oak (*Q. alba*) (Schuler et al. 2013). The site is a second growth forest that is about 100 years old. Two prescribed fire treatments have been applied to the site; the first was conducted in April 2002 and the second in April 2005. The maximum temperature probe readings recorded during these two burns were 576 °F in 2002 and 621 °F in 2005 and the associated rates of spread were 30 and 144 feet per minute, respectively (Schuler et al. 2013). These prescribed fires were designed to minimize damage to overstory trees and were characterized by Schuler et al. (2013: 432) as “moderate to low intensity with flame lengths less than 3 feet resulting from the combustion of leaf litter and 1-hour surface fuels.”

The harvest of shelterwood trees from this site represented an opportunity to evaluate the wood quality (on a macroscale) and potential product value of hardwood trees that had been exposed to prescribed fire. Arrangements were made with the sawmill that purchased the timber to allow us to track the breakdown of about 80 logs at the sawmill. The sample size of this study was dictated by the need to limit our interference with the operations of the cooperating sawmill. The 80-tree sample size was split among 4 species to provide a broader look at the wood quality impacts of prescribed fire on commercial species of the region in anticipation of a more comprehensive future study.

Sample Selection and Quality Assessment in the Woods

Based on comprehensive data sets maintained for the fire and oak regeneration study, four commercial tree species that were prevalent among the larger shelterwood stems (>13 inches diameter at breast height [d.b.h.]) on the study site were selected for this study: red maple (*Acer rubrum*), red oak, white oak, and yellow-poplar (*Liriodendron tulipifera*). Sample trees for this study were selected from within and proximal to the established plots of the fire and oak regeneration study (Schuler et al. 2013). Trees in these plots have been measured about every 3 years since the study was established in 2000. Data loggers positioned in each plot during the prescribed fires provided information on temperature intensity and rate of fire spread. Trees having a d.b.h. of ≥ 13 inches (i.e., large enough to meet Hardwood Tree Grade 2 minimum d.b.h. requirement) in each of the four target species that were scheduled for harvest were included in this study.

Sample selection was conducted by timber technicians at the Fernow who are experienced in species identification, measurement, and tree grading. The criteria for selection were: (1) trees only of the four target species (target of 20 trees per species), (2) trees from among those marked for removal in the upcoming harvest, and (3) live stems only. Inclusion in the sample did not depend on the type or amount of fire scarring evident (if any) on the tree bole. After the sample was selected based on the three listed criteria, the following data were collected for each tree:

- Species
- Type of fire scar (none, black bark, bark sloughing, cat face, butt scar)
- Length of fire scar from base of tree
- Circumference of fire scar around tree at widest point
- Scar location relative to slope (uphill or uphill and side-hill)
- Tree d.b.h.
- Tree grade
- Amount of deduction, if any, due to burn scar



Figure 1.—Red maple logs from Fernow Experimental Forest, West Virginia, at sawmill log yard with paint and arrows designating the face on each log that was affected by fires and a log identifier (X32, X1, X11) for use in tracking the lumber sawn from each study log.

Butt logs from the sample trees were marked with paint and flagged. After harvest, the logs from each tree were again marked with paint and the tree number was marked on the end of each log for tracking purposes at the sawmill. The study sample was segregated at the log landing and hauled to the mill, where it was again segregated from the other logs in the log yard.

Sawmill Study Methods

At the sawmill, in advance of the tracking study, sample logs were spread out in the log yard and the face of each log with the most significant evidence of fire was designated by marks painted on the log ends (Fig. 1). Different colors of paint were used for each species to help with tracking during breakdown in the sawmill. In addition, a log scaler at the cooperating sawmill graded and scaled each of the study logs and provided that information to us so that we could evaluate the value loss associated with the fire-caused cull deductions noted by the timber technicians during sample selection.

For the tracking study in the sawmill, all logs of a given species were processed before the logs of another species were brought into the sawmill. As each log was sawn, the marks on the ends of the logs were used to identify the first two boards sawn from the fire-affected face. These two boards were marked and renumbered and the orientation of the marks was applied so that we could be certain which end of each board was from the butt end of the study log. In the sawmill, 19 red maple, 21 red oak, 16 white oak, and 23 yellow-poplar were sawn as part of this study.

Between communicating with the forklift operator and the lumber marks placed on the sample boards of interest, most of our marked boards were successfully separated for us into unique stacks in the lumber yard to enable us to inspect lumber quality. For 67 of the sample trees, both boards were successfully tracked and evaluated for quality. For seven of the sample trees, only one of the two boards sawn from underneath the fire-affected surface was available for evaluation. Board samples from six sample trees were not successfully segregated, so they could not be recovered and evaluated. In the end, board samples (1 or 2 boards) from 17 red maple trees, 20 red oak trees, 16 white oak trees, and 21 yellow-poplar trees were evaluated for defect potentially caused directly or indirectly by the prescribed fire.

Lumber evaluation at the sawmill was designed only to identify defects occurring on the first two boards removed below the bark on the face of the butt log that was most affected by the heat of the prescribed fire. Defects identified were those that could have been caused by the heat generated by the fire. Thus, defects such as knots and double pith were not documented; but decay, mineral stain, shakes, and checks were recorded. When the defects of interest were located within the bottom 6 feet of the board, they were captured in the tally. A stain or decay defect that was located farther up the board without associated defect area at the bottom of the board, was not included.

Data Coding and Analysis

Butt log quality effects evaluated in the woods that were judged to be associated with the prescribed fire were coded as 0 (no effect) or 1 (grade- or scale-reducing effect). We used a generalized linear model via PROC GENMOD (SAS 9.3, SAS Institute Inc. 2011) with tree species as a fixed effect. We used the binomial distribution and the logit link function. After running the model, we had unstable maximum likelihood estimates because yellow-poplar yielded a “perfect” model (i.e., no variation in the response outcome because none of the yellow-poplar tree boles had grade or scale reductions). We deleted this species and reran the analysis. Statistical differences in the three remaining species were evaluated by least means squares. The test was conducted with a significance level of $\alpha = 0.05$. Pivot table analysis and simple calculations were used to further examine log defects by species and the value loss associated with these defects.

Lumber quality observations tallied at the sawmill were evaluated by using a generalized liner model via PROC GENMOD (SAS Institute Inc. 2011). Using expert opinion, we ordered the six outcomes from least to most detrimental to lumber quality. The lumber quality rating scores were coded as follows: 1 - clear, 2 - mineral stain, 3 - checks, 4 - ingrown bark, 5 - decay, and 6 - shake. The fixed effect in the model was species. Because of the inherent ordering we used a multinomial distribution model with a cumulative logit link function (Allison 1999). To evaluate statistical differences in species we used least squares means.

RESULTS AND DISCUSSION

Tree Quality

Of the 79 trees marked for inclusion in this study, all but 9 trees had visible signs of bark alterations due to the prescribed fires. However, only 10 trees were judged to have suffered tree grade or scale volume reductions. Of the 10 trees that showed a substantial fire effect, 7 were red maple stems (Table 1). Of the 23 yellow-poplar stems measured for the study, none showed visual evidence of grade or scale volume-reducing defects that could have been caused by the prescribed fires (Table 1).

The 10 trees with defects that affected the tree grade and led to a scale deduction each had a large cat face at the bottom of the butt log on the uphill face of the tree (Fig. 2). Four of these trees suffered cull (volume) deductions of 10 percent, four contained cull amounts of 20 percent, and two had cull deductions of 30 percent. None of the other types of defects (black bark, bark sloughing, small cat face, butt scar) were judged to be more than superficial in terms of their effects on the quality of the underlying wood. A log value loss associated with the cull deduction was calculated for each butt log based on the log value assigned by the scaler at the sawmill. The percentage-based log value loss for all

Table 1.—Visual quality assessment of trees selected for study of effects of prescribed fire on wood quality and value, by species, West Virginia

Grade- or scale-reducing defect present?	Evidence of fire	Red maple	Red oak	White oak	Yellow-poplar	Total
No		12	19	15	23	69
	None	4	1	1	3	9
	Black bark	3	17	10	17	47
	Bark sloughing	1	0	2	3	6
	Cat face, small	1	1	1	0	3
	Cat face, large	3	0	0	0	3
	Butt scar	0	0	1	0	1
Yes		7	2	1	0	10
	Cat face, large	7	2	1	0	10
Total		19	21	16	23	79

butt logs in the sample was 0.023 percent for red maple, 0.010 percent for red oak, 0.002 percent for white oak, and 0.000 percent for yellow-poplar.

Based on the statistical results, red maple is more prone than white oak to suffer tree grade- or scale-reducing damage from prescribed fire of low to moderate intensity ($P > 0.039$). Differences in tree grade and scale damage amounts caused by fire for red oak and white oak were not significant. Red oak and red maple tree damage results did not differ, but with a P-value of 0.055, this outcome is not absolute. Yellow-poplar was not included in the model because, with no samples of this species indicating damage, its inclusion made the model unstable. The lower occurrence of damage in yellow-poplar than in white oak shows that yellow-poplar is less prone to fire-caused tree grade and scale defects than white oak or red maple. Therefore, the null hypothesis is rejected as there appears to be a relationship between species and the occurrence of tree grade- or scale-reducing defects.

The occurrence of more fire-related defects in the red maple stems than in the white oak and yellow-poplar stems is an expected outcome. Fire simulations performed by Hengst and Dawson (1994) substantiate the belief that thicker barked species are less vulnerable to damage from certain types of fire as the bark provides cambial protection. The maximum cambial temperatures in thick-barked species are lower and the time to maximum temperature is longer; thus, a fire that spreads more quickly will inflict less damage on thicker barked species (Hengst and Dawson 1994). The average ratio of d.b.h. inside bark to d.b.h. outside bark for the four species included in this study are 0.942 for red maple, 0.921 for red oak, 0.929 for white oak, and 0.896 for yellow-poplar (Martin 1981). These ratios show that yellow-poplar has the thickest bark and red maple the thinnest bark of the species studied.

Lumber Quality

Although 79 trees were evaluated for grade and defect in the field, lumber from only 74 of these trees was evaluated after sawing due to tracking problems noted under Methods. Aligned with the log quality results, the percentage of yellow-poplar boards found to be clear of fire-related defects was higher than for the other species and the percentage for red maple was the lowest among the four



Figure 2.—Large cat face on uphill side of red maple tree that was exposed to two prescribed fires 5 and 8 years before this picture was taken (Fernow Experimental Forest, West Virginia).

species (Table 2). Also, except for red oak, the second board removed from beneath the fire-affected face of the log was clear of targeted defects (decay, incipient decay, mineral stain, checks, shake, and ingrown bark) more frequently than the first board (Table 2). Targeted defect occurrence rates shown in Table 2 indicate that mineral stain was found more frequently in red maple whereas checks and shake were found more frequently in red and white oak, known to be check-prone species.

The lumber quality differences among species were significant with yellow-poplar lumber possessing fewer of the most damaging fire-associated defects (ingrown bark, decay, shake) than the other three species. In evaluating the lumber quality summary data, this relationship between species and defect occurrence stands out (Table 2).

These trees were harvested fairly quickly after the prescribed fires were conducted (7-8 years after the first fire and only 5 years after the second fire); thus, the full impact of the prescribed fire on butt log/tree value and lumber yield and value could not be appraised fully. At most, decay occurring as a result of burn injuries will be found in the outer couple of boards, which will tend to be narrower boards than would be the case if the decay were found further into the wood. As seen in Table 2, some of the defect was “incipient decay,” an early stage of wood decay that may further proliferate and worsen with time, depending on the ability of the tree to compartmentalize the injury. As this was just the first step in the shelterwood removal process, some injured trees are expected to remain in the stand for another 15 years or more while other steps in the prescription are implemented. Time plus an additional prescribed fire could result in substantially more loss in wood volume than documented here.

Table 2.—Number and percentage of clear boards and targeted defects on boards sawn from the log section located under the face of the log most affected by prescribed fires for four species, West Virginia

Species and board position	Number and percentage of boards free of targeted defects		Targeted defects					Total number of boards surveyed
	Clear	Percent clear	Mineral stain	Checks	Decay and incipient decay	Shake	Ingrown bark	
Red maple								
1st board	5	33	6	0	3	1	0	15
2nd board	6	37	7	0	2	1	0	16
Red oak								
1st board	9	47	3	3	1	3	0	19
2nd board	9	45	2	5	2	2	0	20
White oak								
1st board	9	47	1	3	2	1	1	17
2nd board	8	57	2	2	1	0	1	14
Yellow-poplar								
1st board	18	90	2	0	0	0	0	20
2nd board	20	95	1	0	0	0	0	21

Lessons Learned for Future Research

Several elements of this preliminary study, which was planned and conducted over a short timeframe to take advantage of the impending harvest of trees that had been exposed to prescribed fires (Schuler et al. 2013), will inform a more comprehensive, designed study for which planning has begun.

Besides the obvious problems of logs and boards being lost at the sawmill, experimental design adjustments are needed. A control sample of matched trees from a nearby, unburned stand with similar stand attributes will allow for a more meaningful analysis of the impacts of prescribed fire on volume and value yield. The checks and shakes seen in the oak species in this study may or may not have been related to the fire; the control sample would help make that determination. Another sample selection factor to be implemented in future research is the selection of equal-size samples of trees from different fire-affected stands having different slopes. Another factor of interest that may help explain vulnerability of a stem to fire damage is stem d.b.h., so sample selection will include at least two distinct d.b.h. classes (e.g., 12-17 inches and 22-27 inches). Evaluating the effects of prescribed fire on wood quality and value after a greater period of time has passed will add value to the information obtained from a more expansive study.

At the sawmill, the study must capture the thickness of the slab removed as the first boards are sawn so that the depth-below-bark of the boards is known. The most important change needed in the sawmill portion of the study is the allocation of more time to lumber measurements so that specific defect sizes and recovery impacts can be captured.

CONCLUSIONS

Loss in log value associated with two prescribed fires of low to medium intensity can appear minimal upon visual inspection of stems 5 years or more after the burn was conducted. In this case, value loss of <0.25 percent was suffered overall from cull caused by decay that resulted from injury from the heat of the fire. Red maple was the most prone to fire damage among the species evaluated here. Yellow-poplar had the lowest risk of injury because of its thick bark, which insulates the wood from temperature extremes. The outer two boards sawn from the side of logs that are most directly exposed to the heat of the fires develop mineral stain and some level of decay about 65 percent of the time in red maple, about 7 percent of the time in yellow-poplar, and at an intermediate rate in red and white oak. Lumber value loss is to be expected after prescribed fires, and the occurrence rate is greater than would be expected based solely on the visual inspection of the butt logs in the woods. A much more expansive and controlled study design is required to be able to determine value loss rate ranges for different physical and market conditions. As the oak regeneration process may require 15 to 20 years or longer from the time of the first prescribed fire to the final shelterwood harvest, the effects of fire intensity and duration of time until harvest on value impacts need to be elucidated.

ACKNOWLEDGMENTS

The assistance from personnel at Augusta Lumber's Warm Springs Sawmill in Warm Springs, VA, in particular the sawmill superintendent, John Hubbard, the general manager, Leroy Evans, and the log buyer/grader, Ogi Petrov, was extraordinarily helpful and generous. Statistical insights and analysis offered by John Stanovick of the U.S. Forest Service, Northern Research Station provided a more robust analysis and interpretation of our results than we had in our first iteration of this paper.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

INVASIVE SPECIES

EFFECT OF GARLIC MUSTARD INVASION ON ECTOMYCORRHIZAE IN MATURE PINE TREES AND PINE SEEDLINGS

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Abstract.—Ectomycorrhizal fungi are mutualistic fungi that colonize the roots of many terrestrial plants. These fungi increase plant vigor by acquiring nutrients from the soil for their hosts in exchange for photosynthates. We studied the effect of garlic mustard (*Alliaria petiolata*) invasion on the density of ectomycorrhizal symbionts using two approaches. We found that roots from mature pine trees from a garlic mustard-invaded site and roots from pine seedlings grown in soil collected from a garlic mustard-invaded site had lower rates of ectomycorrhizal colonization and fewer fungal tips/cm compared to their respective controls grown in soils from uninvaded areas. These data suggest that garlic mustard invasion has the potential to alter mutualistic fungal interactions and negatively affect plants that depend on these fungi for nutrient acquisition.

INTRODUCTION

Mutualistic interactions between mycorrhizal fungi and plants are extremely important, but these associations can be difficult to observe and understand. The relationship between mycorrhizal fungi and land plants is a symbiotic interaction nearly 400 million years old; a vast majority of terrestrial plants exist in symbiosis with mycorrhizal fungi (van der Heijden and Horton 2009).

Mycorrhizal fungi receive about 10 to 60 percent of plant photosynthates in exchange for nutrients such as nitrogen (N) and phosphorous (P) obtained by the fungi. Plants receive up to 80 percent of their N requirements and up to 90 percent of their P requirements from mycorrhizal interactions (van der Heijden and Horton 2009). To accomplish this nutrient acquisition and exchange, the fungi create an increased surface area of hyphae to obtain soil nutrients, which allows the plant to allocate less energy to build extensive root systems. Fungal colonization also changes the physical structure of the root to facilitate exchange. However, mycorrhizal fungi do more than extend the area of the soil that can be explored by roots; they can also acquire nutrients in pools not available directly to plants. For example, many directly decompose litter (Leake and Read 1997) or prey upon live animals such as springtails (Klironomos and Hart 2001). Mycorrhizal symbionts also improve plant growth through other mechanisms such as influencing plant water relations (Allen 1991) and increasing resistance to pathogens (Gehring and Whitham 1994, Gehring et al. 1997, Peay et al. 2010). Mycorrhizal colonization is even more important for plant performance when plants are under stress, such as when environmental stress is high or nutrient availability is low (Gehring and Whitham 1994, Gehring et al. 1997).

Two of the most common types of mycorrhizal fungi are arbuscular mycorrhizal fungi (AMF) and ectomycorrhizal fungi (EMF). Both AMF and EMF exchange nutrients with their plant partners (van der Heijden and Horton 2009). EMF are located on roots of trees in the Pinaceae (pine), Betulaceae (birch), and Fagaceae (beech) families as well as several other families, and estimates suggest that approximately 2,000 species of woody perennial plants are host to EMF (Gehring et al. 1997, Peay

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et al. 2010). The majority of EMF-plant relationships are obligate symbioses (Koske 1992). In fact, all species of the Pinaceae family are considered to be involved in an obligate symbiosis with EMF, meaning that these plants cannot survive without their EMF partners (Horton et al. 1998).

Changes to plant community structure or soil characteristics can alter the belowground mutualistic interactions. Plant species can negatively affect EMF by allelopathic means as well as through competition (Wolfe et al. 2008). Exotic plant species may have similar detrimental effects on fungi because these novel species can cause ecological changes in their introduced range (Stinson et al. 2006). Alien invaders such as garlic mustard (*Alliaria petiolata*) may disrupt beneficial nutrient exchange between trees and fungi (Stinson et al. 2006). Garlic mustard (GM) is an herbaceous invasive species that has invaded the understory of North American forests. GM is a biennial plant that does not form symbiotic interactions with mycorrhizal fungi. GM-invaded soil has been shown to negatively affect AMF colonization of tree seedling roots (Stinson et al. 2006). The effects of GM invasion on EMF colonization are not as well known, but some studies have suggested GM may have deleterious effects on these symbiotic relationships (Wolfe et al. 2008). GM is believed to impact native plants by releasing allelopathic exudates such as flavonoids and glucosinolates (Inderjit et al. 2008, Stinson et al. 2006). The effects on microbial communities are also believed to be mediated through these compounds and breakdown products resulting from their metabolism (Inderjit 2005).

In this study we examined the effect of GM presence on EMF colonization of pine tree roots in sandy soils. Literature suggests mutualistic relationships should be even more important in stressful environments such as nutrient poor sandy soils; therefore, there is great value in evaluating mycorrhizal colonization on this site. Due to the disruptive effect of garlic mustard on ecological associations, it was predicted that sites with GM would have pine roots with decreased EMF colonization and a decreased number of EM fungal tips compared to nonGM sites. It was further predicted that seedlings grown in field soils exposed to GM in a controlled laboratory environment would have decreased EMF colonization as well as a decreased number of EM fungal tips compared to seedlings grown in nonGM soil.

METHODS

Site Description

This study was located in Sand Ridge State Forest (SRSF), a glacial floodplain consisting of alluvial deposits from the late Woodfordian era near Forest City, IL (40.41°N latitude, 89.87° W longitude). Sample sites consisted of red (*Pinus resinosa*) or white (*Pinus strobus*) pine stands established as a plantation in rows in the mid-1940s on former sand prairie with Bloomfield-Plainfield association soils of loamy sand texture. Stands were thinned several years after planting and did not have significant harvests in the intervening years. Red pine and white pine have been found in northern Illinois and are common on well-drained soils (IDNR 2014, NCSU 1998, U.S. Forest Service 2014).

Experimental Approach

Soil samples were collected from 36 GM and 36 nonGM areas at Sand Ridge State Forest in October 2010 and used to grow red pine (*Pinus resinosa*) seedlings. Seedlings were grown in Cone-tainers[™] within environmentally controlled growth chambers for approximately 2 months. Seeds were watered two to three times weekly, as needed, using a Ruakura solution (Smith et al. 1983).

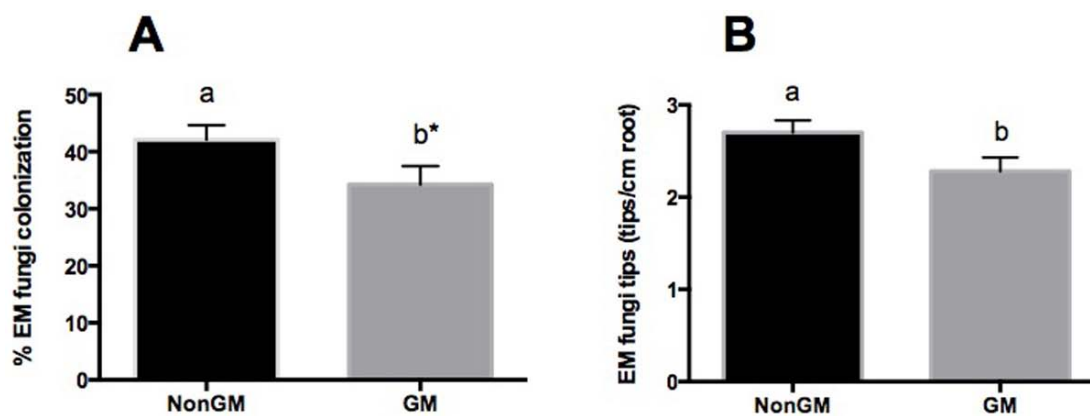


Figure 1.—Percent ectomycorrhizal colonization (A) and number of EM tips/cm of root (B) for seedlings in a growth chamber study grown in nonGM and GM soil ($n_1=34$, $n_2=29$) collected from Sand Ridge State Forest, reported as mean $\pm$ 1 S.E. Significance was determined at $p < 0.05$, *indicates a marginal significant difference, $0.05 \leq p \leq 0.10$.

Pine root samples were collected at random from a known GM ($n=15$) and nonGM site ($n=15$) at Sand Ridge State Forest in April 2012 by using a trowel and collecting roots to a depth of 10 cm. Another sample set was collected from two GM sites ($n=29$) and two nonGM sites ($n=30$) in December 2012 using a soil corer of 5 cm diameter to a depth of 10 cm. Soils were sieved and roots were removed.

Roots collected from the field and from seedling Cone-tainers™ were rinsed after removal. EMF colonization was evaluated morphologically on roots using a dissecting microscope. Root tips were counted, and mycorrhizal and nonmycorrhizal root lengths were measured using a gridded petri dish.

Percent EMF colonization and number of EM tips for GM and nonGM field samples were analyzed using a single-factor ANOVA. Outliers were removed using Grubs Test for Outliers. Assumptions for normality were met, including kurtosis, skewness, and homogeneity of variance. GM and nonGM soil seedling data were analyzed using a two-tailed two-sample t-test. Means denoted by the same letter are not significantly different within each figure. Significance was determined by p-values < 0.05 and marginal significance from $0.05 \leq p \leq 0.10$.

RESULTS

There was a marginally significant decrease in percent EMF colonization for seedling roots grown in GM soil compared to those grown in nonGM soil in the Cone-tainer™ study (Fig. 1). There was also a significantly lower number of EM tips/cm on seedling roots grown in GM soil compared to those grown in nonGM soil (Fig. 1). For field-collected roots, percent EMF colonization on the nonGM sites was significantly higher than for roots collected from the GM sites (Fig. 2). Also, number of EM tips/cm on field-collected pine roots from nonGM sites was significantly higher compared to pine roots collected from GM sites (Fig. 2).

DISCUSSION

We examined the effect of GM presence on EMF colonization of pine tree roots in a pine plantation on sandy soils. As predicted, the pine roots collected from the GM site had a decreased amount of EMF colonization and EM tips/cm compared to those collected from the nonGM site (Fig. 2). These results suggest that the presence of GM has a negative effect on EMF colonization. Studies

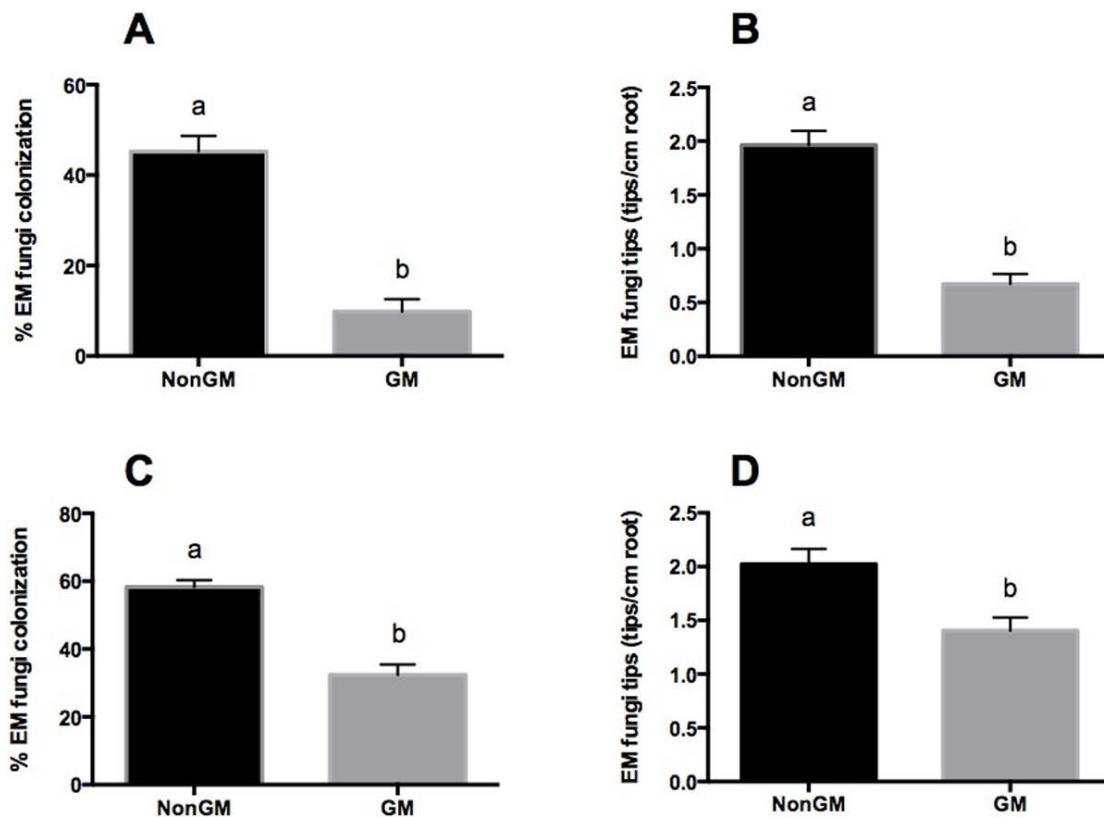


Figure 2.—Percent ectomycorrhizal colonization (A) and number of EM tips/cm of root (B) for field samples collected in April 2012 ($n_1=15$; $n_2=15$) and percent colonization (C) and EM tips/cm (D) for field samples collected in December 2012 ($n_1=30$; $n_2=29$) from nonGM and GM sites at Sand Ridge State Forest, reported as mean ± 1 S.E. Significance was determined at $p < 0.05$.

have shown GM to have detrimental effects on EMF. One study found that the EM fungal root tip biomass was significantly lower in sites that had experienced GM invasions, and for pine seedlings grown in pots containing GM (Wolfe et al. 2008).

Similar results have been found in studies using greenhouse experiments to evaluate ectomycorrhizal inoculum potential. Wolfe et al. (2008) grew pine seedlings in pots containing soil exposed to GM. Again, there was a significant difference in EMF root tip biomass between pines in pots with GM exposed soil and pines in pots with soil not exposed to GM. Similar to Wolfe et al. (2008), our study found decreases in EMF in seedlings grown in GM soil in the greenhouse (Fig. 1). As predicted, seedlings grown in GM soils had a significantly lower, but marginal, percent EMF colonization compared to seedlings grown in nonGM soil (Fig. 1). There were also significantly lower numbers of EMF tips/cm on the pine seedling roots grown in GM soil compared to those grown in nonGM soil (Fig. 1). The results of this study suggest that GM invasion inhibits EMF in the field, and that these findings can be replicated in a controlled greenhouse environment, which can allow for further experimentation on GM and its effects on properties of EMF.

Because many plants rely heavily on their associations with EMF for nutrient acquisition, alterations to mycorrhizal relationships will impact nutrient acquisition and ultimately forest health (Amaranthus and Perry 1994). Decreases in ECM inoculum can also impact ecosystem level processes such as succession and recolonization following disturbance (Allen et al. 1992). Further, the widespread invasion of GM may cause changes to the distribution of EMF and lead to the loss of

EMF and plants that require EMF in GM-invaded areas (Wolfe et al. 2008). Whether these impacts are the result of an overall decline in EMF inoculum or a change in EMF species diversity is unclear. Jones et al. (2003) examined impacts of clearcut logging on EMF decline. They suggested that loss of diversity or inoculum loss impacts productivity and overall forest health. To gain a stronger understanding of the effects of GM on native forest structure and function, further studies should be conducted to determine the mechanism(s) by which GM exerts negative effects on mycorrhizal fungi and the degree to which the impacts are species specific. Increased knowledge of the impacts of GM invasion may provide a better understanding of how to mitigate the negative effects of GM invasion on ecosystems.

ACKNOWLEDGMENTS

The authors wish to thank Brittany Pham for assistance with the manuscript development, Jaclyn Anderson for assistance with data collection, and Kim Lang for oversight on all aspects of our laboratory endeavors. We would also like to thank the Bradley Biology Department for the funding that allowed us to engage undergraduates in research. This research was sponsored by the National Science Foundation grant DEB-0816621 to K.D. McConnaughay and S.J. Morris.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

PREScribed GRAZING FOR MANAGEMENT OF INVASIVE VEGETATION IN A HARDWOOD FOREST UNDERSTORY

Ronald A. Rathfon, Songlin Fei, Jason Tower, Kenneth Andries, and Michael Neary¹

Abstract.—Land managers considering prescribed grazing (PG) face a lack of information on animal stocking rates, timing of grazing, and duration of grazing to achieve desired conditions in natural ecosystems under invasion stress from a variety of nonnative invasive plant (NNIP) species. In this study we tested PG treatments using goats for reducing NNIP brush species and measured impacts to native vegetation after 1 year. The hardwood forest understory was dominated by nonnative multiflora rose (*Rosa multiflora*) and the native spicebush (*Lindera benzoin*). Treatments consisted of two levels of grazing intensity (16 and 48 goats per acre) and two levels of grazing frequency: a single late spring grazing and both late spring and a repeat early fall (October) grazing. All grazing treatments greatly reduced leaf cover of most species of ground layer vegetation at the time of grazing. One year later multiflora rose leaf cover was reduced by an average of 8 to 10 percent from pretreatment cover with no significant differences between grazing treatments. Spicebush cover was reduced by 12 to 16 percent. Although some herbaceous species increased and some decreased under PG treatments, herbaceous species diversity increased slightly overall. Herbaceous cover declined for high stocking rate PG treatments. Multiple years of prescribed grazing may be needed to substantially reduce NNIP cover.

INTRODUCTION

The ecology, impacts on native forest vegetation, and control of many nonnative invasive plant (NNIP) species have received extensive study in recent years. However, research and control efforts in eastern U.S. forests have focused on various combinations of mechanical removal and herbicide treatment. Research on biological controls exclusively refers to insect predators or pathogens of target NNIPs. Prescribed or targeted grazing (PG) is the use of grazing animals as a component in an integrated vegetation management system to achieve certain land management objectives or ecosystem conditions. Although the idea of using animal grazing to manage vegetation is not new, the use of prescribed grazing for managing unwanted NNIP populations is a relatively recent development (Hedtke et al. 2009, Johnstone and Peake 1960, Kleppel and LaBarge 2011, Sharrow et al. 1989). Much of the research and experience using PG for NNIP management comes from western U.S. rangelands. More recently, eastern U.S. researchers and land managers have looked to PG to help manage undesirable vegetation in a wide variety of management contexts, from maintaining power line rights-of-way to managing habitat vegetation for the endangered bog turtle (*Clemmys muhlenbergii*) in New Jersey (Reshetiloff 2011). The development of intensive rotational grazing techniques and mobile fencing systems allows entrepreneurs to provide PG services to landowners (i.e., herd for hire). As such, PG services can provide an alternate source of income for livestock growers. However, such PG services are few in number in the eastern U.S., and research on the effectiveness of PG to suppress NNIP species and impacts to nontarget native vegetation in eastern hardwood ecosystems is almost nonexistent.

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Many factors must be considered when implementing PG for achieving desired ecosystem conditions. The variable feeding behavior of livestock species and breeds makes certain ones better suited for selectively feeding on target vegetation, thus better meeting management objectives. Age, sex, and previous foraging experience may also influence feeding behavior. Domestic goats (*Capra aegagrus hircus*) are well suited to providing prescribed grazing services in hardwood forest environments, particularly where reducing invasive woody species is desired (Hart 2001). The narrow muzzles and tough, dexterous tongues of goats allow them to more efficiently reach and extract leaf tissue from dense, thorny thickets (Campbell and Taylor 2006). They also browse much more than other domestic livestock and are better adapted to extracting nutrients and detoxifying secondary compounds such as tannins and terpenes from woody plants.

In addition to the lack of PG services, land managers face a lack of information on animal stocking rates needed, timing of grazing, and duration of grazing to achieve desired conditions in natural forests under invasion stress from a variety of NNIP species. Information is also needed on how grazing impacts native plant communities so land managers can weigh their options when designing an integrated vegetation management plan.

The objectives of this study were (1) to test grazing intensity (low and high goat stocking rates) and two grazing frequencies within a growing season using goats to reduce a NNIP infestation in a mature hardwood forest understory, and (2) to quantify impacts of those treatments on nontarget native vegetation, including hardwood tree regeneration. This paper reports first-year results.

METHODS

The study was located on a 10-acre site at the Southern Indiana Purdue Agricultural Center (SIPAC) in Dubois County, Indiana. The site was an east-northeast aspect with 30 percent slopes on Gilpin silt loam soils. Yellow-poplar site index at base age 50 was approximately 100 feet. A mature, well-stocked, mixed hardwood forest grew on the site. Average basal area stocking for trees greater than 1 inch diameter at breast height (d.b.h.) was 130 square feet/acre. Overstory tree composition consisted of mature yellow-poplar (*Liriodendron tulipifera*), accounting for 50 percent of the basal area; maple (mostly sugar maple, *Acer saccharum*) and oak (black, *Quercus velutina*; chinkapin, *Quercus muhlenbergii*; northern red, *Quercus rubra*; and white, *Quercus alba*) each accounted for 11 percent of the basal area. Other minor species included sycamore (*Platanus occidentalis*), black walnut (*Juglans nigra*), white ash (*Fraxinus americana*), hickory (*Carya spp.*), and black cherry (*Prunus serotina*). Several exotic invasive shrub and vine species were common in the forest understory; dense, mature multiflora rose (*Rosa multiflora*) (MFR) provided a uniform, nearly impenetrable understory shrub layer throughout most of the stand. Amur honeysuckle (*Lonicera mackii*) (BHS), Japanese honeysuckle (*Lonicera japonica*) (JHS), and autumn olive (*Eleagnus umbellatum*) (AO) also occurred as minor nonnative invasive components in the understory. The native spicebush (*Lindera benzoin*) was also abundant.

Treatments included two levels of grazing intensity (goat stocking) and two levels of grazing frequency. A conventional invasive brush control treatment was included to compare levels of control of target invasive vegetation and levels of damage to nontarget native vegetation. A control in which no disturbance occurred was included as well. The treatments were:

1. Control (C)
2. Manual cutting + herbicide (M+H)
3. Low stocking with two grazings (LS2)
4. Low stocking with one grazing (LS1)
5. High stocking with two grazings (HS2)
6. High stocking with one grazing (HS1)

Treatments were assigned to ¼-acre plots in a completely randomized design, with three replications. In the M+H treatment, target shrubs more than 5 to 6 feet tall were cut using a clearing saw followed by application of a 50 percent solution of Garlon® 3A (31.8 percent acid equivalent triclopyr) to the cut stumps. Target shrubs too small or otherwise missed in the cutting operation were sprayed, using a backpack sprayer, with a 4 percent solution of glyphosate (41 percent a.i.) + ¼ percent non-ionic surfactant to the foliage. Target species included MFR, BHS, AO, JHS, and spicebush. M+H treatment was applied between October 18 and November 2, 2012.

Goats assigned to the project were mature commercial does that were not pregnant and not lactating (no kids). The goats were meat goats from the existing herd at SIPAC and from a Kentucky State University herd. They were a relatively hardy, self-sustaining cross-breed of Boer, Kiko, Savanna, and Spanish influence. Mature does used in the study ranged from 80 to 120 pounds, averaging 90 pounds, and ranged from 2.8 to 3.2 feet in head height. A total of 48 goats were randomly assigned to treatment paddocks (the ¼-acre treatment plots). Paddocks were fenced using portable electric netting. Initial stocking in the first grazing period for the high stocking treatments (HS2 and HS1) was 12 goats per plot (48 goats per acre) and for the low stocking treatments (LS2 and LS1) was 4 goats per plot (16 goats per acre).

Due to the limited number of available goats, only half the treatment plots for the initial grazing period could be grazed at a time. Thus, the first grazing for LS2 and HS2 began on May 9, 2012. The duration of a grazing event at each plot depended on the amount of suitable forage in the plot. Grazing treatments were monitored almost daily, particularly as plots neared being depleted of forage. Goats were moved off of paddocks as they depleted plot forage. Between May 18 and May 21, goats were moved off HS2 paddocks and rotated to HS1 paddocks after 9 to 12 days. On June 11, the goats were moved off LS2 paddocks and rotated to LS1 paddocks after 33 days of grazing. Between May 30 and June 1, goats were removed from HS1 paddocks after 9 to 11 days. The first grazing period ended for all treatment plots on July 2 when goats were removed from LS1 paddocks after 21 days of grazing. The second grazing period was to begin when sufficient regrowth of forage justified returning goats to the paddocks. Severe drought began in early July and lasted through the end of summer. Sufficient regrowth did not materialize until early October and then forage quantity was marginal at best. Goat stocking rates were reduced by half to better match the low amounts of forage available. The second grazing began on October 8 with 6 goats (24 goats/acre) in the high stocking paddocks (HS2) and 2 goats (8 goats/acre) in the low stocking paddocks (LS2). Again, plots were monitored daily and goats were removed when forage was depleted. Goats were removed from HS2 paddocks on October 15 after 7 days of grazing. The second grazing period ended on October 24 when goats were removed from LS2 paddocks after 16 days of grazing.

Within each experimental unit (¼-acre treatment plot), sixteen 6.6 foot by 6.6 foot sample quadrats were permanently marked in a systematic grid on an approximate 26 foot spacing. Foliage cover of ground layer woody vegetation (less than 2 inches d.b.h.) was visually estimated by species as a percent. Height of the tallest point of living foliage was measured for each species of ground layer woody vegetation that had at least one individual that was 1.6 feet tall or taller. Five randomly selected 6.6 foot by 6.6 foot sample quadrats were subsampled within each treatment plot to estimate cover of ground layer herbaceous vegetation. Cover of herbs was visually estimated for each species as the percentage of ground covered by all living tissue. Some species were grouped, such as grasses, sedges, and violets. Each subsample also measured tree reproduction by tallying individual seedlings and saplings by species according to the following size classes:

1. 0 – 1.6 feet
2. 1.6 – 3.3 feet
3. > 3.3 feet, < 2 inches. d.b.h.

All measurements were taken immediately before treatments (May 2012) and 1 year later (May 2013).

ANOVA was used to determine whether measured differences occurred among treatments. Because percent cover data do not meet assumptions of normality, ANOVA was used to test the difference between pretreatment and post-treatment cover estimates. The count data of the tree reproduction tallies were converted to an aggregate height (AH) index combining numbers of seedlings and seedling size in the following formula:

$$AH = 0.25s_1 + 0.75s_2 + 1.5s_3$$

Where AH = aggregate height, $s_{1...3}$ = number of seedlings tallied in each of size classes 1 - 3, respectively. ANOVA then was performed on AH . Duncan's multiple range test was used to separate means where significant differences occurred.

RESULTS

Goats in the high stocking rate treatments (HS1 and HS2, 48 goats per acre) took 9 to 12 days in the spring grazing period to deplete the forage (Table 1). In the low stocking rate treatments (LS1 and LS2, 16 goats per acre), the goats took 21 to 33 days. In the fall grazing period the goats depleted the forage in 7 and 16 days in the high stocking (HS2, 24 goats per acre) and low stocking (LS2, 8 goats per acre) treatment plots, respectively. Severe mid- to late-summer drought prevented vigorous regrowth of shrub layer foliage and stems. Even by early October when the second grazing was deployed, regrowth was still quite low, thus necessitating the lower goat stocking levels. Substantial to nearly complete defoliation of all shrub layer vegetation up to 6 to 7 feet tall was achieved before goats were removed from treatment plots. Defoliation occurred through grazing of leaf blades, browsing of stem growth, and in some cases debarking or breaking of small stems that resulted from horn rubbing. No evidence of feeding on bark was observed, nor did the goats cause any other damage to midstory and overstory trees. The ground layer herbaceous vegetation was also heavily grazed.

Table 1.—Goat stocking and grazing duration in a study testing prescribed grazing for control of invasive brush in a hardwood forest understory

Treatment	1 st Grazing		2 nd Grazing	
	Stocking (goats/acre)	Duration (days)	Stocking (goats/acre)	Duration (days)
LS2	16	33	8	16
LS1	16	21	n/a	n/a
HS2	48	9-12	24	7
HS1	48	9-11	n/a	n/a

Table 2.—Woody plant species diversity before (0) and 1 year following (1) and changes in species distribution following prescribed grazing and conventional mechanical + herbicide treatments in a hardwood forest understory

Treatment	Species diversity				Species distribution change				Cover		
	0	1*	0	1	new	incr.	decr.	no change	0	1 [†]	
	(spp./quadrat)		(tot. spp.)				(spp)		(percent)		
Control	4.9	5.4ns	28	30	3	13	3	12	99.0	101.5	b
M+H	6.7	6.6	34	35	2	14	13	7	68.9	18.1	a
LS2	4.0	4.3	29	26	3	9	12	8	80.8	54.0	ab
LS1	4.5	5.0	28	25	1	12	9	7	102.5	71.8	a
HS2	4.0	4.1	27	25	2	8	12	7	95.7	68.7	ab
HS1	4.6	5.1	30	33	5	15	10	5	96.3	70.4	ab
Overall	4.8	5.1	43	46	4	21	12	10	90.5	64.1	

* Treatment differences tested for the difference in number of spp/quadrat between time 0 and 1. $Pr>F=0.6360$

† Treatment differences are indicated for the difference in cover ($cov_1 - cov_0$) between time 0 and 1. $Pr>F=0.0411$, $\alpha=0.05$.

Woody Shrub Layer Distribution, Cover, and Height

Before treatment, 43 different species of woody plants (trees, shrubs, and vines) occurred in the forest understory; five of those species were nonnative invasive species (Table 2). The mean number of woody species occurring per sample quadrat across the entire study area was 4.8. The most abundant NNIP species, MFR, occurred on 278 of 288 of the sample quadrats, or 96 percent (Table 3). Mean MFR cover ranged from 39 percent to 67 percent (Fig. 1) with overall mean cover of 56 percent across all sample quadrats. Average height ranged between 5 and 6 feet (Fig. 2); some individual MFR reached 20 feet high. JHS distribution ranged from 13 to 44 percent, with an overall mean distribution of 34 percent, but had less than 2 percent cover. BHS distribution ranged from 4 to 21 percent (11 percent overall mean), and AO ranged from 0 to 13 percent (7 percent overall mean). BHS and AO each had 1 percent or less of overall mean cover.

The most abundant and widely distributed native understory woody plant species was spicebush, occurring in 69 percent of sample quadrats (Table 3) and ranging from 11 to 27 percent cover (Fig. 3) with a mean of 19.7 percent cover overall. Spicebush mean height ranged from 4.8 to 7.1 feet (Fig. 4) with an overall mean height of 6.2 feet. The next most widely distributed native understory woody plant species were Virginia creeper (*Parthenocissus quinquefolia*) (67 percent), sugar maple (36 percent), and hydrangea (*Hydrangea arborescens*) (35 percent). However, all other native woody species each provided 4 percent or less cover.

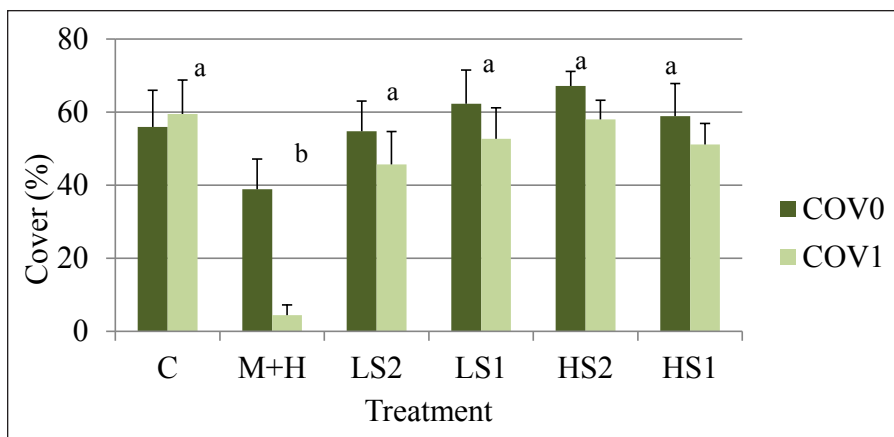


Figure 1.—Percentage of ground cover for multiflora rose growing in plots before (COV0) and 1 year after (COV1) prescribed grazing and conventional mechanical + herbicide treatments. Change in cover (COV1 – COV0) for treatments marked with the same letter is not significantly different from one another using ANOVA, $P > F = 0.0043$, $\alpha < 0.05$.

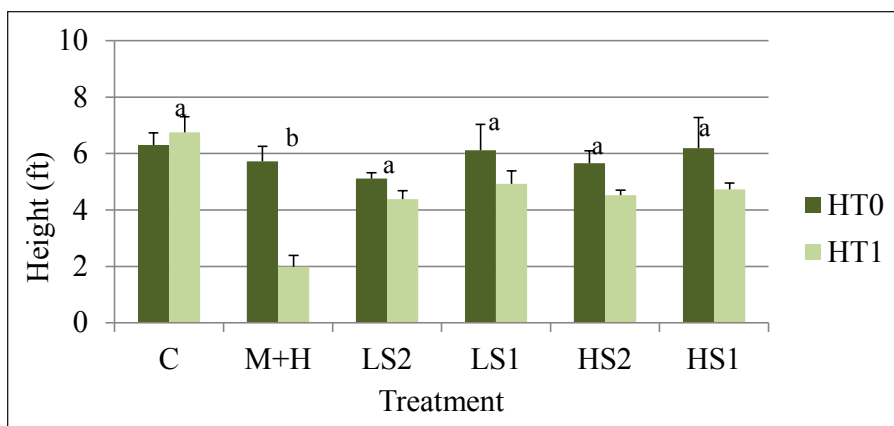


Figure 2.—Multiflora rose height in plots before (HT0) and 1 year after (HT1) prescribed grazing and conventional mechanical + herbicide treatments. Change in height (HT0 – HT1) for treatments marked with the same letter is not significantly different from one another using ANOVA, $P > F = 0.035$, $\alpha < 0.05$.

One year after treatment, 46 woody plant species were recorded across all sample quadrats, for a net gain of 3 (Table 2). All but one treatment, M+H, increased slightly in woody species diversity. Neither the M+H nor any of the PG treatments were significantly different from the control. Few new woody species were tallied after the first growing season (Table 2). Twenty-one species increased their overall distribution, 12 decreased, and 10 did not change. The control had the fewest species decreasing in distribution. The PG treatments with two grazing periods in the growing season (LS2 and HS2) had fewer woody species increasing and more species decreasing their distribution than PG treatments with only one grazing period (LS1 and HS1).

While MFR distribution increased by 1 sample quadrat in the control, it declined by 15 quadrats, or 31 percent, in M+H (Table 3). There was no change in MFR distribution in any of the PG treatments. JHS distribution increased by 8 percent in the control and decreased by 6 percent in M+H. There was virtually no change in JHS distribution among the PG treatments. Little change occurred in distribution of BHS and AO across all treatments, except a 6 percent increase in BHS in the control and a 7 percent increase in the LS2 PG treatment. Overall distribution of spicebush decreased to 65.3 percent, a 3.5 percent decline, with small decreases occurring in every treatment. Although not widely distributed, pawpaw (*Asimina triloba*), where it occurred, was not grazed or browsed at all. Some stem debarking and breaking did occur, however.

Table 3.—Frequency distribution as percent of sample quadrats occupied by the most widely distributed woody and herbaceous species in a hardwood forest understory before (0) and 1 year following (1) prescribed grazing and conventional mechanical + herbicide treatments

Mechanized grazing and conventional mechanical herbicide treatments												
Species	Treatment											
	Control		M+H		LS2		LS1		HS2		HS1	
	Time Relative to Treatment (year)											
	0	1	0	1	0	1	0	1	0	1	0	1
Woody:	----- percent -----											
nonnative												
multiflora rose	94	96	88	67	96	96	96	96	98	98	98	98
Jap. honeysuckle	38	46	27	21	44	44	17	17	33	31	13	15
Amur honeysuckle	21	27	8	8	8	15	4	2	8	6	4	2
autumn olive	10	10	13	10	4	4	4	4	0	0	6	4
native												
spicebush	73	69	56	54	60	54	92	90	67	63	65	63
Virginia creeper	73	87	87	100	53	47	53	60	60	60	73	67
maple	31	31	69	69	27	27	31	23	31	25	35	40
hydrangea	8	19	42	44	10	8	69	73	25	31	58	63
elm	19	15	33	38	23	25	19	27	21	19	23	31
Herbaceous:												
sedge	87	93	93	93	73	87	60	73	60	73	80	93
violet	80	73	93	100	73	80	87	87	100	87	93	87
jewelweed	87	87	67	73	53	73	60	67	73	87	67	80
white snakeroot	73	87	87	100	40	53	60	53	67	60	60	67
Christmas fern	60	60	60	60	53	47	80	73	67	67	60	60
white avens	67	67	53	53	47	53	47	53	67	53	13	60
grass	73	53	47	40	73	80	40	33	53	53	53	67
jack-in-the-pulpit	53	67	27	40	47	53	27	40	40	40	40	33

The control had practically no change in MFR cover or height (Figs. 1 and 2). Only M+H resulted in large cover and height reductions, from 39 percent pretreatment to 4 percent post-treatment cover, and 3.9 feet in mean height reduction 1 year later. The PG treatments initially resulted in near complete defoliation of MFR immediately after treatment. By the following spring, only modest MFR cover and height reductions were evident. Those reductions ranged from 7.8 to 10 percent for cover and from 0.8 to 1.4 feet for height. Neither cover nor height reductions for PG treatments were significantly different from each other or from the control. JHS, BHS, and AO cover were all low before treatment. PG treatments reduced cover of these minor NNIP species, but not significantly.

Spicebush cover was reduced by 12 to 16 percent and height reductions ranged from 1.6 to 3.1 feet (Figs. 3 and 4) for the PG treatments. M+H reduced spicebush cover by 8 percent and height by 2.6 feet. Although mean cover reduction for PG treatments and M+H were all greater than for the control, no differences were significant, suggesting large amounts of variation. LS2 and HS1, along with M+H, reduced height significantly from the control.

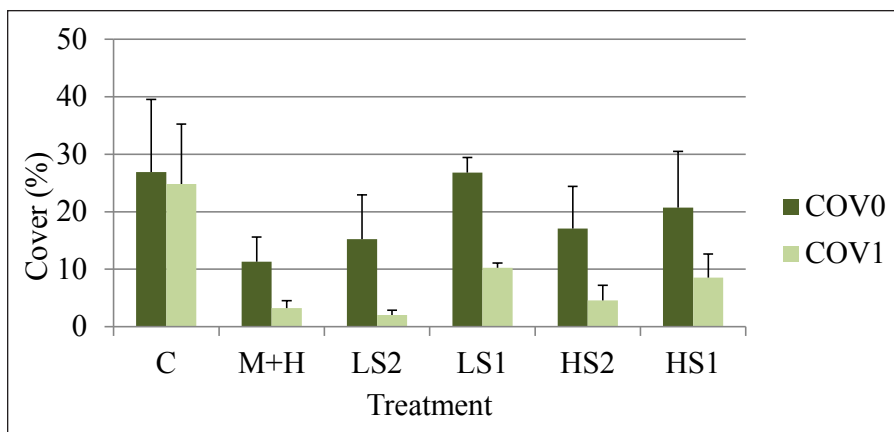


Figure 3.—Percentage of ground cover for spicebush growing in plots before (COV0) and 1 year after (COV1) prescribed grazing and conventional mechanical + herbicide treatments. Change in cover (COV1 – COV0) for treatments marked with the same letter is not significantly different from one another using ANOVA, $P > F = 0.0043$, $\alpha < 0.05$.

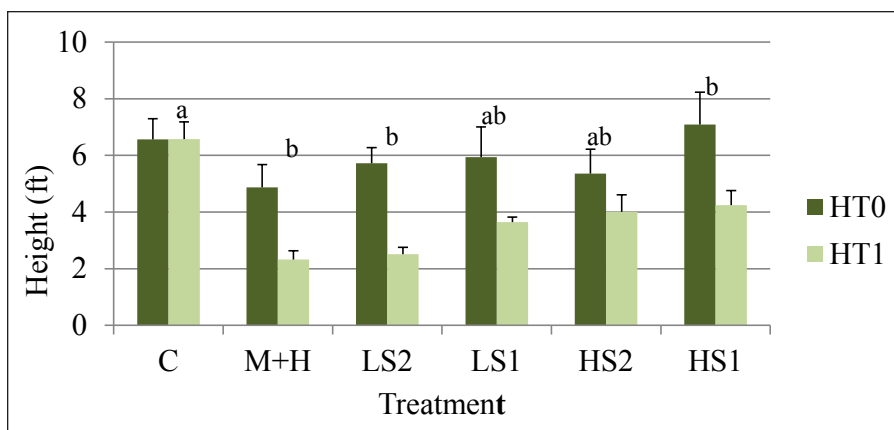


Figure 4.—Spicebush height in plots before (HT0) and 1 year after (HT1) prescribed grazing and conventional mechanical + herbicide treatments. Change in height (HT0 – HT1) for treatments marked with the same letter is not significantly different from one another using ANOVA, $P > F = 0.035$, $\alpha < 0.05$.

Herbaceous Plant Diversity, Distribution, and Cover

Before treatment, 78 different herbaceous plant species occurred across all sample quadrats (Table 4). The mean number of herbaceous species occurring on each sample quadrat across the entire study area was 10.4. Total herbaceous cover ranged from 35 to 55 percent with an overall average of 46 percent. The most widely distributed herbaceous understory plants were violet species (*Viola spp.*), occurring on 88 percent of all sample quadrats. Other widely distributed species were sedge species (*Carex spp.*) (76 percent), jewelweed (*Impatiens capensis*) (68 percent), white snakeroot (*Eupatorium rugosum*) (64 percent), Christmas fern (*Polystichum acrostichoides*) (63 percent), grass species (57 percent), and white avens (*Geum canadense*) (49 percent). Violet species had the highest mean cover for an herbaceous species group with 5.4 percent. White snakeroot, grass species, and sedge species had 4.7, 4.2, and 3.5 percent cover, respectively.

One year after treatment, 86 herbaceous plant species were recorded across all sample quadrats (Table 4). All but one treatment (LS1) increased slightly in herbaceous species diversity. Neither the M+H nor any of the PG treatments were significantly different from the control.

Overall, 18 new herbaceous species were tallied after the first year of treatment. Thirty-two species increased their distribution (Table 4), five of them by 10 percent or more. Thirteen species did not change; 33 species decreased in distribution, the vast majority by less than 5 percent. There were no clear patterns in species distribution changes among the treatments over the first year (Table 4). The

Table 4.—Herbaceous plant species diversity, changes in species distribution, and cover before (0) and 1 year following (1) prescribed grazing and conventional mechanical + herbicide treatments in a hardwood forest understory

Treatment	Species diversity				Species distribution change				Cover	
	0	1*	0	1	new	incr.	decr.	no change	0	1†
	(spp./quadrat)		(tot. spp.)			(spp)			(percent)	
Control	10.5	10.7 ns	30	35	9	8	12	10	34.5	28.3 c
M+H	10.1	11.4	33	37	10	13	12	8	51.6	37.0 abc
LS2	10.8	12.5	43	46	10	16	13	14	41.3	26.4 abc
LS1	10.1	10.1	36	43	12	9	13	14	38.5	28.7 bc
HS2	10.3	11.8	34	40	11	11	11	12	52.6	24.3 a
HS1	10.8	12.5	40	49	15	11	15	14	55.3	31.3 ab
Overall	10.4	11.5	78	86	18	32	33	13	45.6	29.3

* Treatment differences tested for the difference in number of spp/quadrat between time 0 and 1. $Pr>F=0.3078$.

† Treatment differences are indicated for the difference in cover ($cov_1 - cov_0$) between time 0 and 1. $Pr>F=0.0532$, $\alpha=0.05$.

control appeared to have slightly fewer new species observed and slightly fewer species that increased in distribution than occurred in other treatments. M+H appeared to have slightly fewer species that had no change in distribution than all other treatments.

Some patterns in changes in individual herbaceous species distribution began to appear in the first year among the most widely distributed species (Table 3). Sedges appeared to increase their distribution in PG treatments. Jewelweed remained unchanged in the control but increased irrespective of disturbance treatment. Violet species and jack-in-the-pulpit (*Arisaema triphyllum*), on the other hand, increased their distribution in M+H, increased or remained unchanged in the low stocking PG treatments but declined in the high stocking PG treatments. Christmas fern remained stable in all treatments except the low stocking PG treatments where declines in distribution occurred. Although not widely distributed, ginger (*Asarum canadense*) appeared to be one of the few herbaceous species completely avoided by goats.

Herbaceous cover declined for all treatments (Table 4). The control declined by 6 percent and the M+H treatment declined by 15 percent. Herbaceous cover declined by 10 to 15 percent for the low stocking PG treatment. Only the high stocking rate PG treatments reduced herbaceous cover significantly from that of the control, by 24 (HS1) and 28 (HS2) percent.

Tree Reproduction

Before treatment, 3,912 small (<1.6 feet), 292 medium (1.6 to 3.3 feet), and 135 large (3.3 feet to 2 inches d.b.h.) tree reproduction stems per acre (Table 5) grew across the study site. The most abundant regeneration species were the maples, primarily sugar maple, with more than 1,664 total stems per acre, accounting for 38 percent of advance regeneration stems. Elm (*Ulmus spp.*), black cherry, and white ash made up more than 641, 573, and 472 stems per acre, respectively. Aggregate height for total tree advance regeneration ranged from 2,268 to 7,082 feet/acre.

The control had a 606 stems per acre decline in total advance regeneration. M+H reduced total advance regeneration by 135 stems per acre. Total advance regeneration loss of PG treatments ranged

Table 5.—Tree reproduction stocking by three size classes before (0) and 1 year following (1) and regeneration aggregate height reduction following prescribed grazing and conventional mechanical + herbicide treatments

Treatment	Size class						Aggr. Ht.* Reduction
	<1.6 feet		1.6 – 3.3 feet		3.3 feet – 2 inches d.b.h.		
	Time relative to treatment						
	0	1	0	1	0	1	
	-----stems/acre -----						(ft./ac)
Control	4,249	3,440	67	337	67	0	332 ns
M+H	5,801	6,070	540	270	202	67	1,107
LS2	3,777	2,833	472	270	472	270	2,545
LS1	3,979	2,023	0	0	0	0	1,604
HS2	2,158	1,619	202	67	0	0	775
HS1	3,575	3,035	472	337	67	67	775
Overall	3,912	3,170	292	214	135	67	1,134

* Treatment differences tested for the difference in aggregate height between time 0 and 1. $Pr > F = 0.7328$.

from 674 to 1,956 stems per acre; the low stocking rate treatments had the largest losses on average. Average aggregate height declined across all treatments and the control. Aggregate height reduction for the control was 332 feet/acre. M+H had a 1,107 feet/acre reduction. Aggregate heights were reduced by 775 feet/acre for each of the high stocking rate PG treatments. The low stocking rate PG treatments reduced aggregate height by 1,604 and 2,545 feet/acre. However, because of high variability in the data, no differences between treatments were significant.

DISCUSSION

The high stocking treatments used three times the number of goats as the low stocking treatment, yet low stocking treatments took from 1.8 to 3.7 times longer in the summer grazing, or an average of 2.75, and 2.3 times longer in the fall grazing compared to high stocking treatments, to deplete plot forage. Variability of total pretreatment forage quantity between treatment plots may account for some of the variability in time to depletion of forage and the phenomenon of the low stocking treatment requiring, on average, less than three times the amount of time to deplete plot forage. Although efforts were made to remove goats from plots based on visual depletion of forage as judged from daily inspections, it is also possible that subjectivity in judging forage depletion resulted in goats being removed from some plots before complete forage depletion and some being removed one to several days beyond complete forage depletion. Yet another potential explanation for this apparent incongruity in forage depletion rates between high and low stocking PG treatments may have been a more rapid loss of grazing efficiency in high stocking rate treatments as forage was nearing depletion—a sort of diminishing returns for effort expended by the goats to extract forage that was increasingly difficult to access. Goats in this circumstance may have had less than optimal nutrition the last few days before removal. For example, the most palatable and easily accessed forage—forage acquired with the least amount of physical effort—was consumed first. In the case of MFR, the newest, most tender, and succulent leaves and shoot tips located at the goats' head level were first grazed and browsed. When that was gone, the goats would stretch their necks a little higher. Finally, expending more energy, goats would stand on hind legs to reach more woody stems or work harder to penetrate to the interior of MFR thickets to graze older interior leaves. This point of diminishing

returns may be reached more quickly, and before managers decide the forage is depleted, under high stocking conditions. High intensity grazing using high stocking rates for short durations are frequently recommended for controlling invasive brush infestations in western U.S. rangelands (Campbell and Taylor 2006).

Visual impacts immediately after PG treatments showed heavy to near complete defoliation of woody plants up to 6 to 7 feet high and severe reductions in herbaceous plant cover for most species. PG had a somewhat muted effect, however, on MFR cover and height 1 year after initiation of treatments and had no impact on distribution. Additional reductions in woody cover and height occurred through goat debarking or girdling of stems by horn rubbing, particularly on spicebush stems. Large variations in height reduction occurred between replicate plots within treatments. Manually cutting shrubs more than 8 feet high and the random girdling of stems by goats rubbing their horns may have confounded first-year data, particularly for spicebush.

Although reductions in MFR and other NNIP species cover and height under PG treatments were not statistically different from the control, they may indicate a trend beginning to develop in the first year toward future reductions. Followup multiple grazings in one growing season, if applied soon after sufficient regrowth from the first grazing, should significantly amplify the weakening effect of PG on target vegetation (Olson and Launchbaugh 2006). However, no real differences in woody plant cover or height occurred between PG treatments for the target brush species. Sufficient vegetative regrowth did not occur until early October due to severe drought. Thus, the second grazing periods for LS2 and HS2 treatments were delayed until early fall. The fall grazing may have had minimal effect on plant health. Animal stocking rate did not influence MFR, other NNIP species, or spicebush cover and height in the first year.

Species diversity increased for both woody and herbaceous plant species across most treatments and the control. Although there were no differences between any of the treatments, properly applied PG appeared to promote, or at worst, not diminish species diversity in the first year, even where multiple grazings occurred. However, there were shifts in distribution, with some species increasing and some decreasing. Overall, these increases and decreases seemed to balance each other, especially for herbaceous species. Where two grazings occurred in the growing season (LS2 and HS2), more woody species declined in their distribution than increased. Because a similar pattern did not emerge for herbaceous species, we might conclude that these treatments may have had a disproportionately negative impact on woody species. Considering that the second grazing occurred in the fall after a severe drought, it is likely that many herbaceous species had already completed their annual life cycle leaving relatively little herbaceous vegetation available as forage.

On this site, well-distributed species like the violets and jack-in-the-pulpit decreased in their distribution under the high stocking rate treatments. These and similar species may serve as early indicators of grazing intensity that exceeds optimal levels needed to reduce target plants while minimizing negative impacts to nontarget plants (Hendrickson and Olsen 2006). High stocking rate treatments seemed to disproportionately decrease herbaceous cover compared to low stocking rate treatments but did not differ from low stocking rates in its reduction of woody vegetation cover. The combination of far less herbaceous forage being available compared to woody forage and herd behavior changes at high stocking rates may have reduced grazing selectivity (Adler et al. 2001).

Goats are considered generalists in their feeding behavior. However, some selectivity occurs depending on palatability and the individual animal's previous experience with different plants. In this study few plant species within reach were not ultimately fed upon. Notable exceptions included pawpaw and wild ginger.

Selective feeding by sheep reduced competing vegetation, red alder (*Alnus rubra*) in particular, in coastal Oregon Douglas-fir (*Pseudotsuga menziesii*) plantations, resulting in increased tree height and d.b.h. compared to non-grazed plantations (Sharrow et al. 1992). PG with Scottish Highland cattle was used to help restore an oak savannah community in Wisconsin (Harrington and Kathol 2009) where woody stem densities declined by 44 percent under grazing. In this case management objectives called for the reduction of woody shrub layer vegetation to benefit herbaceous species. In Central Hardwood forest management, PG objectives call for the selective grazing/browsing of competing shrubs and herbs to benefit desirable tree reproduction. In this study, tree advance regeneration was grazed along with other woody brush. Low stocking rate treatments appeared to produce larger, yet not significant, aggregate height reductions in tree regeneration than high stocking rate treatments. The M+H allowed greater selectivity to encourage desirable regeneration.

An immediate benefit of PG in dense nonnative brush infestations was the significant reduction of both the visual and physical barriers to applying conventional mechanical and herbicide treatments. PG cleared areas between large shrubs, which could allow workers much easier access to much of the area that was largely inaccessible due to thorny, dense brush. The reductions in leaf cover and height would also reduce herbicide application costs.

CONCLUSIONS

First-year results show only modest reductions in target NNIP cover and height. PG will likely need to occur over several years to gradually reduce target species and improve conditions for desirable species to recruit and gain the competitive advantage. Results from this research thus far are preliminary. Long-term results are needed to determine whether and how PG can be deployed to selectively and substantially reduce nonnative invasive woody vegetation to favor desirable native species in hardwood forest environments. PG alone will likely be insufficient to eradicate nonnative invasive infestations but may be used in combination with conventional mechanical and herbicide treatments to reduce costs and chemical inputs into the environment.

ACKNOWLEDGEMENTS

This research was partially funded through a Mary S. Rice grant administered through the Purdue Research Foundation.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

WOODY BIOMASS

EVALUATION OF TOTAL ABOVEGROUND BIOMASS AND TOTAL MERCHANTABLE BIOMASS IN MISSOURI

Michael E. Goerndt, David R. Larsen, and Charles D. Keating¹

Abstract.—In recent years, the state of Missouri has been converting to biomass weight rather than volume as the standard measurement of wood for buying and selling sawtimber. Therefore, there is a need to identify accurate and precise methods of estimating whole tree biomass and merchantable biomass of harvested trees as well as total standing biomass of live timber for resource assessments and silvicultural planning. In this study, we compared the traditional whole tree diameter-based biomass model currently used with alternative model forms fitted to tree data collected from four southeast Missouri species. Additionally, we reassessed each nonlinear model with total tree height and crown ratio included as covariates. Finally, we assessed the best model identified from the aforementioned analyses for estimation of merchantable biomass. Results of the analysis yielded several nonlinear models for estimating aboveground tree biomass with relatively high precision and low bias. The optimal model was chosen based upon precision and bias of estimation for all four species and was shown to produce precise estimates of merchantable biomass as well as total aboveground biomass for each species.

INTRODUCTION

The calculation of tree biomass is a fairly new concept to the forest products industry in Missouri. Traditionally, the industry has focused on volume estimation, generally in scaled board feet. In the late 2000s, a number of industry pressures pushed a change to buying and selling wood by weight. This change is common in the forest industry nationwide. Additionally there has been an interest in the estimation of carbon content of wood in standing and harvested trees. The equations generally used for biomass and carbon estimation are less precise than those used to estimate volume in Missouri forests. This disparity in precision is primarily an artifact of exclusive use of diameter at breast height (d.b.h.) as a covariate in biomass equations combined with regression coefficient estimates derived from sampled trees in other regions of the United States. The importance of biomass/weight estimation for the Missouri forest products industry highlights a need to derive more precise methods of estimating biomass for Missouri tree species.

Currently the most common method for estimating total aboveground tree biomass is the diameter-based nonlinear model provided by Jenkins et al. (2003). One advantage to this model is that it provides estimates of biomass using only one covariate (d.b.h.). This means that biomass estimates can be derived with minimal effort and cost to forest managers and loggers alike. However, the generalized nature of this method can often lead to inflated estimates and low precision of estimation for individual species. Additionally, most Missouri hardwood species utilize the same set of coefficient estimates based on the hardwood species grouping conducted by Jenkins et al. (2003), which reduces the flexibility of biomass estimation between individual hardwood species and species groups found in Missouri.

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Table 1.—Summary statistics by species for sampled trees

Species	N	d.b.h. (cm)	Total Height (m)	Merchantable Height (m)	Total Weight		Merchantable Weight	
					Green (kg)	Dry (kg)	Green (kg)	Dry (kg)
Hickory	33	29.6	16.9	8.2	847.7	548.3	437.5	281.7
White oak	60	34.9	17.8	8.3	1271.9	809.7	630.3	400.7
Black oak	63	35.8	17.5	9.4	1249.4	743.6	771.6	457.3
Post oak	59	33.8	15.5	6.6	934.6	591.5	487.2	307.5

In this study we collected a local data set of total aboveground biomass and merchantable biomass to evaluate the published equation (Jenkins et al. 2003) commonly used to estimate biomass in Missouri. We subsequently used the same data set to refit the model and compare it to several other common model forms for estimation of aboveground tree biomass. Due to the time and expense of collecting biomass data sets, relatively few studies have been conducted that compare biomass estimation methods using covariates other than d.b.h. Therefore, we analyzed the effect of introducing height and crown ratio as additional covariates into each nonlinear model form. Finally, we refit the recommended model from the aforementioned analyses for estimation of merchantable biomass/weight using both d.b.h. and merchantable height as covariates. The last stage of the analysis was particularly important for gauging the usefulness of our recommended biomass model for the Missouri forest products industry.

METHODS

This study is part of a larger biomass harvesting project conducted at the University of Missouri in the School of Natural Resources in collaboration with the Missouri Forest Product Association and the U.S. Forest Service, State and Private Forestry. The Missouri Forest Products Association assisted in identifying a suitable site for sampling 220 trees with a diameter at breast height between 8 and 30 inches. The sample included at least 50 trees in each of four sample groups: white oak (*Quercus alba* L.), black oak (*Quercus velutina* Lam.), post oak (*Quercus stellata* Wangenh.), and hickories (*Carya* spp.). Attempts were made to sample evenly across diameter classes from the minimum tree size up to the largest tree found at the site. Table 1 lists the summary statistics of the sample trees.

Each sample tree was marked and d.b.h., total tree height, and crown ratio were measured on standing trees. Trees were then felled by a professional master logger, who cut the stump as close to the ground as could be safely accomplished. The whole tree with tops and leaves was skidded onto the road for further processing. The Missouri Forest Products Association obtained the use of a Volvo™ front-end loader with a load cell so the entire aboveground portion of the tree could be weighed at once (Fig. 1). The operator was careful to assure full suspension and minimal movement during the measurement.

After the whole tree was measured, branches and leaves were removed and the total merchantable portion of the stem was weighed. If the tree was bucked further at the logger's preference, we weighed each log as well. A disk from the bottom end of each log was removed and weighed green in the field using an electronic scale. These disks were used to obtain moisture content on the day of felling and oven-dry weight for each tree. Oven-dry weight for total biomass and merchantable biomass was estimated using the average moisture content measured from the individual disks cut from each tree. Because of logistics, only one site was sampled near Potosi, MO. Specific gravity was determined using standard methods (Bowyer et al. 2003).



Figure 1.—Front-end loader with a tree fully suspended for weighting.

ANALYSIS

The motivation for this study was to evaluate published biomass equations for use in Missouri and to then use a variety of model forms to produce equations to predict whole tree aboveground biomass and merchantable biomass for the sampled trees. To start the analysis we used the Jenkins et al. (2003) equation to predict aboveground whole tree biomass for each tree and compared it to the weight measurements in the field.

Many of the widely-used equations for aboveground biomass of U.S. trees use diameter as the sole tree measurement for estimation. Aboveground biomass equations have been developed in both linear and nonlinear forms, greatly dependent upon intended scale of use and the combination of region and tree species for which it was derived.

Common National-scale Model Form

Previous work by Jenkins et al. (2003) yielded a set of generalized allometric regression equations for estimating total tree biomass using tree inventory data for U.S. forests at the national scale. One of the most widely used aboveground woody biomass equations has the following form (Jenkins et al. 2003, Jenkins et al. 2004):

$$bm = \exp(\beta_0 + \beta_1 \ln d.b.h.)$$

where

bm = total aboveground biomass (kg) for trees 2.5 cm and larger in d.b.h.

d.b.h. = diameter at breast height (cm)

exp = exponential function

ln = natural logarithm.

The first step in our analysis was to estimate aboveground biomass separately for each of the four species using this equation. We assessed each species separately throughout the analysis in order to compare the performance of all models between species and to assess any significant changes to estimates of aboveground biomass with the inclusion of additional variables such as height and crown ratio. Because our species of interest only included hickory and oak species, the aboveground biomass equation used the same regression coefficients for all four species: $\beta_0 = -2.0127$ and $\beta_1 = 2.4342$ (from Table 4 in Jenkins et al. 2003).

Comparison with Other Common Model Forms

The first step to determining an appropriate model form for estimating aboveground biomass for southeast Missouri hardwood species was to compare the model form used by Jenkins et al. (2003) to several other common model forms within the United States. We ultimately compared six model forms using d.b.h. as a covariate. These included three nonlinear model forms in addition to Jenkins et al. (2003) and two linear model forms. Many of the alternative models did not have coefficient estimates available for the species of interest in our study. Therefore, a comparison based on existing coefficient estimates for each model was not possible. Instead, we fit each model to our collected tree data using the R statistical package (R Foundation for Statistical Computing, Vienna, Austria). For all analyses we fit the models separately for each species, thereby deriving species-specific coefficient estimates for each model. Resulting models were validated and compared using summary statistics of precision and bias as well as residual plots to identify species- and model-specific trends in estimation of aboveground tree biomass.

Influence of Height Measurements in Nonlinear Model Forms

A common characteristic of many aboveground tree biomass models is the exclusive use of d.b.h. as a covariate. Recall that the initial comparison of alternative models described in the previous section preserved this characteristic to assess the prediction capabilities of the models when fit specifically to tree measurements for the four species of interest. One of the objectives of this study was to analyze the influence of height on prediction of aboveground tree biomass when included in the models. We were particularly interested in assessing the influence of height as a multiplier for d.b.h. squared (dbh^2). Our interest in this use of height comes from the similar use of height in many traditional volume equations based on the concept of a simplified cylindrical measurement of the tree stem. One of the most common volume equations using this concept takes the following form (Hahn 1984):

$$V = \beta_0 + \beta_1 * dbh^2 * ht$$

Where

V = gross volume

ht = merchantable height (m).

We know from past studies and analyses of forest data, such as from U.S. Forest Inventory and Analysis (FIA), that aboveground forest biomass is highly correlated with aboveground volume (Chojnacky 2012, Goerndt et al. 2012). Therefore, in all cases where the original aboveground biomass model included a covariate of dbh^2 , we included total tree height as a multiplier to dbh^2 . Due to issues of overlapping model form between alternative models, we added height as an additional covariate for models that did not initially include dbh^2 .

Effects of Crown Ratio

The primary reason for assessing the influence of crown characteristics in aboveground biomass models was to detect differences in model fit between the four species of interest. This is particularly important from a stand dynamics aspect as some hardwood species (e.g., post oak) tend to reach a height apex at a relatively young age. They can, therefore, produce trees of varying age and specific gravity with only moderate variation in both d.b.h. and height, depending upon crown closure and competition within the stand. Therefore, following the inclusion of total height into the biomass models, crown ratio was included as an additional covariate to test for significant model effects and to assess any noticeable changes in prediction of aboveground biomass between the different species. The crown ratio metric used for this study was calculated as follows:

$$CR = \frac{ht}{HCB}$$

where CR = crown ratio (%), HCB = height to crown base, and ht = total tree height (m).

Merchantable Biomass Estimation

Analyzing the influence of height in the estimation of total aboveground biomass provided valuable information as to which model form was most optimal for estimating aboveground biomass across the four species in this study. In order to expand usability of this model form to merchantable biomass, the model form was refit for each species using merchantable height and d.b.h. to estimate merchantable biomass weight. Although merchantable height can be approximated from a ground measurement on standing trees, in this analysis we calculated merchantable height as a summation of the lengths of merchantable logs cut from each tree. The observed merchantable biomass weights used to fit the models were derived from the summation of green weights of merchantable logs per tree and adjusted using the average moisture content (%) by species as with total aboveground dry biomass.

Model Validation

Each model was validated by using summary statistics that were calculated based upon the species-level validation and included relative root mean squared error (RRMSE) and relative bias (RB) calculated as follows:

$$RRMSE_s = \frac{\sqrt{\frac{1}{R} \sum_{r=1}^R (\hat{Y}_{iP} - \hat{Y}_{iO})^2}}{\bar{Y}_{iO}}$$

where

$RRMSE_i$ = the relative root mean squared error for species s ,

R is the number of trees sampled for species s ,

$\hat{Y}_{iP}$ is the predicted aboveground biomass for tree i , and

$\hat{Y}_{iO}$ is the observed aboveground biomass for county i .

Similarly, overall bias was assessed using relative bias for species s calculated as:

$$RB_s = \frac{1}{R} \sum_{r=1}^R \left(\frac{\hat{Y}_{iP} - \hat{Y}_{iO}}{\bar{Y}_{iO}} \right)$$

In addition to the aforementioned summary statistics, models were also validated and compared using residual plots. This enabled us to visually assess prediction bias and trends in prediction as observed values increased, and to see outliers which may affect model fit and coefficient estimation.

RESULTS AND DISCUSSION

Common Diameter-based Nonlinear Model Form

Recall that the coefficients applied to the common aboveground biomass model and obtained from the Jenkins paper (see Table 4 in Jenkins et al. 2003) were $\beta_0 = -2.0127$ and $\beta_1 = 2.4342$. Predicted values from this model produced RRMSE estimates of 27.9 for hickory, 37.2 for white oak, 41.2 for black oak, and 69.9 for post oak. This model also produced relative bias (RB) estimates of 1.5 for hickory, 8.9 for white oak, 25.7 for black oak, and 36.1 for post oak. The summary statistics reflected a relatively low level of precision and relatively high bias using the original model, particularly for black oak and post oak. Error associated with these estimates is better understood by observing residual plots (Fig. 2).

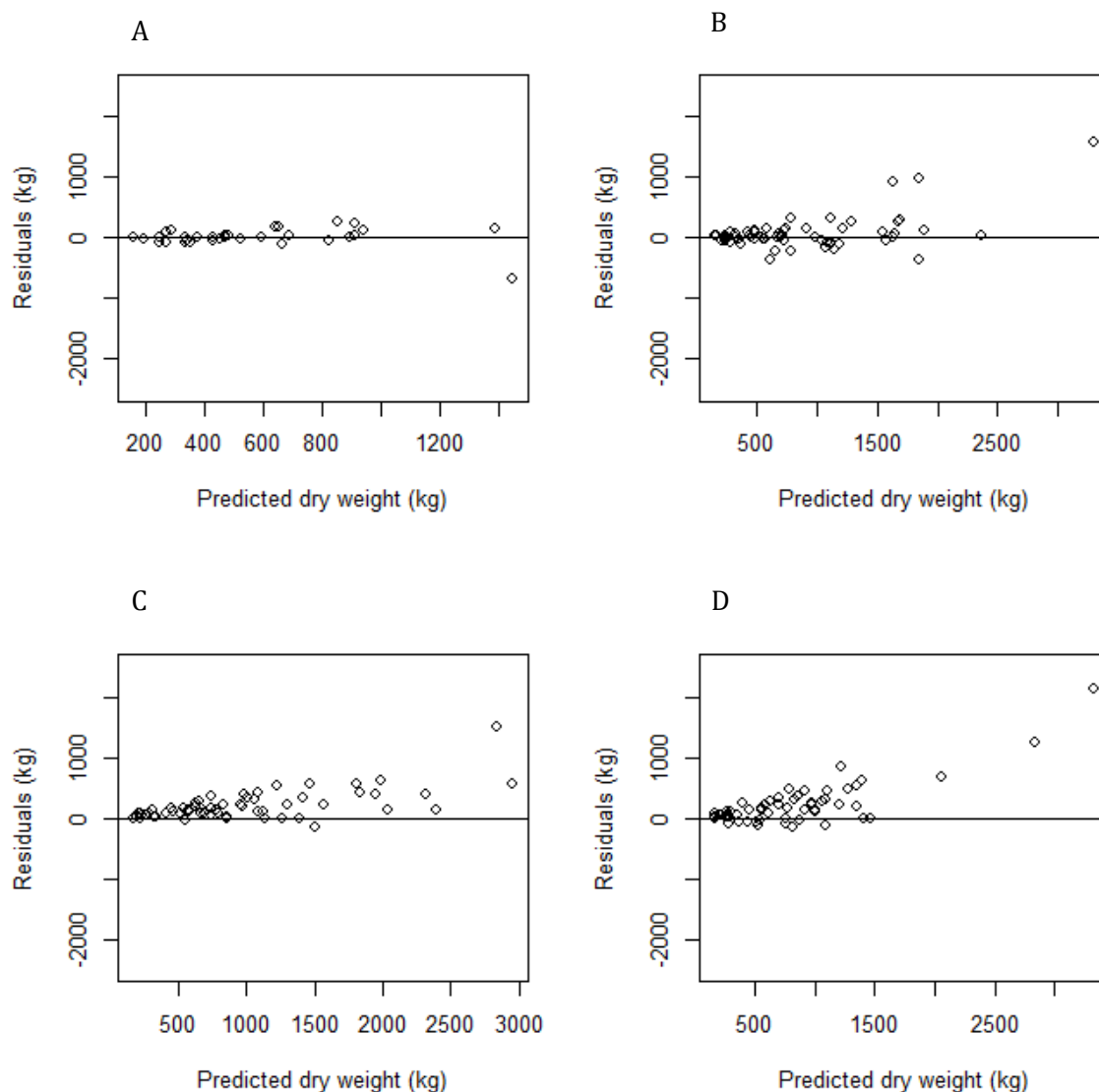


Figure 2.—Residuals of prediction of aboveground biomass by species using model form and hardwood coefficient estimates from Jenkins et al. (2003) for hickory (A), white oak (B), black oak (C), and post oak (D).

The original model for hardwoods from Jenkins et al. (2003) has a tendency to overestimate aboveground biomass for the four species of interest. This effect was least noticeable for hickory, which does not show obvious overestimation until about the 75th quartile of predicted biomass values. For black oak and post oak, overestimation begins at very low predicted values and increases as the predicted value increases. In addition to a lack of sensitivity to the individual hardwood species, another probable reason for this tendency is variation in the range of tree sizes observed in our data compared to that of the data used to develop the model. The data used to develop the model relied heavily upon measurements from eastern forests which, due to climate and soil conditions, tend to have greater height growth relative to d.b.h. compared to many hardwood species in Missouri. This possibility corresponds well to the results obtained from fitting the model to our data, as an assumption of greater height relative to d.b.h. would lead to compounded overestimation as predicted values increase. This assessment of the original diameter model reinforced the need to refit the model to our measurement data and to compare the results to several other forms of the model.

Comparison with Other Common Diameter-based Model Forms

The models chosen for comparison to the original diameter-based model from Jenkins et al. (2003) represent model forms developed in several regions of the United States and Canada (Jenkins et al. 2004). Each model form uses some derivation of d.b.h. as its only covariate. Unlike the model from Jenkins et al. (2003), it was not possible to assess many of the alternative models using predetermined coefficients due to a lack of available coefficient estimates for the species of interest in this study. Therefore, this analysis focused on a refitting of the Jenkins et al. (2003) model (hereafter referred to as Model A) to our data for each of the four species and comparing it to several alternative model forms also fitted to our data. In all, we compared predictions from four nonlinear models and two linear models. Table 2 shows the coefficient estimates for each model form (hereafter referred to as Model A through Model F) fitted to our tree data by species.

Note that the model form was maintained for each fitted model, regardless of whether or not all coefficients were statistically significant. Most of the coefficient estimates lacking significance were intercepts, which is understandable due to the logic of aboveground biomass passing through the origin with respect to d.b.h. Model E had the greatest number of nonsignificant coefficients. Calculation of variance inflation factors (VIF) for Model E indicated high multicollinearity for the coefficients associated with d.b.h. and $(d.b.h.)^2$. This likely influenced the significance of coefficients for this model because multicollinearity can make estimates of coefficient standard error inaccurate, though it has no effect on the prediction capabilities of the model. Table 3 shows the summary statistics for precision and bias for each model by species.

Fitting Model A to our data drastically improved the precision and bias of prediction compared to the original coefficient estimates from Jenkins et al. (2003). Additionally, Models A, B, and D produce estimates that are very similar, to the point that RRMSE and RB are nearly indistinguishable between these models. Model C, which uses coefficients for both d.b.h. and $(d.b.h.)^2$, consistently outperforms the other models with regard to precision, though it does not do quite as well in terms of bias. However, with the maximum difference between estimates being about 4 percent for RRMSE and about 2 percent for RB, the models are fairly comparable. Although summary statistics indicate the general performance of the models, analysis of residuals is much more revealing of the key differences in prediction between the models. For illustration of residual plots, we chose to focus on hickory, as it was the species with the greatest variation in precision and bias (Fig. 3).

Table 2.—Coefficient estimates for alternative diameter-based aboveground tree biomass models fitted to the tree data for our four species of interest

Model	Model form	Hickory			White oak			Black oak			Post oak		
		β_0	β_1	β_2	β_0	β_1	β_2	β_0	β_1	β_2	β_0	β_1	β_2
Nonlinear													
A	$bm = \exp(\beta_0 + \beta_1 * \ln(dbh))$	-3.39	2.82		-0.09*	1.89		-1.67	2.28		0.33*	1.71	
B	$bm = \exp(\ln(\beta_0) + \beta_1 * \ln(dbh))$	0.02*	3.01		0.91	1.89		0.14	2.36		1.38	1.71	
C	$bm = \exp(\beta_0 + \beta_1 * dbh + \beta_2 * \ln(dbh))$	5.79	0.11	-0.46*	-7.76	-0.07	2.35	-7.1	-0.04	2.12	-7.54	-0.07	2.29
D	$bm = \exp(\beta_0 + \beta_1 * \ln(dbh))$	-3.39	1.41		-0.09*	0.95		-1.66	1.14		0.33*	0.86	
Linear													
E	$bm = \beta_0 + \beta_1 * dbh + \beta_2 * dbh^2$	1222.41	-91.69	2.21	-772.56	43.76	0.04*	-74.91*	-2.64*	0.66	-661.76	40.76	-0.09*
F	$bm = \beta_0 + \beta_1 * dbh^2$	-166.85	0.76		-3.78*	0.61		-121.89	0.62		58.61*	0.43	

Values marked with * are not statistically significant.

Equations from Goldsmith and Hocker 1978; Goltz et al. 1979; Hahn 1984; Jenkins et al. 2003; Jenkins et al. 2004; Raile, 1981; Ruark and Bockheim 1988.

Table 3.—Summary statistics for all diameter-based models by species

Model	Hickory			White oak			Black oak			Post oak		
	N	RRMSE	RB	N	RRMSE	RB	N	RRMSE	RB	N	RRMSE	RB
A	33	26.82	-2.11	60	28.72	1.44	63	24.23	0.72	59	32.61	1.57
B	33	26.82	-2.11	60	28.72	1.44	63	24.23	0.72	59	32.61	1.57
C	33	24.11	-4.56	60	26.49	-0.45	63	23.77	-2.81	59	30.01	-0.57
D	33	26.82	-2.11	60	28.72	1.44	63	24.23	0.71	59	32.61	1.57
E	33	26.34	-4.32	60	27.48	<0.01	63	24.14	-2.13	59	30.97	<0.01
F	33	29.61	-3.91	60	28.87	<0.01	63	24.12	-2.14	59	33.49	<0.01

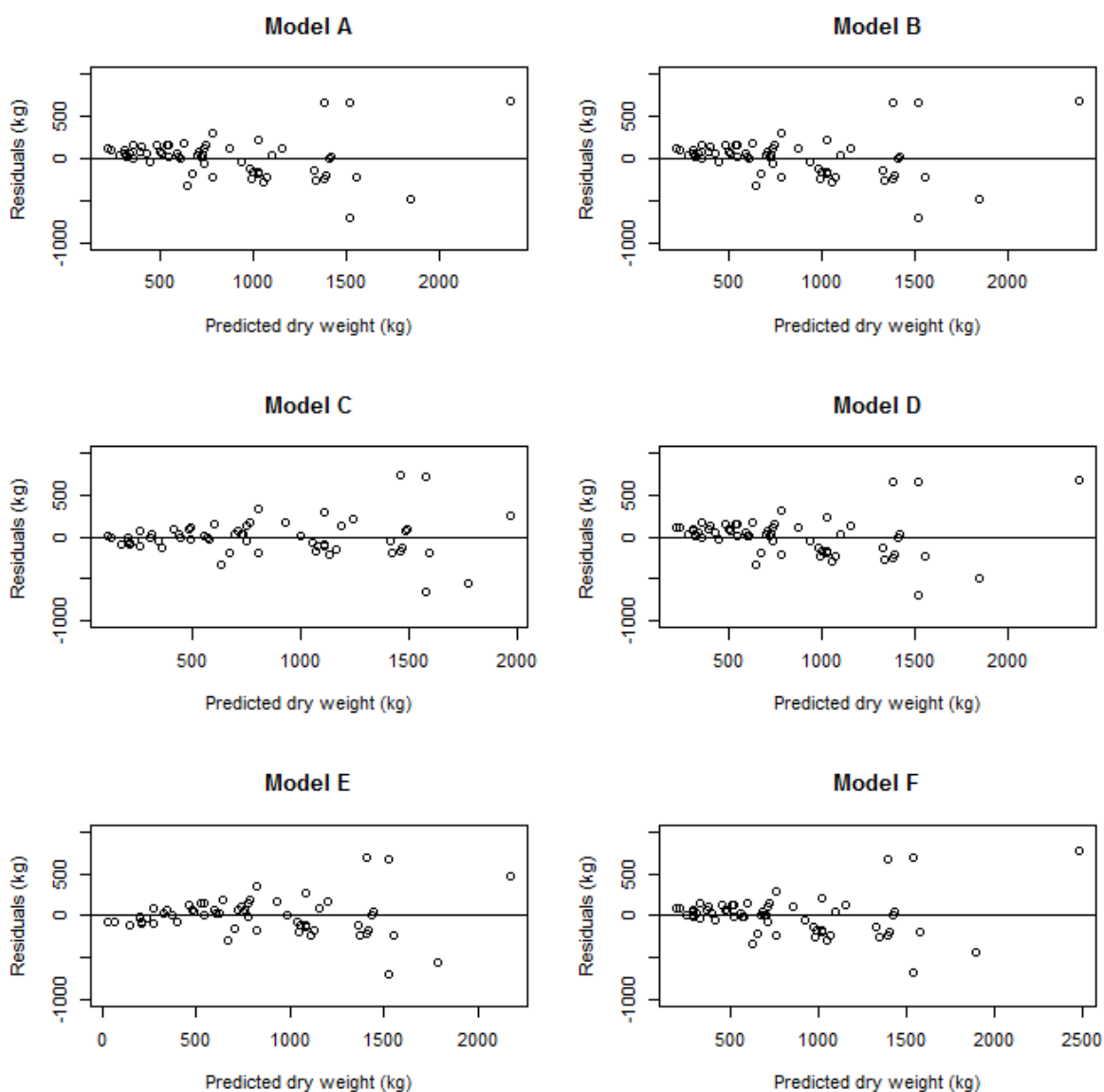


Figure 3.—Residuals of prediction of aboveground biomass for hickory using diameter-based model forms listed in Table 2.

There is a tendency for most models to overestimate for low predicted values; however, Model E appears to actually underestimate instead. Model C was the only model that did not appear to overestimate or underestimate for low predicted values. Additionally, Model C stood out from the other models in that it produced a more even distribution of predicted values, as well as a tighter arrangement of residuals compared to many of the other models. The linear models (Models E and F) tended to have the widest arrangement of residuals with regard to relatively extreme values.

Although the linear models performed reasonably well with regard to prediction of aboveground biomass using d.b.h., we opted to drop these models for the remainder of the analysis. Linear models can be unreliable when predicting values for trees that are beyond the size range of the trees originally used when creating the models because forest attributes, such as volume and aboveground biomass, are often inherently nonlinear in nature (hence the prevalence of nonlinear models in the literature). Nonlinearity can cause transformed linear models such as Models E and F to lose precision and accuracy when covariate values are not represented by the range of values in the data set used to develop the model. Nonlinear models are often more robust to extrapolation of this kind, in part due to the greater ability of nonlinear regression to produce reliable estimates of coefficients with relatively small data sets.

Influence of Height in Nonlinear Models

As previously stated, there were two general strategies regarding inclusion of total height into the nonlinear models for aboveground biomass. The first strategy applies to Models A and B in which height (ht) is added into the model as a separate covariate. The second strategy applies to Models C and D and consists of including height as a multiplier to (d.b.h.)². Recall that this strategy stems from a desire to mimic the use of height in traditional volume equations under the assumption that the volume and aboveground woody biomass are highly correlated. Table 4 shows the coefficient estimates for each nonlinear model form utilizing total tree height.

As with the original diameter-based models, the most common coefficient to show nonsignificance was the intercept. However, with black oak several models had nonsignificant coefficients. For Models A and B, the coefficient for height was not statistically significant for black oak, indicating that height was not very influential for these model forms. We postulate that low variation in the black oak sample could be a likely cause for this. Simply put, the black oak sample used for the study displayed lower variation of height relative to diameter than some of the other species, in which case height as a separate coefficient would not provide much additional information regarding aboveground woody biomass after accounting for d.b.h. A similar effect existed in the post oak sample, which ultimately influenced the results of the models when crown ratio was included as a covariate, as will be shown in the next section. Table 5 shows the summary statistics for precision and bias for each model by species.

The nonlinear models that included height generally yielded greater precision for most species when compared to the diameter-based models. Even though Models A and B showed nonsignificance for the height coefficient in the case of black oak, there was still a slight improvement in precision. The change in relative bias between the diameter-based models and models including height was somewhat more sporadic. Notably, the RB for all models except Model C showed an increase in relative bias for hickory. This was in contrast to the general tendency of the models to produce

Table 4.—Coefficient estimates for alternative nonlinear aboveground tree biomass models fitted to the tree data for our four species of interest using both d.b.h. and height

Model	Model Form	Hickory			White Oak			Black Oak			Post Oak		
		β_0	β_1	β_2	β_0	β_1	β_2	β_0	β_1	β_2	β_0	β_1	β_2
Nonlinear													
A	$bm = \exp(\beta_0 + \beta_1 * \ln(dbh) + \beta_2 * \ln(ht))$	-5.19	2.65	0.82	-3.03	1.72	1.21	-2.17	2.14	0.35*	-1.15*	1.46	0.86
B	$bm = \exp(\ln(\beta_0) + \beta_1 * \ln(dbh) + \beta_2 * \ln(ht))$	0.01*	2.66	0.82	0.05*	1.72	1.21	0.11	2.14	0.35*	0.32*	1.46	0.86
C	$bm = \exp(\beta_0 + \beta_1 * dbh + \beta_2 * \ln(dbh * ht))$	-1.18*	0.04	0.63	-4.75	-0.02	1.19	-1.04*	0.02*	0.71	-5.96	-0.03	1.38
D	$bm = \exp(\beta_0 + \beta_1 * \ln(dbh^2 * ht))$	-5.65	1.23		-2.46	0.91		-2.82	0.93		-0.98*	0.75	

Values marked with * are not statistically significant.

Table 5.—Summary statistics for all diameter and height models by species

Model	N	Hickory			White oak			Black oak			Post oak		
		RRMSE ^a	RB ^b	N	RRMSE	RB	N	RRMSE	RB	N	RRMSE	RB	N
A	33	23.06	-3.31	60	22.16	0.09	63	23.78	0.64	59	30.84	1.84	59
B	33	23.06	-3.31	60	22.16	0.09	63	24.78	0.64	59	30.84	1.84	59
C	33	21.25	-2.41	60	21.85	-0.27	63	24.7	0.93	59	29.08	0.88	59
D	33	24.23	-3.52	60	22.65	0.34	63	24.99	0.54	59	30.87	1.84	59

^a RRMSE=relative root mean squared error.

^b RB=relative bias.

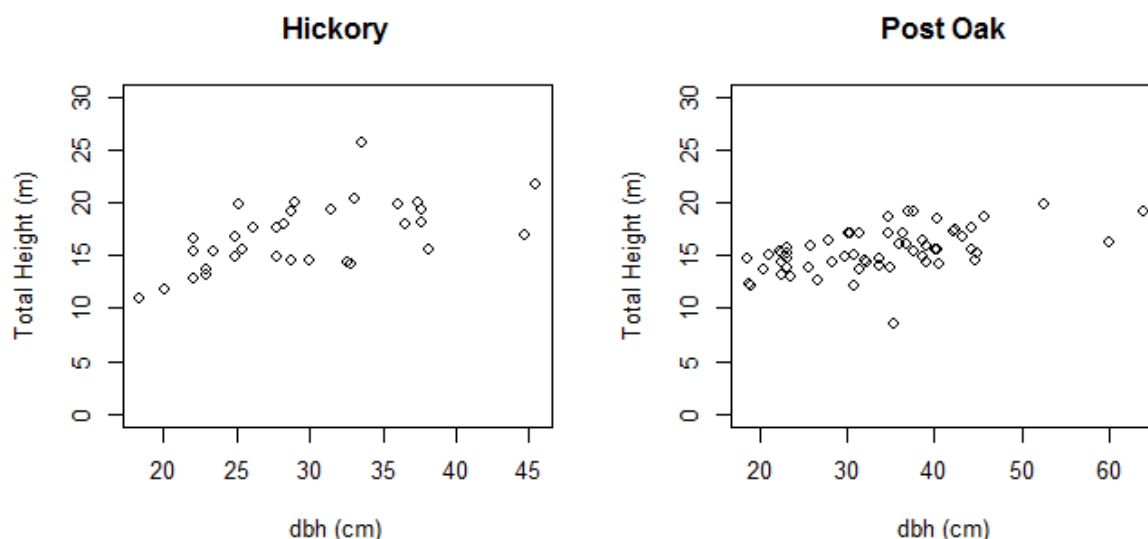


Figure 4.—Scatterplots of total height versus d.b.h. for hickory and post oak based on the sample data for this study.

lower RB for most species in the study. The primary reason for an increase in RB pertained to the relationship between height and d.b.h. in the sample for hickory used in this study. To illustrate how the relationship between d.b.h. and height can vary among species, Figure 4 shows scatterplots of total height to d.b.h. for hickory and post oak.

Hickory displays much greater variability with regard to the correlation between total height and d.b.h. The correlation between d.b.h. and height for post oak is quite linear and shows very little variation from low to high observed d.b.h. In contrast, for hickory height increases very quickly at low d.b.h. values but then plateaus at d.b.h. values greater than 25 cm. This particular trend in height versus d.b.h. for hickory likely contributed to the increased RB observed in Table 4, which is an artefact of including height as a covariate in the models. Note, however, that the inclusion of height had very little negative effect on the RB of Model C, which when combined with the superior performance of this model form for diameter-based estimation of aboveground biomass creates a strong argument for the use of Model C as a preferred model form for selected Missouri hardwood species.

Influence of Crown Ratio in Nonlinear Models

Including crown ratio as a covariate generally resulted in most nonlinear model forms performing poorly when estimating aboveground woody biomass. The only exception was with the estimation of aboveground biomass for black oak and post oak. For black oak, crown ratio was only significant in Model C. For post oak, crown ratio was statistically significant in each of the nonlinear model forms. To explain this occurrence, we must once again refer to the differences in growth patterns between the different species.

Recall from Figure 3 that post oak had a very small slope for the linear relationship between total height and d.b.h. This was most likely an artifact of the tendencies of post oak to reach a height and d.b.h. apex at a fairly young age. In short, the range of ages for the post oak trees sampled in this study had much greater variability than the d.b.h. and height ranges would indicate, creating a situation where trees of similar volume have very different biomass weights due to higher specific gravity for older trees. One variable that can help to explain differences between older and younger trees of similar size is crown ratio, due to the occurrence of relatively smaller crowns for older trees

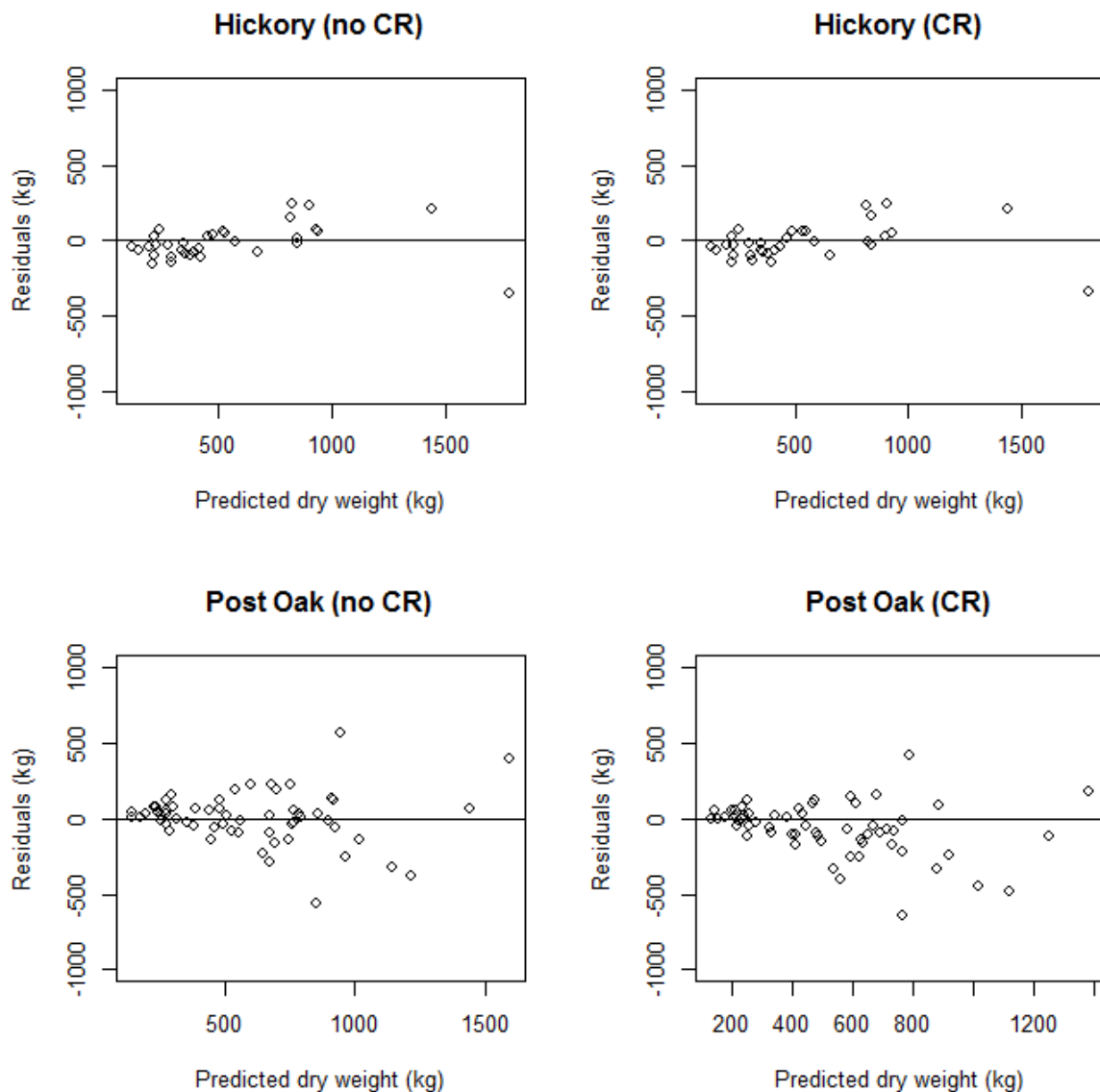


Figure 5.—Residual plots of estimates for aboveground biomass for hickory and post oak using Model C with and without crown ratio (CR) as a covariate.

that have spent much of their lifespan in closed canopy conditions. To visualize the effect of crown ratio on estimation of aboveground biomass for post oak, it is useful to compare residuals between post oak and one of the species that did not have crown ratio as a significant variable. Figure 5 shows residual plots for Model C with and without crown ratio as a covariate for hickory and post oak.

The difference between the residual plots with and without the inclusion of CR was quite subtle but revealed the variation in how CR influenced estimation of aboveground biomass between the two species. Most importantly, notice that the inclusion of CR in Model C for post oak slightly reduced bias at low predicted values. Additionally, it reduced some of the more extreme residuals for high predicted values observed from the model excluding CR. This demonstrated that CR is statistically significant for post oak primarily because it provided information that is useful in estimating aboveground biomass for trees that are at either the low end or the high end of the range of d.b.h. and heights for the sample of that species. This compliments the argument that a full understanding of aboveground woody biomass weight for post oak could go beyond a simple measure of volume based on d.b.h. and height alone.

Best Overall Model for Estimating Aboveground Biomass for Selected Species

We showed that the diameter-based model form by Jenkins et al. (2003) can produce fairly precise and accurate estimates of aboveground woody biomass for selected hardwood species if fit specifically to tree data from those species. However, we also showed that other common model forms may perform even better if fit using the same data. The model form that consistently performed the best overall with regard to precision and bias was Model C. This model consistently outperformed all other models with regard to precision whether using only d.b.h. as a covariate or incorporating height as a multiplier to (d.b.h.)². While this model did not always produce the lowest bias, it produced RB values well within acceptable tolerances and was the only nonlinear model that showed a decrease rather than an increase in RB with the inclusion of height as a covariate. Although all nonlinear models performed fairly well based on our tree data, Model C would be the recommended model form to use for these four species whether using only d.b.h. or d.b.h. and height combined.

Application of Optimal Model for Merchantable Biomass

The analyses indicated that of all the model forms assessed in this study, the Model C form was optimal with regard to both precision and bias. Therefore, it was logical to assess this model form for estimation of merchantable woody biomass. Note that CR was omitted from this particular analysis as CR had minimal effect on estimation of total aboveground woody biomass, and models for estimating merchantable biomass should be tailored to use measurements that can be taken by loggers on merchantable logs obtained from felled trees. Recall that for this version of Model C, total tree height was replaced by merchantable height measured as a sum of merchantable log lengths cut from each tree. Table 6 shows the coefficient estimates and summary statistics for the final model of merchantable biomass by species.

White oak showed the poorest fit for the merchantable biomass model as indicated by the low statistical significance of the coefficient for d.b.h. as well as higher RRMSE and RB than any other species. For the other species, the merchantable biomass model actually yielded considerably lower RRMSE and RB than the total aboveground biomass counterpart models using the Model C form. This was not entirely surprising given that a biomass estimate based solely upon merchantable stem should have high correlation with the traditional height x d.b.h.² method of volume estimation without additional variation caused by inclusion of tops, branches, and leaves. The under performance of the model for white oak compared to the other species was most likely caused by

Table 6.—Coefficient estimates and summary statistics for the final model of merchantable biomass by species

Species	Coefficient estimates			Summary statistics		
	β_0	β_1	β_2	N	RRMSE ^a	RB ^b
Hickory	-0.49 ^{*c}	0.023	0.61	33	12.49	0.21
White oak	-0.66 [*]	0.001 [*]	0.71	60	22.95	1.42
Black oak	0.53 [*]	0.019	0.51	63	16.15	1.27
Post oak	-1.85	0.007	0.82	59	15.67	0.17

^a RRMSE=relative root mean squared error.

^b RB= relative bias.

^c Values marked with * are not statistically significant.

inconsistencies in taper of the merchantable stems. This was most apparent in the contrast with post oak, which typically has very little taper by comparison. This study has shown that traditional nonlinear models forms can be applied to major Missouri hardwood species to derive relatively precise and accurate estimates of both total aboveground biomass and merchantable biomass. For estimated model coefficients for predicting merchantable biomass in oven-dry pounds using d.b.h. measured in inches and merchantable height measured in feet, please refer to the Appendix.

CONCLUSION

The goal of this study was to compare the traditional model for estimating aboveground woody biomass with estimates derived from the same model form refit to tree data taken in an intensive inventory for southeast Missouri. Additionally, it was our objective to compare estimates from the refit standard model to other model forms fit to the same tree data for estimating both total aboveground biomass and merchantable biomass.

Comparisons of summary statistics and residuals from both nonlinear and linear diameter-based models indicated that refitting traditional model forms to data collected from Missouri hardwood species improved upon precision and accuracy of estimates from the original model of Jenkins et al. (2003). The inclusion of height into the nonlinear model forms generally resulted in somewhat higher precision of estimation for total aboveground biomass, though bias increased slightly for some species. Increase in bias was mainly an issue for hickory, most likely due to particular trends in d.b.h. vs. height for this species group. The only species that benefited from the inclusion of CR with regard to estimation of biomass was post oak.

Although the refitting of the Model A form showed considerable improvement over the coefficient estimates provided by Jenkins et al. (2003), the analysis indicated that the Model C form performed the best overall for all species with the inclusion of height as a covariate. The resulting models for merchantable biomass showed considerable improvement in both precision and bias when compared to the counterpart models for total aboveground biomass for most species. This study has shown that many traditional nonlinear tree biomass equations can be used to obtain precise and accurate estimates of both total aboveground biomass and merchantable biomass when fit specifically to Missouri hardwood species. Additionally, the resulting models from this study provide practical tools for the forest products industry of Missouri to efficiently estimate harvested biomass prior to sale at a precision similar to volume estimation.

ACKNOWLEDGMENTS

The authors would like to acknowledge the Missouri Forest Products Association and the U.S. Forest Service, State and Private Forestry for funding and assistance in implementing this project. Additionally we would like to thank Rudd Equipment Company for assistance provided in weighing the trees.

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APPENDIX

Coefficient estimates for the final model of merchantable biomass in overdry pounds by species using d.b.h. in inches and merchantable height in feet

Species	Coefficient estimates		
	β_0	β_1	β_2
Hickory	0.70177	0.05791	0.60755
White oak	0.61557	0.00373	0.71159
Black oak	1.67079	0.04796	0.51286
Post oak	-0.50714	0.01655	0.81549

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

A PRELIMINARY ABOVEGROUND LIVE BIOMASS MODEL FOR UNDERSTORY HARDWOODS FROM ARKANSAS, LOUISIANA, AND MISSISSIPPI

Don C. Bragg and D. Andrew Scott¹

Abstract.—Hardwood understories can contribute significantly to total ecosystem biomass and fuel loads, but few models are available to directly quantify this component. In part, this is due to the small size of the hardwoods. Many understory trees simply do not reach the height required to determine diameter at breast height (d.b.h.), so conventional models (e.g., the National Biomass Estimators [NBE]) that rely on this predictor are unavailable. Further, understory hardwoods can be present in such numbers or have inconvenient growth forms such that biomass estimates based on diameters are impractical. However, a quick and easily measured attribute, stem length, can be used instead of diameter to facilitate understory hardwood biomass estimation. We destructively sampled 513 small hardwood shrubs and trees in Arkansas, Louisiana, and Mississippi and oven dried their aboveground live biomass (stems, branches, leaves) to a constant weight. The high degree of variability in plant form, branch patterning, and wood density among the 31 different taxa sampled suggested that a single hardwood grouping would be as effective as more specific equations. Nonlinear ordinary least squares regression was then used to predict aboveground live biomass with a modified version of the NBE (using stem length rather than d.b.h.). The coefficient of determination of the resulting model was reasonably high ($R^2 = 0.71$), particularly for data comprising such varied individuals. Further confirmation of the utility of this understory biomass model followed a comparison of several species with varying wood density.

INTRODUCTION

Research into the characterization of biomass resources has increased greatly in recent years as witnessed by a proliferation of articles, and even entire research journals, dedicated to this field. There are many practical reasons to study biomass, including the estimation of commercial product yields, quantification of fuel loads, determination of carbon sequestration trends, or description of habitat conditions. To date, most efforts have concentrated on the more economically valuable species. The commercial importance of forests in the southeastern United States, for example, has supported the development of scores of biomass-related predictions (Baldwin 1987, Bullock and Burkhart 2003, Parresol 1999).

Because trees constitute the majority of the aboveground biomass in most forest ecosystems, the prediction of individual stem biomass has been a high priority for most modelers. This has led to the development of a range of models, from finely tuned local designs (e.g., McElligott and Bragg, in press) to more widely developed regional (e.g., Bullock and Burkhart 2003) and national models (e.g., Jenkins et al. 2003, Ruiz-Peinado et al. 2012). These approaches have their strengths and weaknesses, and virtually all of them rely on the use of diameter at breast height (d.b.h.) to predict

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the aboveground biomass of major tree species. Although hardwood biomass research has generally lagged behind that of conifers, a growing number of predictive models have been developed recently, such as those for the United States (Jenkins et al. 2003) and Europe (Ruiz-Peinado et al. 2012, Zianis et al. 2005).

The low value of smaller trees in the central hardwoods has limited commercial development and constrained silvicultural practices, although biomass-based markets could present new opportunities (Kabrick et al. 2013). To take advantage of these opportunities, a better accounting of the entire forest biomass resource is required. However, as in conifers, most hardwood research has focused almost entirely on larger stems. This tendency overlooks one potentially major source of biomass, understory hardwood trees and shrubs. Unfortunately, these hardwoods are a difficult resource to assess. Only a handful of models capable of directly estimating the biomass of understory trees and shrubs with a measurable d.b.h. exist. For example, Phillips (1981) predicted aboveground biomass of understory hardwoods between 2.5 and 12.5 cm d.b.h., and the National Biomass Estimator (NBE) hardwood groups extend down to 3 cm d.b.h. (Jenkins et al. 2003). Given the use of d.b.h. in most allometric relationships, the scarcity of biomass models for hardwoods that fail to reach this height threshold (1.37 m) is understandable. Height and some measure of diameter have also been used in combination to improve biomass estimation (Joosten et al. 2004, Phillips and Saucier 1979, Ruiz-Peinado et al. 2012). Such an effort requires the measurement of two variables (height and diameter), which can add to the time it takes to measure this component in the field.

For the smallest hardwoods, measuring diameter means sampling something other than d.b.h. since they may not reach the necessary height (1.37 m). Typically, this means ground line (root collar) diameter or basal diameter, which is often defined as stem thickness at 15 cm above the ground surface. A few studies have evaluated the biomass of woody shrubs and understory trees in terms of these alternative forms of stem diameter or some other measure of plant size. For example, Brown (1976) and Smith and Brand (1983) used basal diameter (stem diameter at ground line or 15 cm above the ground) to predict biomass for a number of shrubs in the northern latitudes of North America, and Bentley et al. (1970) and Vora (1988) both predicted the biomass of some California shrubs using measures of crown volume.

These dimensions can be challenging to measure, especially in dense understories or for multi-stemmed specimens, leaving a regrettable knowledge gap. Understory hardwoods can contribute significantly to total ecosystem biomass and related properties such as fuel loading, nutrient accumulation, or carbon sequestration. More choices for modeling understory hardwood biomass compatible with existing assessments without unduly burdensome measurement requirements are needed. Height classes are frequently used in understory inventories (Bragg and Heitzman 2009, Brose 2011, Gould et al. 2006), making stem length a convenient and logical option. Preliminary work by Scott et al. (2006) suggested that stem length alone may prove an effective alternative for diameter for understory hardwoods. Hence, our work represents a further exploration of the utility of a stem length-based aboveground biomass model for understory hardwood trees and shrubs from the middle southern states.

METHODS

Study Locations and Sampling Protocols

Samples were opportunistically selected from a number of sites in Arkansas, Louisiana, and Mississippi. Hardwood trees and shrubs growing in the understory of naturally regenerated, pine-dominated stands (both even and uneven aged) from compartments across the Crossett Experimental Forest in southeastern Arkansas were sampled during the summer of 2012. Small trees and shrubs were also harvested during the summers of 2003 and 2004 from 5- and 12-year-old loblolly pine plantations on the Palustris Experimental Forest in central Louisiana and from a 10-year-old loblolly pine plantation on the DeSoto National Forest in southeastern Mississippi.

Over 500 understory hardwood trees and shrubs were destructively collected to provide the aboveground live biomass (stems, branches, leaves) for this study, encompassing a range of different sizes for the most common species found at each site (Tables 1 and 2). Most sampled hardwoods were individual stems. For the specimens that had more than one stem, the samples were weighed based on total biomass for the whole plant, but only the length of the longest stem was used for that variable. To standardize for individual stems, the total biomass was then divided by the number of stems in these multi-stemmed understory hardwoods, and each was assigned the measured longest length. Stem length of each specimen was measured to the nearest centimeter in the field prior to being cut flush at ground level and packed into paper bags for further lab processing. Biomass samples were oven dried (at temperatures of at least 70 °C) to a constant weight, which was recorded to the nearest gram.

Model Selection and Evaluation

For this project, the following exponential function based on the NBE equation (Jenkins et al. 2003) was fit to the data:

$$AGB = e^{b_1 + b_2 \ln(L)} \quad (1)$$

where *AGB* is the oven-dry weight of aboveground live biomass (kg), *L* is the stem length (cm), and *b*₁ and *b*₂ are coefficients fit using nonlinear ordinary least squares regression. The following coefficient of determination for this general equation was provided by the fitting software:

$$R^2 = 1 - \sum(y - \hat{y})^2 / \sum(y - \bar{y})^2 \quad (2)$$

Because Equation 1 is nonlinear, Equation 2 cannot be interpreted in the same fashion as in linear regression, so the coefficient of determination is called a fit index or “pseudo-*R*²”. Even though it is commonly generated by statistical software packages, the use of Equation 2 for nonlinear regression has been roundly criticized as a tool to compare models (e.g., Kvålseth 1985, Spiess and Neumeyer 2010). However, because we are describing a specific predictive tool rather than making comparisons, we present *R*² as a simple expression of goodness of fit between the model and data.

Note that Equation 1 provides results on a per stem basis. Determining the total biomass for a multi-stemmed hardwood (e.g., a clump of stump sprouts) would require summing individual stem estimates. The high degree of variability in plant form, branch patterning, wood density, and limited

Table 1.—Understory hardwood trees and shrubs sampled from stands in Arkansas, Louisiana, and Mississippi used to develop the aboveground biomass model

Common name	Scientific name	Specific gravity ^a	n
Sweetgum	<i>Liquidambar styraciflua</i>	0.52	54
Red maple	<i>Acer rubrum</i>	0.54	53
Common persimmon	<i>Diospyros virginiana</i>	0.74	43
American beautyberry	<i>Callicarpa americana</i>	n/a	43
White oak	<i>Quercus alba</i>	0.68	30
Water oak	<i>Quercus nigra</i>	0.63	30
Southern red oak	<i>Quercus falcata</i>	0.59	29
Winged sumac	<i>Rhus copallinum</i>	n/a	23
Winged elm	<i>Ulmus alata</i>	0.66	20
Post oak	<i>Quercus stellata</i>	0.67	19
Mockernut hickory	<i>Carya tomentosa</i>	0.72	18
Gallberry	<i>Ilex glabra</i>	n/a	18
Sassafras	<i>Sassafras albidum</i>	0.46	18
American holly	<i>Ilex opaca</i>	0.57	16
Flowering dogwood	<i>Cornus florida</i>	0.73	15
Ash	<i>Fraxinus</i> spp.	n/a	15
Horse-sugar	<i>Symplocos tinctoria</i>	n/a	15
Buckthorn	<i>Rhamnus</i> spp.	n/a	11
Black hickory	<i>Carya texana</i>	n/a	8
Oak	<i>Quercus</i> spp.	n/a	5
Wax myrtle	<i>Morella cerifera</i>	n/a	5
Blueberry	<i>Vaccinium</i> spp.	n/a	4
Privet	<i>Ligustrum</i> spp.	n/a	4
Blackgum	<i>Nyssa sylvatica</i>	0.50	3
Viburnum	<i>Viburnum</i> spp.	n/a	3
Baccharis	<i>Baccharis halimifolia</i>	n/a	3
Yaupon	<i>Ilex vomitoria</i>	n/a	2
Black cherry	<i>Prunus serotina</i>	0.50	2
Chinese tallowtree	<i>Triadica sebifera</i>	n/a	2
Willow oak	<i>Quercus phellos</i>	0.69	1
Devil's walkingstick	<i>Aralia spinosa</i>	n/a	1

^a Specific gravity of wood only, based on oven-dry weight and 12 percent moisture content for volume; adapted from Table 1A in Miles and Smith (2009); n/a = not available.

sample size for some species among the 31 different taxa sampled suggests that a single hardwood predictive model is probably as useful as more specific equations in this study. To further consider the utility of Equation 1 for a given species, actual data from the following three hardwood species with relatively large sample sizes (at least 25 individuals) and a range of wood specific gravities (SGs) were visually compared to the predictions from the equation: sweetgum (SG = 0.52), southern red oak (SG = 0.59), and persimmon (SG = 0.74). See Table 1 for scientific names and SGs for all species.

Table 2.—Range of stem length and aboveground live oven-dry (OD) biomass data used to derive understory hardwood model

Common name	Stem length				Aboveground live OD biomass			
	Min.	Max.	Avg.	Standard deviation	Min.	Max.	Avg.	Standard deviation
	<i>centimeters</i>				<i>kilograms</i>			
Sweetgum	30	427	118.5	74.7	0.003	1.460	0.123	0.2202
Red maple	32	396	108.0	62.6	0.015	1.333	0.081	0.1810
Common persimmon	30	549	134.3	115.7	0.003	3.336	0.256	0.6330
American beautyberry	35	225	116.1	50.5	0.010	0.475	0.072	0.0840
White oak	32	176	84.9	44.3	0.016	0.229	0.063	0.0584
Water oak	31	213	109.1	53.9	0.016	0.330	0.090	0.0915
Southern red oak	41	216	99.0	41.4	0.021	0.264	0.104	0.0767
Winged sumac	43	222	102.8	48.4	0.020	0.520	0.117	0.1264
Winged elm	37	216	107.0	66.2	0.013	0.212	0.078	0.0727
Post oak	35	170	81.9	37.5	0.019	0.345	0.107	0.1014
Mockernut hickory	37	182	94.8	42.3	0.016	0.245	0.106	0.0579
Gallberry	30	351	151.6	96.9	0.002	0.501	0.067	0.1130
Sassafras	41	381	119.3	77.5	0.003	1.000	0.130	0.2311
American holly	34	229	109.6	56.2	0.018	0.440	0.145	0.1312
Flowering dogwood	39	290	126.2	69.2	0.019	0.505	0.141	0.1559
Ash	40	189	96.0	42.9	0.019	0.206	0.069	0.0592
Horse-sugar	33	168	89.6	41.7	0.014	0.227	0.072	0.0565
Buckthorn	32	224	92.3	56.6	0.014	0.282	0.080	0.0949
Black hickory	40	175	121.0	54.3	0.035	0.522	0.215	0.1725
Oak	46	427	213.4	162.4	0.020	2.020	0.685	0.8238
Wax myrtle	30	274	140.2	123.5	0.000	0.821	0.258	0.3678
Blueberry	76	107	91.4	12.4	0.020	0.167	0.080	0.0655
Privet	107	457	304.8	148.8	0.040	1.690	0.789	0.7101
Blackgum	91	305	213.4	109.9	0.060	0.840	0.470	0.3915
Viburnum	152	244	203.2	46.6	0.093	0.310	0.217	0.1116
Baccharis	107	168	147.3	35.2	0.010	0.240	0.150	0.1229
Yaupon	122	213	167.6	64.7	0.047	0.820	0.433	0.5468
Black cherry	290	351	320.0	43.1	0.480	1.540	1.010	0.7495
Chinese tallowtree	61	366	213.4	215.5	0.005	0.428	0.216	0.2991
Willow oak	81	81	81.0	--	0.026	0.026	0.026	--
Devil's walkingstick	229	229	228.6	--	0.200	0.200	0.200	--

RESULTS AND DISCUSSION

The fitted understory hardwood biomass model and original data are shown in Figure 1. For the species evaluated (Table 1), $b_1 = -12.764$ and $b_2 = 2.161$ with a reasonably high (0.71) coefficient of determination. Although Equation 1 was not as good as some understory hardwood equations (e.g., Bentley et al. 1970, Phillips 1981, Telfer 1969, Vora 1988), this multispecies equation performed well, particularly given the known variation in growth form between a number of apically dominant taxa such as sweetgum and those with more spreading forms or multiple stem species such as baccharis or American beautyberry. Growth form dissimilarity (including branch patterns, proportions of foliage and bark to stem and branch wood, vigorous versus stunted, differential browsing) undoubtedly contributed to added noise in our understory hardwood biomass measurements. Other researchers have found similar levels of variation in their data (Brown 1976, Smith and Brand 1983, Vora 1988). The broad geographic distribution of the sampled hardwoods (encompassing multiple sites from three different states) would incorporate localized variation in form and growth habit, further contributing to the modest fit of Equation 1.

Nevertheless, this model should prove useful for many applications, particularly if needed to predict biomass for large-scale or aggregated assessments as opposed to projecting for specific individuals. For example, Equation 1 should adequately yield stand or landscape level predictions of understory hardwood fuel loadings or carbon sequestration. A distinct advantage of this model design is that measuring stem length is easier and quicker than diameter for understory hardwoods, especially in dense vegetation or when the plant form is shrubby. This should permit more efficient sampling of understory hardwood biomass, thereby reducing overall uncertainty when using aggregate applications of this model design.

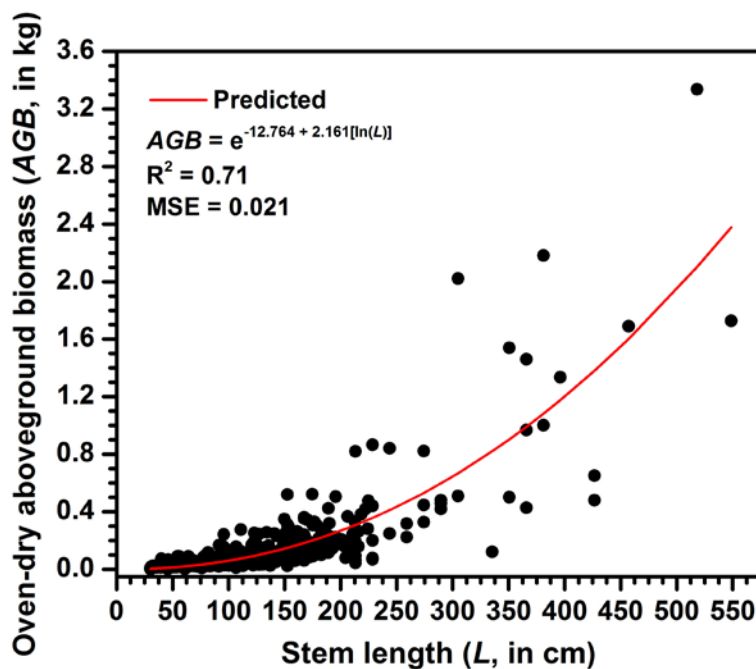


Figure 1.—Predicted (line) understory hardwood oven-dry aboveground live biomass (AGB) as a function of stem length (L) based on Equation 1, with all 513 data points included.

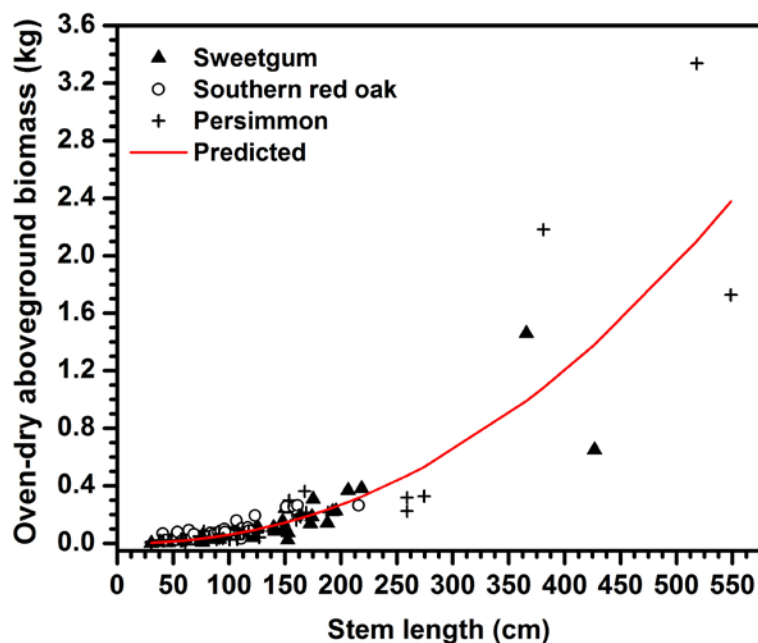


Figure 2.—Predicted and observed values of oven-dry aboveground biomass for three understory hardwood species across a range of specific gravities.

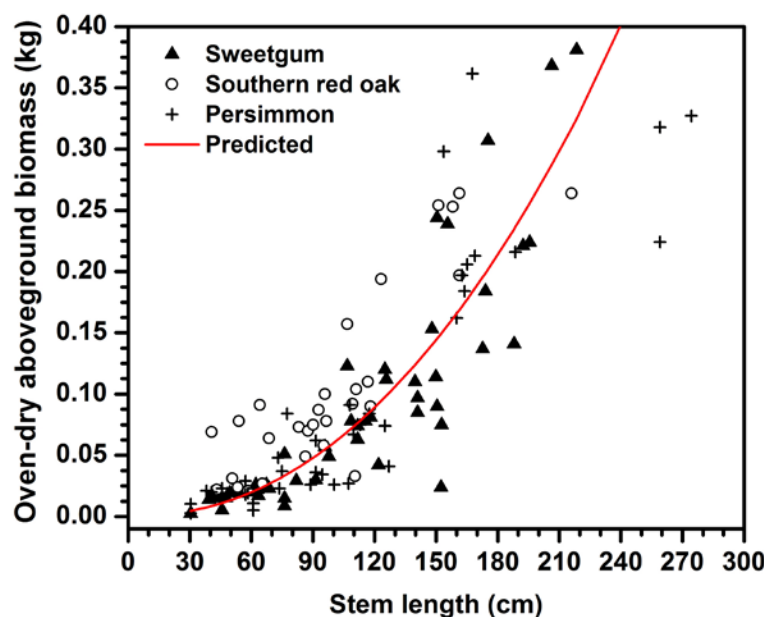


Figure 3.— Predicted and observed values of oven-dry aboveground biomass for a more limited subset (30 to 300 cm stem lengths) of the different understory hardwoods, used to highlight the nature of the fit across the majority of the observations.

Equation 1 predicted sweetgum, southern red oak, and persimmon about equally as well (Fig. 2), with no obvious trend with underestimation of AGB for species with high SG or overestimation of species with low SG. Based on a closer visual examination (no statistical contrasts were run) of the smallest trees (Fig. 3), southern red oak AGB tended to be under predicted using the model. Given that southern red oak was intermediate in SG, this result seemed counterintuitive at first. However, wood density is only one of several factors that contribute AGB for these small stems. We believe the under prediction of southern red oak can be explained by the greater amount of branching of this species in the understory.

CONCLUSIONS

Given the growing interest in describing comprehensive vegetative structure and dynamics for a variety of purposes (e.g., Alaback 1982, Gower et al. 2001, Lugo 1992, Reiner et al. 2010), the use of an understory-specific biomass predictor should help managers and researchers better understand the systems they are evaluating. Large scale or aggregated biomass estimates of regional carbon sequestration patterns or stand level fuel loads, for instance, can be collected quickly and more reliably if less effort is put into time consuming measurements of stem attributes, such as diameter or crown volume, and more time is invested into determining spatial patterns of understory distribution.

Although we anticipate further refinement, our preliminary results show that a simple length-based model can reasonably predict understory hardwood biomass for many different species across a range of site conditions. The noise in our data, even within species, favors the use of a single hardwood biomass model instead of multiple models based on individual taxa. This length-based approach appears to be an acceptable method for estimating biomass or fuel load, even when considering the range of different growth forms and wood densities. Such an aboveground biomass model has considerable utility for managers, permitting them to better quantify the attributes of their hardwood ecosystems.

ACKNOWLEDGMENTS

We would like to thank the following U.S. Forest Service staff for their assistance in collecting the field data for this project: Morris Smith, Jr., Kirby Sneed, and Rick Stagg. Comments by Nancy Koerth, Mike Shelton, and two anonymous reviewers helped improve this paper. Our research was supported by USDA CSREES Prime Agreement 2009-35103-05356, the U.S. Forest Service, the Arkansas Forest Resources Center, and the University of Arkansas-Monticello.

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NEW EFFORTS IN EASTERN COTTONWOOD BIOMASS PRODUCTION THROUGH BREEDING AND CLONAL REFINEMENT

Jason W. Cromer, Randall J. Rousseau, and B.Landis Herrin¹

Abstract.—First generation biofuels (also known as traditional biofuels) primarily use corn to produce ethanol. Newer techniques and knowledge are now allowing ethanol production from renewable resources such as trees that have more complex molecular structures that inhibit access to sugars. Ethanol production is through an enzymatic process which uses cellulose, or pyrolysis which uses lignin from trees. When nonedible renewable resources, such as trees or agricultural crops, are converted into alcohol or some other energy source, they are termed advanced biofuels (second generation biofuels). In order to supply the demand for advanced biofuels, companies are looking for fast-growing species for the production of biomass. *Populus* species including eastern cottonwood (*Populus deltoids*) and hybrid poplars (*Populus* spp.) are key species. *Populus* breeding programs are developing new individuals to maximize biomass production under plantation settings. Dedicated energy plantations of select *Populus* species and hybrids, if shown to be economically viable, could provide a significant source of biomass for the southern United States. Although poplars have shown exceptional productivity (tons/acre/year) on suitable sites in the lower Mississippi Alluvial Valley (LMAV), the key will be to increase adaptability and yields across the south with minimal input over a 3- to 5-year period.

The majority of the previous *Populus* improvement work has focused on the collection of individuals (i.e., phenotypes) from young 1- or 2-year-old native stands or clones developed from open-pollinated seed under nursery type settings. Since the mid-1980s, cottonwood tree improvement work in the southern United States has dwindled. But, with the increased emphasis on biomass production for biofuel and bioenergy, eastern cottonwood and a variety of hybrid poplars are being re-examined.

In 2012, Mississippi State University began breeding efforts under a grant. Selected clones were used as the parental population, and the resulting progeny have been established in field trials to examine survival, growth, and disease resistance. In addition to these breeding efforts, clonal refinement tests were established in 2012 and 2013 to examine clonal performance of cottonwood clones on both alluvial and upland sites. In 2012, a test of 17 highly selected cottonwood clones was established near Stoneville, MS. In 2013, another 47 eastern cottonwood clones, including many untested clones, were selected and are being examined for growth, disease resistance, and rootability on two sites in Mississippi (Newton and Leland, MS). Results from these trials will be used to select the best clones for inclusion into larger block trials as well as to provide new selections for inclusion into the breeding program. Our goal is to produce clones that exhibit rapid growth, high survival rates, increased disease resistance, and wood characteristics suitable for use in the bioenergy and biofuels programs in the United States as well as worldwide.

In the spring of 2012, the first test site was established in Stoneville, MS with 17 *P. deltoides* clones in a random complete block design consisting of 10 blocks. All 17 clones were planted as 18 inch unrooted cuttings at a spacing of 6 feet x 9 feet. Prior

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to planting, all cuttings were soaked in Admire® Pro (Bayer CropScience, Research Triangle Park, NC) to control cottonwood leaf beetles during the first year. After planting, weed and grass competition was maintained on all sites by chemical and mechanical methods to ensure survival and growth. Height growth at the Stoneville test was assessed at 2-week intervals during the summer of 2012. Biweekly measurements were taken to determine intervals of growth that may be useful for early age selection. The constant inspection also allowed the determination of when disease and defoliation occurred as well the progression of the disease and/or insect damage. After the 2012 growing season, measurements on the site will be recorded annually.

In 2013, a clonal refinement test was established on two sites in Mississippi. Both sites were previously in agriculture, with one site located near the Leland, MS site which is in the LMAV. The second site was located near Newton, MS on an upland soil.

The test design for the 2013 clonal refinement test was a randomized complete block consisting of 12 blocks and 47 cottonwood clones which were arranged in two-tree row plots at a spacing of 6 feet x 9 feet. Both test sites will be measured annually for the first 5 years to determine the appropriate selection age. Traits measured will include total height and disease resistance at age 1 followed by diameter at breast height, total height, and disease resistance at ages 2 through 5 years. These measurements will be used to assess genetic variation, heritability, genetic and phenotypic age-age correlations, and genetic gain per unit of time for various sites across Mississippi.

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

BLACK WILLOW TREE IMPROVEMENT: DEVELOPMENT OF A BIOMASS SPECIES FOR MARGINAL AGRICULTURAL LAND IN THE LOWER MISSISSIPPI ALLUVIAL VALLEY

Rochelle Brazas Bailey, Randall J. Rousseau, Emile Gardiner, and Jason C. Mack¹

ABSTRACT

Introduction

Short rotation woody crops, such as willows (*Salix* spp.), continue to be examined as biomass species because of their fast growth, ease of vegetative propagation, and ability to be coppice regenerated. Black willow (*Salix nigra* Marsh.) fits well into a biomass program for the southern United States because of its ability to grow on marginal agricultural sites that are poorly drained.

Methods

In 2008, Mississippi State University and the U.S. Forest Service Center for Bottomland Hardwood Research entered into a joint venture to develop genetically superior black willow clones as feedstock for a growing bioenergy and biofuels market. An initial collection of 113 clones was made in the fall and winter of 2008-2009 from five geographic areas. Following 1 year in stoolbed culture, the clonal material was placed into a series of genetic screening trials in 2010 and 2011. A total of four clonal screening trials, two in 2010 and two in 2011, were established on various sites in Mississippi. These trials were annually measured for total height, diameter, and number of stems. Thus, up-to-date measurements include the first 3 years for the 2010 and 2011 trials. However, as a method of hopefully getting ahead of the selection process, the age-two data was used to select clones for the 2012 black willow clone test. In 2013, the age-three data from the 2010 screening trial and the age-two data from the 2011 screening trial were used to select clones for the 2013 black willow clonal test.

Results

Growth from age-three trials indicated that spacing could be reduced from 54 square feet per tree to 18 square feet per tree, thus in 2012 the first clone test employed a 3 foot x 6 foot spacing. Growth was impressive the first year with the crowns closing toward the end of the first growing season. Age-two measurements continued to show increased growth, even at this tight spacing. In addition, the 2013 clone tests used this same spacing.

All of the screening and clone tests to date have shown excellent survival rates. However, in both the 2010 and 2011 trials, a limited number of sandbar willow (*Salix exigua* Nutt.) clones were included. Survival of the sandbar clones in the 2010 and 2011 test sites located near Prairie, MS exhibited nearly total mortality. It was discovered that the mortality was due to the acidic soil (pH of 4.6) as compared to the Stoneville and Hollandale test sites (which exhibited a soil pH of 7.0). Sandbar willow has also been determined to lack the rooting capacity of black willow, which may relate to its

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low growth and survival rates in these tests. Additionally, sandbar willow has a propensity to root sucker, thus creating problems in stoolbed culture.

The age-three data of the 2010 screening trial showed significant geographic source and clone differences. However, as age increased, geographic variation diminished while clone variation increased. Age-two data from the 2010 and 2011 trials were used to select the top 25 performing clones to be established in the 2012 black willow clone test. All geographic sources were present in this trial, although there were a higher number of clones representing the Atchafalaya geographic source. The results of the age-two data at Sessums, MS showed clone ATCR 4-4 was the best performing clone, and the poorest performance was from the only sandbar clone included in the test. The top four performing clones from the combined analysis of the 2010-2011 clonal screening trials have also outperformed the majority of the other clones.

Summary

The primary objective of this study was to identify genetically superior clones of black willow for the production of biomass for bioenergy and biofuels. Suitable biomass growth on marginal sites should be achievable using genetically superior black willow. As screening trials and clone tests age, we will be able to determine the viability of black willow as a biomass species for the production of bioenergy and biofuels. While this venture is new, it has shown promise, and different concepts are being examined to determine future steps for increasing gain and viability of black willow as a biomass species for the production of bioenergy and biofuels.

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RECREATION

COMPARATIVE EFFICACY OF MULTIMODAL DIGITAL METHODS IN ASSESSING TRAIL/RESOURCE DEGRADATION

Logan O. Park¹

Abstract.—Outdoor recreation can cause both positive and negative impacts on associated forest ecosystems. Forest recreation trails localize negative impacts to a controlled spatial extent while providing recreation access beyond developed areas and transportation networks. Current methods for assessing extent and severity of trail and proximal resource degradation require onsite expert assessment. The methods are analog—e.g., tape measure—although data may be recorded digitally by handheld global positioning system (GPS) using: (1) spatially sparse discontinuous point sampling, or (2) continuous problem assessment that relies on site-dependent (i.e., not generalizable) condition classing sometimes built upon classes that are not mutually exclusive.

This study evaluated de novo multimodal continuous digital electronic measurement of multiple simultaneous trail data streams and compared the accuracy and effectiveness against point sampling and problem assessment equivalents. Trail surface data were collected from a stratified sample of the Shawnee National Forest in southern Illinois. Each 100-m trail segment was digitally scanned, continuously assessed, and point sampled for erosion, muddiness, and rugosity (lateral/transverse). Multivariate regression modeling indicates that temporal sampling resolution and high frequency motion correction drive digital assessment accuracy. In addition, parallel sensing modalities extend each other and provide needed error correction. This study highlights the need and capability to reduce large-scale trail management cost and field staffing through uptake of digital surveying and assessment techniques. Further implications for research and management will be discussed and equipment will be demonstrated hands-on.

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OPTIMIZED HORSE TRAIL DESIGN FOR ILLINOIS SOIL

C.J. Jones and Logan O. Park¹

Abstract.—One of the fastest growing forms of outdoor recreation is equestrian trail riding. In a study examining long-term trends of use on Forest Service lands, equestrian-based recreation was identified as one of the top five activities experiencing growth. As the numbers of horse riders rise, the economic impact of equestrian recreation can be expected to increase across the country. However, equestrian use has been identified with several negative impacts on trail systems including soil compaction, increased trail width and depth, and reduced surface litter. Evidence shows that the problems associated with recreation impact are likely the result of poor planning and location rather than the type of use alone. Due to the role of landscape characteristics such as wet soils and steep slopes in influencing common trail issues (i.e., muddy sections and eroded treads), landscape considerations during the planning process should be able to prevent most impacts before they happen. Furthermore, observational evidence shows that factors such as trail position, trail slope alignment angle (TSA), grade, water drainage, and type of use are significant determinants in how a trail erodes. If a trail is properly designed, much of the degradation associated with heavy impact might be prevented.

The goal of our research project was to empirically test for the effects of trail design, proximal landscape attributes, and routing characteristics on equestrian trail soil transport and erosion using an experimental trail built on the University Farms south of Carbondale, IL. Although several recent observational studies identified factors driving erosion in pedestrian trails, they lacked fully controlled designs. In this study, we seek to overcome this basic limitation in our understanding. Researchers and practitioners will be able to use the results of this study to design and maintain trails at lower cost, improved safety, and higher ecological resilience. The trail's close proximity to Southern Illinois University's Horse Center allowed for easy access by participants and gave the study a closely controlled group of users. Because of the erosive nature of the area's soil, the study's results are applicable to a wide range of soil types.

To prevent ambiguous causes of erosion associated with existing trail usage from influencing the study, the study trail was designed de novo in ArcGIS. Raster maps of the study area were added to represent elevation, water bodies, aspect, and structures for analysis. A view shed analysis was also performed on a nearby gun range for extra safety when designing the trail route. Once designed, the mile-long trail was divided into 5- m segments for monitoring. Based on established research, five primary design factors identified with influencing erosion were tracked for each segment: trail slope alignment angle, landform position, percent canopy cover, substrate modification, and benching. In addition, each trail segment's average slope, sinuosity, and azimuth were recorded as independent variables, and horse use was tracked over the course of the study.

Each segment was then monitored for increased erosion, muddiness, rugosity, and soil compaction. Measurements were taken on the first weekend of the month and immediately after rain events where total liquid precipitation exceeded 1.0 inch in

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24 hours. Because water has been identified as a primary factor in erosion along trail surfaces, monitoring the trail after rain events was critical. Monitoring the trail's condition after extensive rain was also important for maintenance during the experiment, because any downed trees or other obstructions had to be cleared quickly to prevent unwanted or unsafe behaviors by riders. For instance, a downed tree blocking part of the trail would cause bypassing, thus widening the trail and influencing the results of the study.

Data collection is ongoing, but early analyses indicate useful correlation of soil conductivity (as measured by human-portable electromagnetic field induction instruments) to primary trail design factors previously linked to active erosion (e.g., trail-slope alignment, landform position). Work is ongoing to use multivariate analysis of variance, or MANOVA, to compare the independent groups to the dependent variables so that we can determine if there are any differences in variance between each independent factor and more than one of the dependent variables. We predict that segments optimizing values for the design variables will erode at lower rates than other segments. Based on the current research, we also predict that trail segment slope, TSA, and landform position (as a proxy for drainage) will be the most influential factors in determining how each segment erodes.

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TESTING TOOLS FOR OUTDOOR RECREATION, ENVIRONMENTAL EDUCATION, AND STEWARDSHIP: ALLOWING CHILDREN TO CHOOSE THE RULES

Laura E. Baird and Logan O. Park¹

ABSTRACT

Introduction

Engaging children in natural settings enhances learning, promotes early childhood development, and makes use of protected natural areas. Unfortunately, many schoolchildren, especially from economically disadvantaged areas, lack support for environmental education (EE) to develop skills and attitudes that increase rates of appropriate outdoor behaviors. Improved access to environmental education should reduce the amount of resource degradation that occurs when children visit protected natural areas. Many of these children's depreciative behaviors can be classified as uninformed or unintentional (Hendee et al. 1990), implying that guided critical thinking *before they visit* will enable them to make better choices when outdoors and raise awareness of situations that otherwise result in such behaviors (Roggenbuck 1992). This research project develops a model program of replicable, low-cost, widely accessible critical thinking activities and materials designed to directly address this problem.

Providing children with previsit activities improves cognition during a trip and can supply the proper preparation to outline expectations of visitor behavior (Griffin and Symington 1997). However, many protected areas in southern Illinois either do not offer previsit materials or their materials do not address outdoor ethics and behavior expectations. The Girl Scouts of Southern Illinois (GSSI) is one group that has not been included in a widespread EE program. GSSI, with more than 14,000 scouts, many of whom come from economically depressed areas, agreed to be a partner in our research project. Additional partners included Camp Ondessonk, a private residential camp in Ozark, IL; the Leave No Trace Center for Outdoor Ethics; and the Crab Orchard National Wildlife Refuge.

GSSI scouts visit protected natural areas such as Crab Orchard National Wildlife Refuge in Makanda, IL, throughout the year. In the summer, Crab Orchard hosts hundreds of girls aged 7-18 years for a series of weeklong residential camps at Camp Cedar Point. The effects of these visits on the refuge have not been documented, and there are concerns that, despite a short session in which the counselors go over the rules for the week, the girls are not adequately prepared to interact with the protected natural areas within the refuge.

Methods

The study had two objectives: (1) to engage campers with a variety of small-group interactive workshops to cultivate resource-appropriate behaviors, and (2) to assess the effectiveness of three specific engagement methods. Eighty-five Brownie (grades 2 and 3) and Junior level (grades 4 and 5) Girl Scouts participated in small workshops held during the first full day of each of five weeklong

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summer residential camps at Crab Orchard. In addition, 68 coed campers participated in a workshop held during the first full day of a 4-day private residential camp at Camp Ondessonk. An additional 72 campers across the two study sites participated in a control treatment, a pre-camp survey and a post-camp survey without any workshop.

Original activities involving a discussion-based journal, handmade wristband, and interactive games were conducted during these workshops. All activities were developed according to the North American Association of Environmental Education (NAAEE) Guidelines for Excellence. Surveys were administered before and after the workshops to determine camper attitudes; a followup survey was administered at the end of camp. Camper behavior on a hike was observed as well.

Results

It was hypothesized that a combination of all three methods of engagement would be the most effective approach in reducing depreciative behavior and changing attitudes toward resource protection by addressing multiple motivations. Overall, campers' attitudes grew more favorable toward resource protection after any of the activity combinations; the most significant increase followed the treatments with all three engagement methods. The post-camp followup survey indicated a slight decrease in scores, although averages were still higher than those of the pre-program survey. Depreciative behavior levels on the hikes were correlated with the post-session survey scores, although precipitation also had an effect on behavior.

Conclusions

The most effective combination of activities—the journal, bracelet, and games—will be modified into a “kit” for regional use by scouting groups and school groups before attending field trips to protected natural areas. When used as a set of previsit activities, the hands-on program should prevent depreciative behaviors within the park by encouraging environmental stewardship. This project cultivates a population that is environmentally literate and willing and able to translate knowledge and skills into decisions and actions when outdoors. In addition, because visiting students would be more prepared from the outset, park educators could spend more time interpreting the natural resources and less time explaining and enforcing the rules.

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COMPARISONS OF SEDIMENT LOSSES FROM A NEWLY CONSTRUCTED CROSS-COUNTRY NATURAL GAS PIPELINE AND AN EXISTING IN-ROAD PIPELINE

Pamela J. Edwards, Bridget M. Harrison, Daniel J. Holz, Karl W.J. Williard,
and Jon E. Schoonover¹

Abstract.—Sediment loads were measured for about one year from natural gas pipelines in two studies in north central West Virginia. One study involved a 1-year-old pipeline buried within the bed of a 25-year-old skid road, and the other involved a newly constructed cross-country pipeline. Both pipelines were the same diameter and were installed using similar trenching and backfilling techniques. Erosion was measured from both pipelines at the outfall of waterbars, and sediment losses were expressed on a per area basis to compare the pipeline segments. Average sediment yields per sampling period (i.e., generally individual storm events) were a magnitude larger from the pipeline installed in the skid road than from the cross-country pipeline. Compaction and poor vegetation establishment on the skid road pipeline appear to have resulted in excessive runoff and elevated soil losses, even though the skid road segments were less steep and shorter than the cross-country pipeline segments. Reducing compaction to encourage infiltration and successful vegetation establishment is essential for controlling sediment losses, regardless of the land management activity or type of disturbance.

INTRODUCTION

Natural gas development is undergoing massive expansion in the eastern United States, particularly in the Mid-Atlantic States. Substantial concerns about the social, economic, and environmental effects of drilling and hydraulic fracturing have spurred the initiation of research to address those concerns.

To transport the natural gas extracted from the new wells, pipeline construction has simultaneously been expanding. For example, 1,854 km of pipeline were constructed in 2005, and 7,662 km were planned for construction in 2008 (Energy Information Administration 2009). More recently, in 2012, construction length was projected to be more than 5.5 times that constructed in 2011 (Smith 2013). Because of their substantial length, pipelines will have more spatially extensive impacts than those associated with drill pad development; however, there has been little research into their effects. Fragmentation may be the most common environmental concern associated with pipelines, but many other concerns also exist, including water quality effects from pipeline leaks or ruptures, stream sedimentation, and creation of corridors that could speed the spread of invasive species.

Best management practices (BMP) developed by the oil and gas industry exist to help control these adverse effects, but they do not eliminate all undesirable outcomes. One BMP recommended to address some of the concerns associated with pipelines is to install them in existing corridors, such as within existing transmission (power or pipeline) lines or roadways or in the rights-of-ways of those corridors. Although this construction technique is broadly accepted as effective, there are few data to illustrate or support its advantage.

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Consequently, this paper compares erosion losses from two previously completed case studies in north central West Virginia. One involves a pipeline installed in an existing, but no longer used skid road, and the second involves a cross-country pipeline installed outside an existing transportation corridor. The two case studies were typical of their respective types of installations. They were performed 2 years apart with slightly different experimental designs, but essentially the same equipment and sampling approaches.

METHODS

Both studies were performed on pipelines located in Tucker County, West Virginia. The two study locations are about 11.8 km apart. The area is characterized by relatively steep hillsides that support mixed mesophytic hardwoods. Precipitation in the area averages about 130 cm annually and is distributed fairly evenly throughout the year. Mean air temperature is 9.25 °C; mean growing season (May through October) and dormant season (November through April) temperatures are about 16.2 °C and 2.1 °C, respectively (Edwards and Wood 2011).

Data for the first case study used in this analysis originate from Holz (2009). This study was conducted in the lower Sugarland area of Tucker County. An 8.9-cm-diameter natural gas pipeline was installed during the summer of 2006 by burying it beneath the longitudinal center of a 3- to 4.5-m-wide skid road that had been constructed 25 years earlier. Due to the slope of the hillside, the skid road was built using cut and fill techniques, but it was constructed as a temporary, unimproved road and was used only for removing logs using a rubber-tired skidder (dragging with one end on the road surface); there was no truck traffic on the road. After the skid road was “put to bed” after logging was completed, waterbars were installed to control runoff.

Following backfilling of the pipeline, waterbars were re-installed for water control on the skid road. The skid road surface and cutbank were limed, fertilized, and reseeded by hand; all of the amendments were completed within 2 weeks of installing the pipeline (by October 15, 2006). Lime and fertilizer (10-20-20) rates were 4,483.4 kg/ha and 168.13 kg/ha, respectively. A mixture of 19 native herbaceous and grass species were included in the seed mixture (Table 1); annual rye grass (*Lolium multiflorum*) and partridge pea (*Chamaecrista fasciculata*) served as nurse crops.

Barriers were installed at the base of the skid road where it intersected a county road after the waterbars were installed. However, no barriers were installed at the top of the road where it ended at private land. There was evidence of unauthorized all-terrain vehicle (ATV) use on the skid road between the period of pipeline installation and the beginning of equipment installation for this study. Tire tracks ranging from 30 to 46 cm wide and less than 1 cm deep were evident on the road, particularly on the waterbars. Consequently, at about the time of study-equipment installation, additional barriers were installed at the top of the skid road to eliminate all vehicular use there.

In 2006, the skid road held segments (defined below) that could be visibly separated into those that were densely vegetated and those that were sparsely vegetated. Consequently, two segments of each type were included to represent the overall skid road/pipeline surface conditions. Because densely and sparsely vegetated segments were interspersed longitudinally throughout the skid road, differences in vegetative cover were assumed to be due to factors such as incoming solar radiation rather than

Table 1.—Native seeds and rates applied to the skid road after pipeline installation

Common name	Scientific name	Seeding rate (kg/ha)
Annual rye grass	<i>Lolium multiflorum</i>	16.81
Partridge pea	<i>Chamaecrista fasciculata</i>	16.81
Virginia wild rye	<i>Elymus virginicus</i>	50.44
Rough avens	<i>Geum laciniatum</i>	11.21
False Solomon's seal	<i>Smilacina racemosa</i>	10.09
Heath aster	<i>Aster pilosus</i>	6.73
Riverbank wild rye	<i>Elymus riparium</i>	5.60
Thimbleweed	<i>Anemone virginiana</i>	4.48
Ox-eye sunflower	<i>Heliopsis helianthoides</i>	4.48
Zig zag aster	<i>Aster prenanthoides</i>	3.36
Blue cohosh	<i>Caulophyllum thalictroides</i>	3.36
Black cohosh	<i>Actaea racemosa</i>	3.36
Big leaf aster	<i>Eurybia macrophylla</i>	2.24
Sweet cicely	<i>Osmorhiza berteroi</i>	2.24
Blackberry	<i>Rubus allegheniensis</i>	2.24
Eastern columbine	<i>Aquilegia Canadensis</i>	0.56
Jack-in-the-pulpit	<i>Arisaema triphyllum</i>	0.56
White wood aster	<i>Eurybia divaricata</i>	0.56
Greek valerian	<i>Polemonium reptans</i>	0.56
Total		112.09

soil compaction. Soils in the skid road are Gilpin channery silt loam, which is described as highly erodible, largely due to the steep hillside slope (Losche and Beverage 1967).

Data for the second case study are from Harrison (2011). This study was performed on the Fernow Experimental Forest where a 9-m-wide new cross-country pipeline was constructed during fall 2008 through late spring 2009. Erosion was measured from 15 sections of pipeline in the Harrison (2011) study, but data from only three sections were used in this analysis because they were similar in slope to the skid road. The soil associated with this section of the pipeline was mapped and classified from a soil pit excavated immediately adjacent to the pipeline. It was described as a residuum Calvin silt loam soil (Harrison 2011), which is considered moderately erodible.

After the forest overstory was removed and stumps were grubbed from the corridor, the 8.9-cm-diameter pipeline was buried at a 76-cm depth. After the trench was backfilled, waterbars were installed for water control using a trackhoe, and no further mechanical traffic was permitted on the pipeline. During the last few days of April 2009, the pipeline was seeded with a mixture of native seeds (Table 2). Annual rye grass and partridge pea, along with oats, again were used as nurse crops. Fertilizer (10-20-10) was applied at a rate of 672 kg/ha, and lime and uncut straw mulch were applied at 4.48 metric tons/ha each.

In both studies, erosion was measured from segments of the skid road or pipeline defined by waterbars (i.e., the area extending from crest to crest of adjacent waterbars) (Fig. 1). Physical characteristics of each of the segments are given in Table 3. The slopes of the skid road pipeline segments are less than those of the cross-country segments because transportation requirements and skid road BMPs

Table 2.—Native seeds and rates applied to the cross-country corridor after pipeline installation

Common name	Scientific name	Seeding rate (kg/ha)
Annual rye grass	<i>Lolium multiflorum</i>	33.6
Partridge pea	<i>Chamaecrista fasciculata</i>	2.24
Oats	<i>Avena sativa</i>	3.36
Canada milkvetch	<i>Astragalus canadensis</i>	2.24
Little bluestem	<i>Andropogon scoparius</i>	3.36
Autumn bentgrass	<i>Agrostis penennans</i>	4.48
Deer tongue	<i>Panicum clandestinum</i>	6.72
Total		56.0

Table 3.—Physical characteristics of the pipeline segments

Section	Slope	Length	Area	Aspect	Vegetative cover ^a
	(%)	(m)	(m ²)		(%)
Skid road pipeline					
Segment 1	13.57	32.50	121.61 ^b	NE	16.45
Segment 2	12.44	25.99	84.02	E	20.20
Segment 3	13.32	32.37	125.24	NE	82.13
Segment 4	12.75	28.60	129.32	NE	77.06
Cross-country pipeline					
Segment 1	26.8	18.94	119.69	NW	26.46
Segment 2	20.7	25.58	143.32	NW	47.45
Segment 3	18.6	19.44	110.64	NW	29.25

^aPercent vegetative cover determined using photographic image analysis techniques described in Holz (2009) for the skidroad pipeline and in Harrison (2011) for the cross-country pipeline.

^bSkid road pipeline area includes the road surface and the cutbank because both can contribute sediment and runoff to the waterbar.

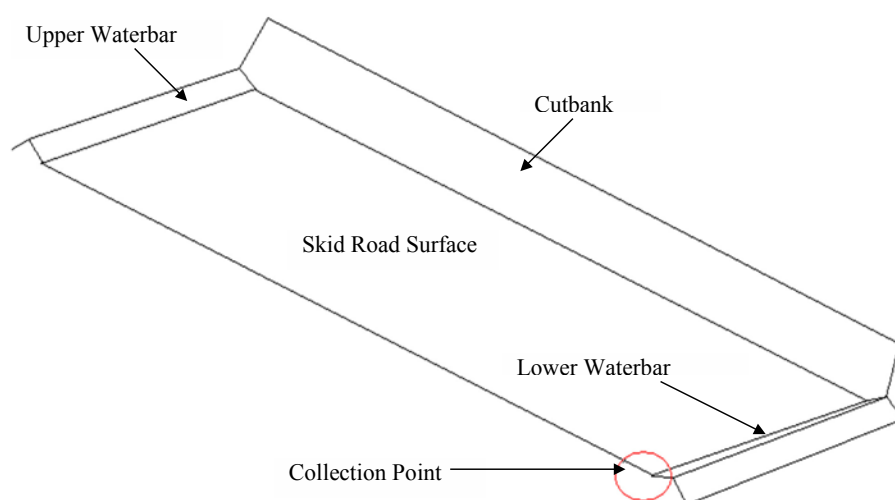


Figure 1.—Schematic illustrating a pipeline segment that extends from crest to crest of adjacent waterbars. The flume is installed at the base of the interior side of the downslope waterbar (i.e., the collection point). For the pipeline installed in the skid road, as shown here, the segment included the cutbank as well as the road surface. For the cross-country pipeline, the segment included only the surface of the disturbed corridor.



Figure 2.—Photograph showing a sediment collection device at a waterbar outlet. Runoff and sediment collected in the flume were diverted downslope into a collection tank by gravity. Samples were collected from a spigot at the base of the tank. Photo by Pam Edwards, U.S. Forest Service.

necessitate gentler slopes. The cross-country segments are shorter because the steeper slopes require more closely spaced waterbars. The contributing areas of the two types of segments are similar because the cross-country pipeline right-of-way is wider than the skid road (including the cutbanks).

A small metal or wooden flume was installed at the outlet of each waterbar, and the soil and flume were sealed together using hydraulic cement. Drainage and associated eroded soil were diverted from the skid road/pipeline section by the waterbar through the flume and then transported by gravity drainage to a collection tank downslope of the waterbar (Fig. 2).

Samples were collected from the pipeline in the skid road from July 25, 2007, through December 12, 2007, and then from April 11, 2008, through May 28, 2008. From December to April, the inlet pipes were disconnected from the tanks to avoid freezing and breakage (Holz 2009). Sampling from the cross-country pipeline began June 12, 2009, following equipment installation after seeding and mulching. Sampling continued for one full year (Harrison 2011). Tanks were not disconnected during the winter, but there was little melt from about mid-January through March. In both studies, sampling was primarily performed after individual precipitation or melt events; however, some collection periods included multiple events when they occurred on weekends or holidays.

Three replicate samples, approximately 1L each, were collected from each tank in both studies per sampling event. Before and during sample collection, the contents of the tanks were stirred

with a long-handled brush to keep the solid materials suspended and help ensure each sample was representative of the tank contents. The volume of water present in each tank also was recorded to the nearest gallon before collecting the samples, using the volume demarcations printed on the side of each tank. Tank contents were emptied after sample collection.

All samples were analyzed for sediment concentrations using U.S. EPA method 160.2 (Keith 1991). This procedure involves vacuum filtering samples to separate solids from water. However, in the case of the cross-country pipeline, some of the samples collected during the first several months after pipeline construction had solid concentrations that were too high to allow direct filtration. These samples were centrifuged before filtering to separate most of the solids from the water. The organic portion of the samples was then removed from the mineral sediment by combusting the filters and the centrifuged solids at 550 °C until they reached a constant post-combustion weight (a minimum of 2 hours). Tank volumes were applied to the mineral sediment concentrations to obtain total mineral sediment losses from each section by sampling period and for the entire study.

In both case studies, percentage of vegetative cover was determined using ArcGIS and image analysis of digital photographs. Vegetative cover of the cutbank was included in the analyses for the skid road corridor because both the road surface and cutbank could contribute sediment to the associated segment. The field and image analysis techniques follow those described in Bold et al. (2010) and are detailed for each case study in Holz (2009) and Harrison (2011). In brief, the entire area of each corridor section was photographed using a digital camera mounted on a prism pole. This was accomplished by dividing each section into multiple subsections using PVC-pipe frames and photographing each subsection individually. The percentage of vegetative cover in each subsection was quantified after developing and validating signature files capable of isolating green shaded pixels (vegetation) from all other pixel colors in each photograph. Total percentage of vegetative cover for each section was determined from the results of all the subsections.

RESULTS AND DISCUSSION

The cross-country pipeline segments all had lower sediment yields than the pipeline segments installed in the skid road (Table 4). The overall mean sediment loss from segment 1 of the cross-country pipeline (10.44 kg/ha) was close to the lowest mean loss from the skid-road pipeline (14.34 kg /ha for segment 4), but the former still was lower by almost one-third. Additionally, sediment yields from cross-country segments 2 and 3 were a unit of magnitude lower than any of the skid road pipeline segments. The mean sediment loss across the four skid road pipeline segments over the 8 months of measurements (27.1 kg/ha/8 mo) was more than 4.5 times greater than the mean sediment loss across the three cross-country pipeline segments over a full year (5.65 kg/ha/yr). Because the numbers of runoff events and sampling period lengths were not the same for the two studies, the focus of this analysis is not on total losses (i.e., the right hand column of Table 4). However, given these results, it is not surprising that the total sediment losses over the shorter study (i.e., from the skid road segments) were greater from each of the skid road pipeline segments.

Because precipitation affects runoff and erosion and the two studies were performed during different time periods, the influence of precipitation on the results deserves some attention. Not surprisingly, both Holz (2009) and Harrison (2011) reported that rainfall intensity (i.e., 30-minute intensities) was the most important precipitation variable, of the several examined, for explaining sediment

Table 4.—Mineral sediment load summaries for the two pipelines

Section	Sediment load per sample period		Mean load across all segments ^a	Total load for each segment ^a
	Mean	Standard error		
	-----kg/ha-----		(kg/ha/8 mo)	(kg/ha/8 mo)
Skidroad pipeline				
Segment 1	48.58	19.30		1,797.6
Segment 2	32.87	9.15		1,183.3
Segment 3	15.48	4.21		572.9
Segment 4	14.34	3.13		530.5
			27.1	
Cross-country pipeline			(kg/ha/yr)	(kg/ha/yr)
Segment 1	10.44	3.18		396.9
Segment 2	3.08	1.12		110.8
Segment 3	3.42	1.09		116.2
			5.65	

^aMean loads and total loads for the skid road pipeline are expressed for an 8-month time period, while those for the cross-country pipeline are over a full year.

concentrations. Other investigators also reported rainfall intensity as important in sediment losses (Bold et al. 2010, Reid and Anderson 1999). However, the years associated with these studies were not characterized by the occurrence of extreme individual events or by unusual (high or low) precipitation totals (U.S. Forest Service, unpublished data). Total precipitation for the 8 months of the study involving the skid road pipeline was about 77.3 cm (Holz 2009), and total precipitation during the year of the cross-country pipeline study was about 128.5 cm (Harrison 2011). If the 8-month study were normalized to a yearlong period, the resulting annual precipitation (116.0 cm) would be similar to, but still less, than the Harrison study (128.5 cm). Consequently, the greater loads from the pipeline associated with the skid road do not appear to be attributable to differences in precipitation between the two years.

Loadings are the product of sediment concentrations and the runoff volume in the collection tank at each sampling event; therefore, examination of these two variables can provide information about whether the greater loadings (kg/ha per sampling event) from the skid road pipeline were attributable to greater runoff, greater erosion rates (i.e., sediment concentrations), or the combination of both. Overall, differences in sediment losses from the skid road segments appear to be controlled primarily by erosion rates rather than runoff. Mean runoff volumes were similar among skid road pipeline segments; indeed, segments 1 and 2 had lower mean total runoff values than segments 3 and 4 (Table 5). By comparison, segments 1 and 2 had much higher mean sediment concentrations (Table 5) and higher loadings than segments 3 and 4 (Table 4).

So what drives the differences in erosion rates among segments on the skid road pipeline that are relatively close to one another and are influenced by the same general soil characteristics? In this situation, the density of vegetation (Table 3) appears to be an important factor in determining the degree of erosion that occurs. Although only two replicate segments in each of the dense and sparse vegetative cover categories cannot be compared statistically, sediment concentrations (mg/L) and

Table 5.—Mean sediment concentrations and runoff volumes by location and pipeline section

	Sediment concentration		Runoff volume		
Section	Mean	Standard error	Mean	Standard error	Total
	-----mg/L-----		-----L-----		
Skid road pipeline					
Segment 1	2,039.8	1094.5	366.5	24.8	13,561
Segment 2	698.0	190.4	322.3	32.1	11,602
Segment 3	457.4	118.9	378.3	24.8	13,997
Segment 4	429.5	85.2	381.8	24.4	14,125
Cross-country pipeline					
Segment 1	322.7	89.4	311.3	43.3	11,829
Segment 2	280.3	64.5	83.1	16.9	2,993
Segment 3	467.4	124.8	37.1	8.9	1,262

loadings (kg/ha) were clearly greater from segments with low vegetative cover (segments 1 and 2) (Tables 4 and 5). Greater sediment losses with sparse vegetative cover are consistent with scientific literature. Many other studies found that effective erosion control occurs only when cover reaches 50 to 75 percent (Gifford 1985, Gutierrez and Hernandez 1996, Loch 2000, Orr 1970, Quinton et al. 1997, Snelder and Bryan 1995).

Sediment concentrations from all three cross-country pipeline segments were relatively similar to those from the densely vegetated segments of the pipeline in the skid road (Table 5), even though the percent cover values on the cross-country segments were much lower (Table 3) and were less than the 50 to 75 percent levels needed to control erosion. This finding may seem at odds with the statements above about the importance of vegetation and erosion control, particularly because the slopes of the cross-country segments were greater. However, the higher runoff volumes and more erodible soil associated with the skid road pipeline appear to be of greater consequence than vegetation differences between the two sites.

Segment 1 of the cross-country pipeline is somewhat of an anomaly in terms of runoff volumes compared to the other two cross-country segments. It had 3.7 to 8.4 times more runoff compared to segments 2 and 3 (Table 5). For segment 1, the elevated runoff alone appears to be the cause of the greater sediment yields (Table 4) because the concentrations are comparable among all three segments. The runoff volume for segment 1 is believed to be attributable to its location at the head of the bench, immediately below a much steeper section of pipeline. High runoff volumes were reported for other steeper segments located immediately upslope of segment 1. These runoff volumes were attributed to the presence of a fragipan-like layer at a 50-cm depth, which was identified from a soil pit excavated adjacent to the pipeline within those steeper segments (Harrison 2011). The waterbars were believed to have intercepted subsurface drainage that was diverted along the dense layer (Harrison 2011). We speculate that upslope subsurface flow diverted from this layer contributed to the elevated runoff from segment 1 even though soil mapping did not find the layer to extend into the bench (Harrison 2011). However, it is likely that much of the runoff became emergent near or in the waterbar of segment 1 so that it did not enhance erosion from the entire face of the segment. This response is supported by the elevated runoff and sediment loads with no concomitant elevation of sediment concentrations.

MANAGEMENT IMPLICATIONS

Where there were high runoff volumes, the source and pathway of the drainage influenced sediment yields. On the skid road segments, overland flow appeared to be a dominant runoff mechanism. Surface soil compaction limited infiltration, as evidenced by visible sheetflow and concentrated overland flow during and following all but the smallest precipitation and snowmelt events. Rills also developed where surface runoff concentrated. These conditions elevated soil losses.

Dense vegetation clearly helped control erosion from the skid road segments, but erosion was still higher than where waterbar discharge originated primarily from subsurface flow. There was evidence of surface erosion on the cross-country pipeline, but interrill erosion appeared to predominate: small soil pedestals were evident, but rill development was limited and surface runoff was not nearly as visible during events as on the skid road segments. Thus, vegetation density contributes to erosion control, but overland flow ultimately may trump much of the advantage that vegetative cover provides for controlling soil loss on a compacted site.

The comparisons of these two studies indicate that taking steps to increase or maintain high infiltration rates also can provide substantial benefit. Although using existing infrastructure reduces disturbance and fragmentation, this BMP may provide little advantage from an erosion and sediment control perspective if infiltration issues on existing corridors are not addressed. For example, while the excavation for pipeline installation may have increased infiltration within that narrow width, infiltration may have been increased much more by ripping the entire road width just before installing the pipeline. Ideally, ripping at the time the original skid road was closed out probably would have provided substantial benefit for the 25 years before pipeline installation. But even in the absence of ripping after skid road closure, soil ripping at the time of pipeline installation could have helped counter the legacy soil compaction and infiltration problems and likely would have improved contemporary vegetative establishment, which in turn could have contributed to better soil erosion control.

Unauthorized use of pipelines, especially by off-road vehicles, also must be controlled to maintain erosion at low levels. ATVs increase compaction, tear out vegetation or decrease its vigor, and create wheel ruts due to wheel slip and braking patterns. Wheel ruts can serve as concentrated flow channels that exacerbate erosion. Controlling unauthorized use may be one of the more difficult long-term tasks on steep, accessible corridors, because these areas attract users interested in hill climbing.

CONCLUSIONS

Sediment loads were greater from the pipeline in the skid road even though the cross-country pipeline was steeper and had only sparse vegetative cover. Greater soil compaction and poor infiltration on the skid road are believed to be the primary reasons for the greater soil yields there. Although the use of existing infrastructure for pipeline installation provides environmental benefits, such as reduced land fragmentation, the comparison of these two pipelines illustrates the importance of maintaining good infiltration on all types of corridors that have not been used for or are no longer used for transportation.

ACKNOWLEDGMENTS

The authors thank Linda Tracy (retired) and Will Wilson, Monongahela National Forest geologists, for their assistance with implementing both of these studies. We also are indebted to Stephanie Connolly, Monongahela National Forest soil scientist, for describing the soils along the cross-country pipeline.

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FOREST BIOMETRICS

ESTIMATING BARK THICKNESSES OF COMMON APPALACHIAN HARDWOODS

R. Edward Thomas and Neal D. Bennett¹

Abstract.—Knowing the thickness of bark along the stem of a tree is critical to accurately estimate residue and, more importantly, estimate the volume of solid wood available. Determining the volume or weight of bark for a log is important because bark and wood mass are typically separated while processing logs, and accurate determination of volume is problematic. Bark thickness is known to vary with species, tree diameter, and location along the stem. This paper examines the bark thicknesses of four Appalachian hardwoods: red oak (*Quercus rubra*), white oak (*Quercus alba*), yellow-poplar (*Liriodendron tulipifera*), and sugar maple (*Acer saccharum*). Using bark thickness measurements from multiple sites, average bark thickness by stem diameter and sample height location were determined. In addition, models were developed that predict bark thickness based on measured diameter outside the bark.

INTRODUCTION

Tree bark is a valuable residue resulting from the manufacture of wood products such as lumber, veneer, or pulp. Bark is used for a wide variety of products including mulch, charcoal filters, as a source of tannins for leather and textile processing, or as boiler fuel. Knowing the thickness of bark along the stem is critical in order to accurately estimate bark residue and, more importantly, estimate the volume of the solid wood contained in a log. Bark thickness also can serve as an indicator of tree quality. Because bark is not typically mixed with wood for most products, determining the volume or weight of bark for a log is important (Stayton and Hoffman 1970). Complicating matters is the fact that tree bark thickness is known to vary with species, tree diameter, and with location along the stem (e.g., the height above ground) (Forbes 1955). Measuring the volume of bark once it has been removed from the log can also present difficulties for at least two reasons: (1) using bark weight is problematic as the moisture content will vary depending on log yard conditions, season, and species; and (2) the volume of bark is difficult to determine due to variations in bark particle size and compaction. In addition, physically measuring bark on trees or logs is economically prohibitive due to the amount of time and effort it would require. While bark gauges exist that allow seasoned users to obtain an approximation of bark thickness at a specific location along the log, they only give results for those specific locations from which the stem thickness for an entire section is then extrapolated. Also, employing bark gauges properly requires experience and a feel for the instrument (Brack 2012) as even experienced users occasionally penetrate into the cambium layer. Furthermore, the pressure applied to take the measurement varies from species to species and is not necessarily consistent within a species.

Over the past 40 years, several studies have been conducted that have sought to predict bark thickness and volume of available hardwood. Stayton and Hoffman (1970) developed a model for sugar maple (*Acer saccharum*) to predict average bark thickness for a whole tree based on the average tree diameter. Stayton and Hoffman (1970) based their model on diameter measurements at

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1-, 8-, and 16-foot points up the stem. Their resulting model had an R^2 of 0.51 using the following equation:

$$\text{AverageBarkThickness} = 0.1986 + 0.0206 * \text{AverageTreeDiameter}$$

Their study was based on a sample of 90 trees. The average bark volume within the sample was 7.38 cubic feet, or approximately 13 percent of the merchantable stem volume. Stayton and Hoffman (1970) also found that bark thickness generally decreases with increased height. However, they were unable to develop equations capable of accurately predicting bark thickness at specific stem heights. Indeed, their research was complicated by sugar maple having four distinct bark types (platy, shell, compound, and ropy), making bark thickness prediction difficult (Sajdak 1968).

Other researchers examined multiple species. For example, Koch (1971) sought to develop models capable of predicting bark volume and thickness for yellow-poplar (*Liriodendron tulipifera*), red oak (*Quercus rubra*), black oak (*Quercus velutina*), and scarlet oak (*Quercus coccinea*), as well as red maple (*Acer rubrum*) in West Virginia. Samples for red oak and yellow-poplar came from two sites in West Virginia while the other species came from a single site in West Virginia. Koch collected approximately 60 sample trees for each species from each site. Using these samples he determined bark factors or ratios between diameter inside bark (DIB) and diameter outside bark (DOB). DOB is the measurement of the diameter to the outside edge of the bark while DIB is the measurement of the diameter to the inside of the bark. Given that logs are not perfect circles, diameter tape measures or calipers are used to measure DOB. Using the ratios allows an average bark thickness and volume to be calculated for the merchantable portion of a tree.

Fowler et al. (1997) examined multiple oak species in Michigan and developed a series of bark factors. Their measurements and analyses were based on 53 red oak, 23 black oak, and 28 white oak (*Quercus alba*) trees. Unlike Koch (1971), Fowler et al. (1997) used linear regression methods to determine a factor for each species from which bark thickness can be calculated. Red oak yielded the best coefficient of determination ($R^2 = 0.418$) for all models. White oak resulted in the weakest correlation ($R^2 = 0.014$), while black oak bark had an $R^2 = 0.270$. Thus, overall, the bark factors developed by Fowler et al. (1997) explained only a relatively small portion of the bark thickness variance observed within the samples.

One of the most comprehensive examinations of bark thickness was accomplished by Hilt et al. (1983) in which the authors studied 10 hardwood species. Their study sampled approximately 200 trees per species. The data used by the authors was originally collected in the 1960s for the development of log and tree grades for hardwoods (Hanks et al. 1980). As such, the authors note that “mostly larger trees were analyzed in the study.” The trees were bucked in 8- and 16-foot long logs and the DIB measured at the small end of each log. The number of measurements per tree depended on tree size, however, all trees had at least two DIB and bark thickness measurements, but no tree had more than seven measurements. A linear regression analysis was conducted for each species. The resulting bark factor equations had the form of:

$$DIB / DOB = \beta_0 + \beta_1 * DOB / DBHOB$$

where DBHOB is the diameter at breast high measured outside the bark. Although no coefficient of determination was reported for their analyses, the authors did test to see if the β_1 term was

Table 1.—Location of log sample sites and number of samples selected

Species	Location	Elevation (feet)	Number of trees	Number of samples
Yellow-poplar	WVU Experimental Forest, WV	2300	33	505
Yellow-poplar	Camp Creek State Forest, WV	2600	33	482
	Total		66	987
Red oak	WVU Experimental Forest, WV	2300	33	355
Red oak	MeadWestvaco Forest, Rupert, WV	3200	33	423
	Total		66	778
White oak	Fernow Experimental Forest, WV	2936	32	497
White oak	Fayette County, Site 1, WV	2451	15	145
White oak	Fayette County, Site 2, WV	2425	16	276
	Total		63	918
Sugar maple	Fernow Experimental Forest, WV	2438	34	449
Sugar maple	MeadWestvaco Forest, Rupert, WV	3451	31	466
	Total		65	915
All species	Total		260	3598

significantly different from zero. In all species except black cherry (*Prunus serotina*), southern red oak (*Quercus falcata*), and American beech (*Fagus grandifolia*), the β_1 term was found to be significantly different from zero ($\alpha = 0.05$).

The findings presented in these past studies were typically based on four bark thickness measurements per tree. In addition, for some studies the number of trees sampled was small. The research presented in this manuscript used a greater number of bark thickness measurements per tree. The findings of bark thickness measurements and modeling of four common hardwood species (yellow-poplar, red oak, white oak, and sugar maple) from the central Appalachian region are presented in this manuscript.

METHODS

Recently, a series of yellow-poplar, red oak, white oak, and sugar maple log defects were collected and studied to determine the relationships among external log defect indicators and internal defect manifestations (Thomas 2008, Thomas 2009b, Thomas 2012, Thomas²). Log defects are imperfections that appear on the surface of the log and indicate the presence of an interior defect. Such defects decrease the grade and value of the log as well as the grade and value of the lumber and products that can be obtained from the log. A total of 66 yellow-poplar, 66 red oak, 63 white oak, and 65 sugar maple trees were randomly selected from sites in West Virginia. On these sites, the total number of trees available for harvest was determined. Depending on the number of trees available, a selection factor was determined which dictated which trees would be selected (i.e., every third tree). In some instances, the forester in charge of the harvest chose to leave one of the selected trees and would pick a substitute tree. From the 260 trees sampled in this study, a total of 3598 bark measurements were recorded (Table 1).

² Thomas, R.E. [N.d.]. Predicting internal sugar maple (*Acer saccharum*) log defect features using surface defect indicator measurements. Manuscript in preparation. On file with author.

Table 2.—Comparison of sample measurements by species and sampling location

Species	Site	Mean bark thickness	Mean DOB ^a	Mean ring count
-----Inches-----				
Yellow-poplar	WVU Experimental Forest	0.54	11.21	51.4
	Camp Creek State Forest	0.55	9.75	58.5
	Overall	0.54	10.49	54.9
	Difference significant	Yes	Yes	Yes
Red oak	WVU Experimental Forest	0.33	11.04	58.7
	MeadWestvaco, Rupert, WV	0.32	13.73	77.6
	Overall	0.33	12.27	67.3
	Difference significant	Yes	Yes	Yes
White oak	Fernow Experimental Forest, WV	0.34	12.96	82.0
	Fayette County, Site 1, WV	0.36	13.67	127.7
	Fayette County, Site 2, WV	0.31	11.01	64.0
	Overall	0.33	12.49	83.9
	Difference significant	Yes	Yes	Yes
Sugar maple	Fernow Experimental Forest, WV	0.28	11.63	64.2
	MeadWestvaco, Rupert, WV	0.28	11.90	65.6
	Overall	0.28	11.77	64.9
	Difference significant	No	Yes	Yes

^aDOB=diameter outside bark.

The trees were originally harvested for defect studies where a random set of defects was selected from each log. The goal of this selection was to obtain as many as four defects of each type from each log. The height of the defect up the stem as well as the diameter of the defect on a given log sample were not used as decision criteria in the defect selection process. Once the defects were selected, the log diameter inside the bark and the bark thickness were measured to the nearest 1/16 (0.0625) of an inch. Log diameter and bark thickness were measured along the minor and major axes (minimum and maximum diameter) of the sample and were averaged. The defect areas were bucked from the logs at a minimum of 8 inches above the defect indicator. Although the measurements were taken near defects, they were taken well beyond the defective area. In addition, the growth rings were counted for every defect sample used. All defect samples missing bark were excluded from this study.

Statistical Methods

Using R (R Foundation for Statistical Computing, Vienna, Austria), a number of statistical tests were used to determine if there were any significant differences for bark thickness among sample sites, DOB, or ring count measurements for each species. First, Anderson-Darling normality tests (Anderson and Darling 1952) were performed to determine if the data were normally distributed ($\alpha = 0.05$). The test revealed that none of the measurements for bark thickness or diameter outside bark from any of the sites or species were normally distributed. Due to the non-normal distributive nature of the data, Wilcoxon rank sum tests (Wilcoxon 1945) were used to determine if there were significant differences ($\alpha = 0.05$) in the measurements of the red oak, sugar maple, and yellow-poplar data due to collection site. Since the white oak samples were collected from three sites, a Kruskal-Wallis rank sum test (Corder and Foreman 2009) was used to test for significant differences ($\alpha = 0.05$) in measurements among these three sites. Table 2 lists the mean bark thicknesses, mean DOB,

and mean ring count by species and site and indicates if the difference between or among the sites was significant. The comparisons for each species involved three statistical tests, thus the significance level was adjusted for each individual test using the Bonferonni correction (Adbi 2007).

A series of general linear regression analyses were performed on the bark thickness data using the R statistical software. Two models were examined, one that used DOB as a single independent variable, and a second that used sample height, (i.e., the height at which bark thickness was being estimated) and DOB as the independent variables. Bark thickness was the dependent variable for both models. The coefficient of determination and the mean absolute error (MAE) were determined for each species using both models.

The main reason for two models was because of the potential for autocorrelation issues with the DOB only model. Very strong autocorrelation can result in apparent trends when there is no real trend underlying the series. Using a series of Durbin-Watson tests (Hogg and Ledolter 1987), it was determined that the autocorrelation present in the DOB only model was significant ($\alpha = 0.05$), but no significant autocorrelation existed in the DOB and height model. The Durbin-Watson test returns a test statistic valued between 0 and 2. The closer the test value is to 0, the stronger the auto correlation. The closer it is to 2, the weaker the autocorrelation. With our models, the lowest Durbin-Watson value observed with the DOB only model was 1.64, indicating relatively weak autocorrelation.

Using these regression results, Cook's distance (Cook and Weisberg 1982) was used to determine if any individual observation had an influence on the accuracy of the regression results. The number of observations identified as having a significant influence by species was: 11 for yellow-poplar, 16 for red oak, 11 for white oak, and 12 for sugar maple. Most of these observations were taken on thicker bark samples collected near the base of the tree.

Bark thickness data were randomly grouped into model development (66.67 percent) and model testing (33.33 percent) sets for each species. In the regression analyses, diameter outside bark was used as the independent variable and bark thickness as the dependent variable. Using these results, prediction equations were developed and tested using the development data set, and the correlation coefficient and the coefficient of determination were determined.

RESULTS AND DISCUSSION

Significant correlations between DOB and bark thickness were found for all species, indicating that bark thickness can be predicted based on DOB. Also, the overall coefficient of determination between bark thickness and DOB as well as the MAE for each species was determined. In addition, significant correlations were found to exist between DOB, height on stem, and bark thickness for all species. These results are listed in Table 3.

Yellow-Poplar

The yellow-poplar bark thickness prediction model was developed using 987 bark thickness measurements collected from 66 trees harvested from two sites (Table 1). Mean bark thicknesses from the two sites were 0.54 and 0.55 inches, and this difference of 0.01 inch was found to be statistically significant ($\alpha = 0.05$). The differences between the mean DOB and the mean ring counts

Table 3.—Bark thickness correlation results for model development, testing, and overall by species and model

Species	Model	Development		Testing		Overall	
		R ²	MAE	R	MAE	R ²	MAE
Yellow-poplar	DOB (Eq. 1)	0.211	0.075	0.495	0.078	0.223	0.076
	DOB + Height (Eq. 2)	0.264	0.072	0.531	0.077	0.270	0.074
Red oak	DOB (Eq. 3)	0.033	0.059	0.145	0.064	0.028	0.060
	DOB + Height (Eq. 4)	0.039	0.059	0.175	0.064	0.036	0.060
White oak	DOB (Eq. 5)	0.110	0.066	0.346	0.067	0.110	0.066
	DOB + Height (Eq. 6)	0.131	0.065	0.383	0.065	0.137	0.065
Sugar maple	DOB (Eq. 7)	0.045	0.055	0.217	0.057	0.045	0.057
	DOB + Height (Eq. 8)	0.065	0.055	0.242	0.077	0.058	0.056

Table 4.—Average observed yellow-poplar bark thickness and bark thickness calculated using different methods.

Diameter	Average by diameter class	Calculated using Thomas and Bennett (Eq. 1)	Calculated using Thomas and Bennett (Eq. 2) ^a	Calculated using Haygreen and Bowyer equation	Calculated using Hilt et al. equation ^b
<i>Inches</i>					
20 >	--	0.77	0.82	0.92	0.73
18 - 20	0.75	0.73	0.76	0.83	0.73
16 - 18	0.74	0.68	0.70	0.75	0.73
14 - 16	0.65	0.63	0.64	0.67	0.71
12 - 14	0.61	0.58	0.58	0.59	0.67
10 - 12	0.55	0.53	0.52	0.50	0.62
8 - 10	0.51	0.48	0.46	0.42	0.54

^aAssuming height of 16 feet.^bAssuming the diameter at breast height measured outside the bark is 20 inches.

of the two sites also were significantly different at $\alpha = 0.05$ (Table 2). Using the yellow-poplar model development data set, the correlations between DOB, height, and bark thickness were determined, and the following prediction models were developed:

$$\text{BarkThickness} = 0.295 + \text{DOB} * 0.024 \quad (1)$$

$$\text{BarkThickness} = 0.159 + \text{DOB} * 0.0302 + \text{Height} * 0.00015 \quad (2)$$

Overall, the coefficients of determination (R^2) and correlation coefficients (R) were stronger with the second model (Eq. 2) which used DOB and height. In addition, Equation 2 had a lower MAE than Equation 1 (Table 3). Table 4 lists by diameter class the average observed bark thickness, the estimated bark thickness using Thomas and Bennett's models developed in this study (Equations 1 and 2), as well as results based on equations from Haygreen and Bowyer (1996) and Hilt et al. (1983). Averages within diameter class were determined using the overall data set with 2-inch diameter intervals. It is interesting to note that the bark thicknesses calculated using Equation 2 fall between the values determined by the other equations in all but one instance. Values from Equation 1 were within the range of Haygreen and Bowyer's and Hilt et al.'s equations in all but two instances. In these cases, Thomas and Bennett's models predicted values that were slightly less than those calculated by the Haygreen and Bowyer and Hilt et al.

Red Oak

Red oak bark thickness prediction models were developed using 778 thickness measurements collected from 66 trees harvested from two sites (Table 1). Mean bark thicknesses from the two sites were 0.33 and 0.32 inch, and the difference of 0.01 inch between sites (Table 2) was found to be significantly different ($\alpha = 0.05$). In addition, differences between the mean DOB and ring count were significantly different ($\alpha = 0.05$) between the two sites. The difference in mean bark thickness could possibly be explained by the difference in age and measured sample diameter between the sites. Samples from the MeadWestvaco site were, on average, nearly 20 years older and 2 inches wider in diameter. Using the model development set, the correlations between DOB, height, and bark thickness measurements were determined and the following prediction models were developed:

$$\text{BarkThickness} = 0.267 + \text{DOB} * 0.005 \quad (3)$$

$$\text{BarkThickness} = 0.291 + \text{DOB} * 0.0042 - \text{Height} * 0.000046 \quad (4)$$

R^2 and R coefficients for both models were weak but statistically significant ($\alpha = 0.05$). Equation 3 had slightly stronger correlations and lower MAE values than were observed with Equation 4 (Table 3). Of the four species examined, red oak samples had the weakest correlations. Table 5 lists by diameter class the average observed bark thickness, the calculated bark thickness using Equations 3 and 4 by Thomas and Bennett, as well as equations from Haygreen and Bowyer (1996) and Hilt et al. (1983), and the bark factor equation from Fowler et al. (1997). Diameter class averages were determined using the overall data set with 2-inch diameter intervals. As seen in Table 5, the bark thicknesses calculated using the Hilt et al. equation are thickest of all the methods. In addition, the thicknesses calculated by Fowler's equation are less than those observed and those calculated using the Thomas and Bennett's models for DOBs of 16 inches and under. For diameters over 16 inches, bark thicknesses calculated using Thomas and Bennett models were less than bark thicknesses calculated using the other methods.

Table 5.—Average observed red oak bark thickness and bark thickness calculated using different methods

Diameter	Average by diameter class	Calculated using Thomas and Bennett (Eq. 3)	Calculated using Thomas and Bennett (Eq. 4) ^a	Calculated using Haygreen and Bowyer equation	Calculated using Hilt et al. equation ^b	Calculated using Fowler et al. bark factor equation
<i>Inches</i>						
20 >	--	0.37	0.37	0.78	0.60	0.51
18 - 20	0.39	0.36	0.36	0.71	0.60	0.46
16 - 18	0.34	0.35	0.35	0.65	0.60	0.37
14 - 16	0.33	0.34	0.34	0.58	0.59	0.29
12 - 14	0.33	0.33	0.34	0.52	0.56	0.22
10 - 12	0.32	0.32	0.33	0.45	0.52	0.16
8 - 10	0.30	0.31	0.32	0.39	0.46	0.11

^aAssuming height of 16 feet.

^bAssuming the diameter at breast height measured outside the bark is 20 inches.

White Oak

The white oak bark thickness prediction model was developed using 918 thickness measurements collected from 63 trees harvested from three sites (Table 1). The mean bark thicknesses measured from the three sites were 0.34, 0.36, and 0.31 inches with a difference of ± 0.05 inch. The Kruskal-Wallis rank sum tests ($\alpha = 0.05$) showed that all three site variables were significantly different ($\alpha = 0.05$) among the sites (Table 2). This was not surprising, given a 2.6-inch difference in mean DOB and a range of 63 for mean ring count. It is interesting to note that the thinnest mean bark thickness was with the youngest and smallest DOB site, Fayette Site 2, while the thickest mean bark thickness was observed at Fayette Site 1 which had the largest mean DOB and was the oldest site. Using the white oak development data sample set, the correlations between DOB, height, and bark thickness measurements were determined and the following prediction equations were developed:

$$\text{BarkThickness} = 0.221 + \text{DOB} * 0.009 \quad (5)$$

$$\text{BarkThickness} = 0.169 + \text{DOB} * 0.0107 + \text{Height} * 0.0000675 \quad (6)$$

White oak correlations were the second strongest of all species examined (Table 3). In addition, all correlations were significant ($\alpha = 0.05$). Table 6 lists the average observed bark thickness by diameter class, and bark thickness calculated using Thomas and Bennett's Equations 5 and 6 as shown above, as well as equations from Hilt et al. (1983) and Fowler et al. (1997). Diameter class averages were determined using the overall data set in 2-inch diameter intervals. The bark thicknesses calculated using Hilt et al.'s equation were thickest of all the methods. As with red oak, the thicknesses calculated by Fowler's equation were less than those observed and less than the ones calculated using the Thomas and Bennett equation for DOBs of 14 inches and under. For diameters over 16 inches, bark thicknesses calculated using the Thomas and Bennett models were less than the other methods but close to the average bark thickness measured.

Table 6.—Average observed white oak bark thickness and bark thickness calculated using different methods

Diameter	Average by diameter class	Calculated using Thomas and Bennett equation 5	Calculated using Thomas and Bennett equation 6 ^a	Calculated using Hilt et al. equation ^b	Calculated using Fowler et al. bark factor equation
<i>Inches</i>					
20 >	0.38	0.41	0.41	0.68	0.51
18 - 20	0.38	0.39	0.39	0.67	0.45
16 - 18	0.38	0.37	0.36	0.64	0.40
14 - 16	0.37	0.36	0.34	0.61	0.35
12 - 14	0.34	0.34	0.32	0.56	0.30
10 - 12	0.30	0.32	0.30	0.50	0.25
8 - 10	0.30	0.30	0.28	0.43	0.20

^aAssuming height of 16 feet.

^bAssuming the diameter at breast height measured outside the bark is 20 inches.

Sugar maple

The sugar maple bark thickness prediction model was developed using 915 bark thickness measurements collected from 65 trees harvested from two sites (Table 1). Mean bark thickness was 0.28 inches at both sites (Table 2). Using Wilcoxon rank sum tests, it was determined that there was no significant ($\alpha = 0.05$) difference in mean bark thickness between the two sites. However, mean DOB and ring count were significantly different ($\alpha = 0.05$) between the two sites, although the differences were small. Using the sugar maple model development set, the correlations between DOB, height, and bark thickness measurements were determined and the following prediction equations were developed:

$$BarkThickness = 0.190 + DOB * 0.007 \quad (7)$$

$$BarkThickness = 0.129 + DOB * 0.0131 + Height * 0.0000828 \quad (8)$$

Sugar maple had the second weakest correlations of all species examined (Table 3), but all correlations were significant ($\alpha = 0.05$). Table 7 lists by diameter class the average observed bark thickness and the calculated bark thickness using Equations 7 and 8 by Thomas and Bennett and equations from Haygreen and Bowyer (1996) and Hilt et al. (1983). Diameter class averages were determined using the overall data set with 2-inch diameter intervals. Bark thicknesses calculated using the Hilt et al. (1983) equation were the thickest of all the methods, and those calculated using the Haygreen and Bowyer (1996) equation were the thinnest. Bark thicknesses calculated using Thomas and Bennett models (Eqs. 7 and 8) were between the values determined by the other two methods.

SUMMARY AND CONCLUSION

Most bark thickness studies conducted prior to this research have been based on a larger set of sample trees, but on fewer bark thickness measurements (typically 3 to 7) per stem. In contrast, this study was based on a fewer number of trees but more bark thickness measurements per stem (overall average of over 8 per stem), indicating the key difference between the studies. In addition, the trees used in this study were taken from active logging operations from forests that were slated for harvest.

Table 7.—Average observed sugar maple bark thickness and bark thickness calculated using different methods

Diameter	Average by diameter class	Calculated using Thomas and Bennett (Eq. 7)	Calculated using Thomas and Bennett (Eq. 8) ^a	Calculated using Haygreen and Bowyer equation	Calculated using Hilt et al. equation ^b
<i>Inches</i>					
20 >	--	0.34	0.42	0.328	0.72
18 - 20	--	0.32	0.39	0.294	0.71
16 - 18	0.30	0.31	0.37	0.260	0.69
14 - 16	0.31	0.30	0.34	0.226	0.65
12 - 14	0.29	0.28	0.32	0.192	0.60
10 - 12	0.27	0.27	0.29	0.158	0.53
8 - 10	0.26	0.25	0.26	0.124	0.46

^aAssuming height of 16 feet.

^bAssuming the diameter at breast height measured outside the bark is 20 inches.

Yellow-poplar had the strongest correlation between bark thickness and DOB with an R^2 of 0.223 and a MAE of 0.078 inch. The yellow-poplar bark thicknesses as calculated using the Thomas and Bennett equations (Eqs. 1 and 2) were similar to those calculated using equations derived by Haygreen and Bowyer's (1996) and Hilt et al. (1983). The weakest overall coefficients of determination between DOB and bark thickness were with sugar maple ($R^2 = 0.045$) and red oak ($R^2 = 0.028$). For these two species, DOB explained only a small percentage of the variance in bark thickness in our samples. However, the correlations were highly significant ($\alpha = 0.05$).

Overall, the largest differences in sample characteristics among the sites occurred with white oak. Samples from the Fayette Site 1 had a mean ring count of approximately 126, which was 62 and 45 more than those from the other sites (Fayette Site 2 and Fernow Experimental Forest, respectively), resulting in significant differences ($\alpha = 0.05$) among sites. Mean bark thickness and mean DOB also were significantly different among the three sites. Despite the differences among sites, white oak had the second highest coefficient of determination ($R^2 = 0.137$) of all the four species examined. Furthermore, the relationship between diameter and bark thickness was stronger in our samples than those encountered by Fowler et al. (1997). Sugar maple had the least differences in sample characteristics among the sample sites. Overall, the second weakest correlations were observed with sugar maple.

One of the main goals of this research was to develop a method of estimating bark thickness from stem measurements. Using an expensive laser scanner and the equations developed here would allow operators to estimate bark thickness and, therefore, bark volume and total solid log volume. Given the low correlations between DOB and bark thickness, more work is needed. However, the low mean absolute error is encouraging. In the future we plan to collect additional samples for each species from a larger geographic area in an effort to improve the models. This would also provide a means of validating the current bark thickness equations. As additional species data is collected for the hardwood log defect databank (Thomas 2009b), bark thickness prediction models will be developed.

ACKNOWLEDGMENTS

The authors would like to thank the West Virginia University Experimental Forest staff, West Virginia Division of Forestry, and MeadWestvaco, Inc. in Rupert, WV for their assistance in sample site identification and sample collection.

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CONVERTING INTERNATIONAL ¼ INCH TREE VOLUME TO DOYLE

Aaron Holley, John R. Brooks, and Stuart A. Moss¹

Abstract.—An equation for converting Mesavage and Girard's International ¼ inch tree volumes to the Doyle log rule is presented as a function of tree diameter. Volume error for trees having less than four logs exhibited volume prediction errors within a range of ± 10 board feet. In addition, volume prediction error as a percent of actual Doyle tree volume was generally less than 10 percent for all trees and less than 5 percent for trees larger than 20-inch diameter at breast height (d.b.h.) This equation was developed for those wanting a more accurate method of converting standard inventory estimates of average volume to an alternative log rule.

INTRODUCTION

Since the advent of log rules in the 1800s, foresters have needed to convert volume calculated using one log rule to a different log rule. In the Appalachian hardwood region, one common conversion is from International ¼ inch (Int ¼) to Doyle or vice versa. A common situation occurs when tree volume in a standard inventory is desired in a different log rule. In the Appalachian region, government entities predominantly report tree volumes in Int ¼ rule while industry more commonly uses Doyle. The best solution is to apply the desired log rule, using either a volume table or an equation, to the original data and avoid the use of any conversions, but this is not always possible. There have been many conversion factors developed over time in search of the best approach, but most of these represent single-value conversion factors which produce solutions based on the assumed average tree size in question (Finley and Rickenbach 1996). When the average inventory tree size is different from that used to develop the conversion factor, large errors can be introduced. This problem also occurs when using published timber market reports. The West Virginia University Appalachian Hardwood Center (2012) utilizes an Int ¼ to Doyle conversion ratio of 1.25 which is based on an average tree size of 21- to 24-inch diameter at breast height (d.b.h.) and merchantable heights between 1 and 5.5 logs. The "Pennsylvania Woodlands Timber Market Report" uses an Int ¼ to Doyle conversion ratio of 1.695 which assumes an average log has a 13-inch diameter at the small end (Pennsylvania State Cooperative Extension 2012). In this case, the conversion would be appropriate for individual logs of this size, but large errors can occur when applying log volume conversion ratios to reports that are based on the average of multiple tree volumes, even if the average tree diameter is 13 inches.

The objective of this study was to provide a means of converting tree volume from Int ¼ to Doyle or from Doyle to Int ¼ as a function of tree size. A conversion equation was developed to permit either individual tree volume conversions or average stand volume conversions based on average stand diameter. The conversion ratio was chosen to be a function of tree diameter since the coefficient of variation within a diameter class was smaller than those within a merchantable log class.

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DATA AND METHODS

From Maine to Texas, most hardwood volumes are sold based on the Doyle log rule, and the Mesavage and Girard form class volume tables (Mesavage and Girard 1946) are the market standard for estimating tree volume. The Int ¼ and Doyle form class 78 volume tables were used to calculate the conversion factor for each published diameter and merchantable height class listed in the original tables.

Models were fit to volume ratio data to strike a balance between accuracy and simplicity of use. Thirty nonlinear models, including several polynomial and exponential functions, were tested to determine which models would best fit this ratio data. The only criteria used for model selection were minimum standard error of the estimate, high coefficient of determination, and the graphic evaluation of the residuals. A rational function was selected for the proposed model which is of the form:

$$Ratio = \frac{a + bx}{1 + cx + dx^2} \quad (1)$$

Where:

Ratio = ratio of International ¼ tree volume to Doyle tree volume,

x = tree d.b.h. (inches), and

a, b, c, and d = coefficients to be estimated from the data.

The resultant equation was used to predict the Int ¼ to Doyle conversion ratio and then to estimate the Doyle tree volume for every entry in the Mesavage and Girard's (1946) form class 78 volume table. Volume prediction error was plotted as a function of d.b.h. to visually evaluate volume prediction residuals.

RESULTS

Equation 1 was fit to the ratio of Mesavage and Girard's Int ¼ to Doyle tree volumes for diameter classes ranging from 10 to 40 inches and merchantable height from one to five logs, in half log intervals. Parameter estimates for Equation 1 are:

Parameter	Estimate
a	-2.03809450
b	0.32414999
c	0.24692282
d	0.00007506

The equation provided a good distribution of residuals when plotted over d.b.h. and provided the lowest standard error of the estimate (0.0212) and the highest coefficient of determination (0.9919) of all equations tested. Volume prediction error was greatest for larger diameter trees and for those trees having at least four merchantable 16-foot logs. Volume error for trees having less than four logs exhibited volume prediction errors within a range of ± 10 board feet (bf) (Fig. 1). Percent volume error, as a percent of actual Doyle volume, is depicted in Figure 2. Volume prediction error as a percent of actual Doyle tree volume was generally less than 10 percent for all trees and less than 5 percent for trees larger than 20 inches d.b.h.

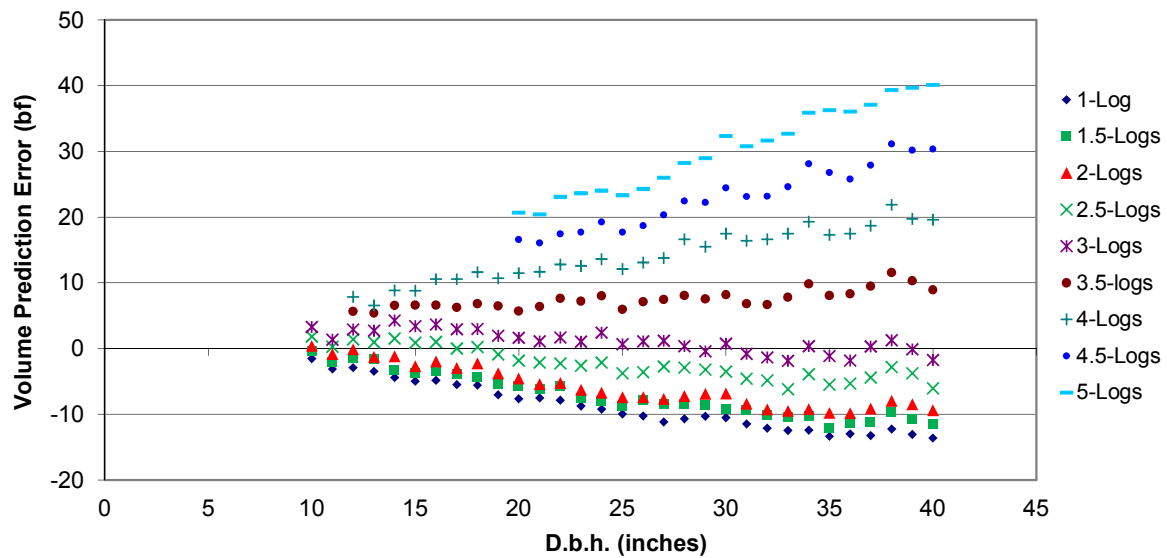


Figure 1.—Doyle board foot prediction error by tree diameter and merchantable log class.

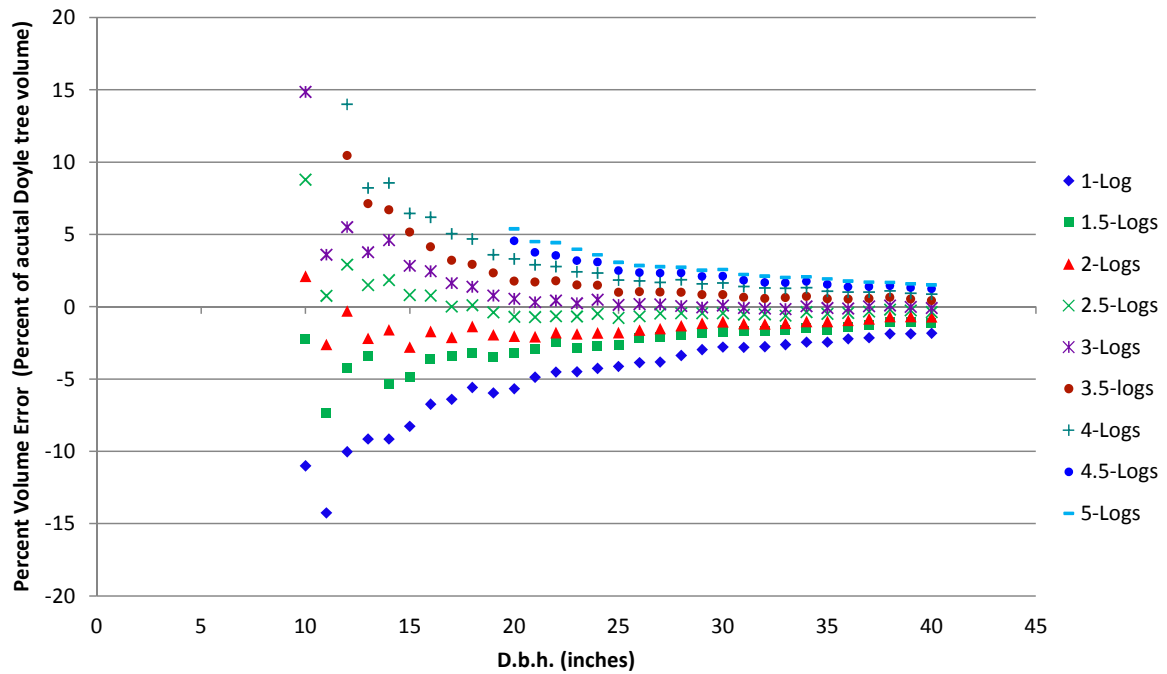


Figure 2.—Percent Doyle board foot prediction error by tree diameter and merchantable log class.

DISCUSSION

An equation was developed to predict the conversion ratio from the Mesavage and Girard form class 78 Int $\frac{1}{4}$ tree volume to Doyle board feet using a rational function. The purpose was to provide a means for predicting board foot volumes that varies by tree d.b.h. The results indicate that this conversion process is reasonably accurate for trees of all diameter classes having merchantable saw log heights of three and a half logs or less. However, when examined by percent volume error, prediction error decreased with increasing tree diameter throughout the d.b.h. range of the original volume table. The equation form selected was based purely on fit statistics rather than biological significance of the underlying data.

EXAMPLE

Suppose Doyle volume is desired for a 22-inch d.b.h. tree with three merchantable logs whose volume was calculated using the International $\frac{1}{4}$ rule. Based on Mesavage and Girard's form class 78 volume table, the International $\frac{1}{4}$ inch volume is 500 bf. Using Equation 1, the conversion ratio for this tree size is 0.78737, resulting in an estimated Doyle volume of 393.68 bf. The volume for this tree calculated directly from the Doyle log rule is 392 bf, an error of less than 2 board feet. The process can also be used to convert Doyle volumes to International $\frac{1}{4}$ inch by dividing the Doyle volume estimate by the conversion ratio listed in Equation 1. In our example, divide the Doyle volume (392 bf) by the calculated conversion ratio to obtain the predicted International $\frac{1}{4}$ inch volume of 497.86 bf. This is an error of less than 3 board feet compared to calculating Intl $\frac{1}{4}$ volume directly. If we assume an inventory results in a per acre volume estimate of 18,000 bf per acre with an average d.b.h. of 22 inches, the conversion ratio from International $\frac{1}{4}$ inch to Doyle is 0.78737, resulting in an estimate of 14,107 bf per acre, Doyle.

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ACCURATELY DETERMINING LOG AND BARK VOLUMES OF SAW LOGS USING HIGH-RESOLUTION LASER SCAN DATA

R. Edward Thomas and Neal D. Bennett¹

Abstract.—Accurately determining the volume of logs and bark is crucial to estimating the total expected value recovery from a log. Knowing the correct size and volume of a log helps to determine which processing method, if any, should be used on a given log. However, applying volume estimation methods consistently can be difficult. Errors in log measurement and oddly shaped logs can make log and residue volume estimates inaccurate. Laser log scanning systems consistently measure the surface points on a log to accuracies of 0.01 inch and finer. Using the measurement data from a laser scanning system and traditional volume equations, or using volume measurements directly from the laser scanning systems, provides highly accurate volume calculations. Such volume measurements are comparable to those obtained using water immersion and displacement methods.

INTRODUCTION

There is much more to a saw log than the lumber that is sawn from it. The remainder, often called residue, is composed of bark removed by a debarker, wood chips produced from grinding the slabs, and sawdust from sawing the lumber. Residue can be measured by cubic volume or weight. There may be as much as 500 pounds of residue for every 1000 board feet of lumber produced (Harkin and Rowe 1971). If the residue is not being used or sold, there is a disposal cost. It is important to grasp the economic value for each saw log that is purchased and to garner maximum utilization of all the components.

Bark has value, be it from selling as landscaping mulch, burning it to heat dry kilns, making charcoal briquettes or carbon filters, or converting it to biomass energy. When logs are purchased, they are priced individually according to size and grade. Various log scales are employed to determine the amount of lumber that can be acquired from each log. The method used when a log is scaled in a log yard excludes the bark content from the purchase price, resulting in it being a free commodity. Every log processed incurs a cost in separating the bark from the wood. Hopefully, the value of the bark exceeds this cost plus the cost of transportation to its end user. Proximity to a market to minimize transportation costs plays an important role in deciding a profitable end use for the bark. Most sawmills have an idea how much bark is produced by how many trucks they fill over a period of time, thus it is volume based or weight based if the trucks are weighed as part of the sales agreement.

Mensurational practices to determine individual log bark volume use equations that assume a log's form is approximate to a geometric form such as a cone, neiloid, or paraboloid. The most commonly used geometric formulas are from Huber and Smalian (Haygreen and Bowyer 1996). These formulas assume saw logs to be approximately the same shape as the frustum of a paraboloid (Fig. 1). The difference between Huber's and Smalian's formulas is Huber's formula assumes the average cross sectional area is located at the middle of the log while Smalian's formula takes the average cross sectional area of the large and small ends of the log.

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Frustum of a paraboloid

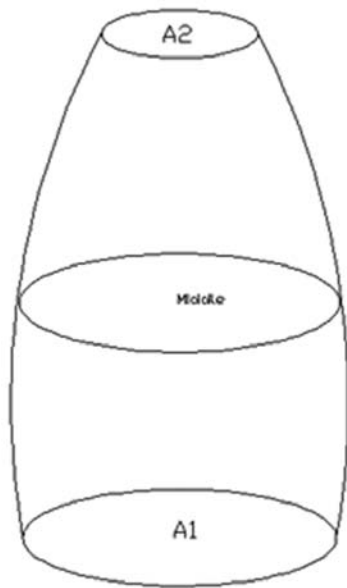


Figure 1.—Frustum of a paraboloid, the common log shape assumed by the Huber and Smalian volume formulas.

The first step in using these equations to calculate bark volume is to calculate whole log volume outside of the bark. Then bark thickness, either measured or taken from published tables, is subtracted to determine log volume of wood under the bark. Bark volume is the difference between the two results. One other method (Dobie and Wright 1975) to determine bark volume is to assume the ratio of bark volume to whole log volume is equal to the ratio of twice bark thickness to log diameter measured outside of the bark.

To date, perhaps the most accurate examination of log volumes was performed by Martin (1984). Martin compared the log volume estimates using Smalian's, Huber's, and other formulas to actual log volume determined using a xylometer or log immersion tank. The volume of displaced water was then compared to the calculated volume of the different volume equations. Martin found that Huber's formula was the most accurate volume estimator with a mean absolute difference of 0.36 cubic feet.

Currently, the U.S. Forest Service research lab in Princeton, WV is conducting a line of research using a high-resolution laser log scanner to create detailed three-dimensional external log images. From this image data the total volume of a log can be very accurately determined. If bark thickness is known, the value can be subtracted from the raw data points to determine wood volume. Bark volume is the difference between total log volume and total wood volume. Determining bark volume of each log becomes a matter of programming the computer to allow a sawmill operator to accurately track the bark produced from each log as it is processed at the sawmill. Laser systems are currently being used in larger sawmills in the East for the purpose of judging the best opening face of a log or predicting lumber volumes that can be produced. Adding bark and volume calculators would be a benefit to the sawmill operator without any added cost to the laser system, thus giving the sawmill operator the ability to determine the economic value of each log and to garner maximum utilization of all the components

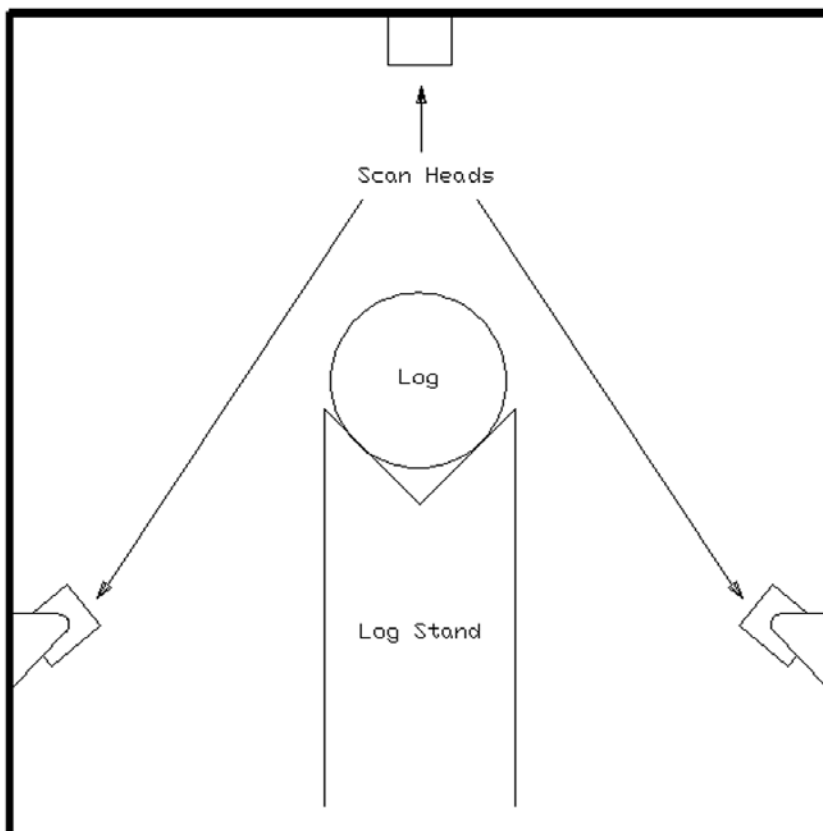


Figure 2.—Schematic diagram of high-resolution log scanner.

METHODS

A high-resolution laser log scanner was constructed for the purpose of scanning hardwood logs to allow automated detection of severe surface defects (Thomas and Thomas 2011). This scanner is composed of three industrial laser scan heads designed for the wood processing industry. The scanners are stationed at 120 degree intervals on a circle with a diameter of approximately 8 feet. This allows the three scanners to collect a complete surface scan of the log. The log is supported in V-stands every 5 feet at the center of the circle of scanners (Fig. 2). The scanner then passes over the log and collects a scan line around the circumference of the log every 1/16 inch. Resolution between points within each scan line varies depending on the size of the log, but is typically around 1/8 inch. All points are measured to the nearest 0.001 inch. A dot-cloud image sample of a scanned log (log 15A) is shown in Figure 3. The two vertical white marks are missing data due to shadowing of the log surface by the V-stands.

Two basic problems with the scanned log data include missing data and outlier data caused by dust, hanging bark, and portions of the V-stands. Outlying data points are removed using a multi-step process. First, the log data is geometrically centered about the z-axis, and the distances of all points to the z-axis are calculated. The mean and standard deviation of point distances to the z-axis are determined, and all points outside the interval mean ± 2 standard deviations are marked as outliers. Next, all missing points and points identified as outliers are filled in using an average of a minimum of 100 neighboring points. These procedures remove most outlying data points and create an accurate log depiction for determining log characteristics.

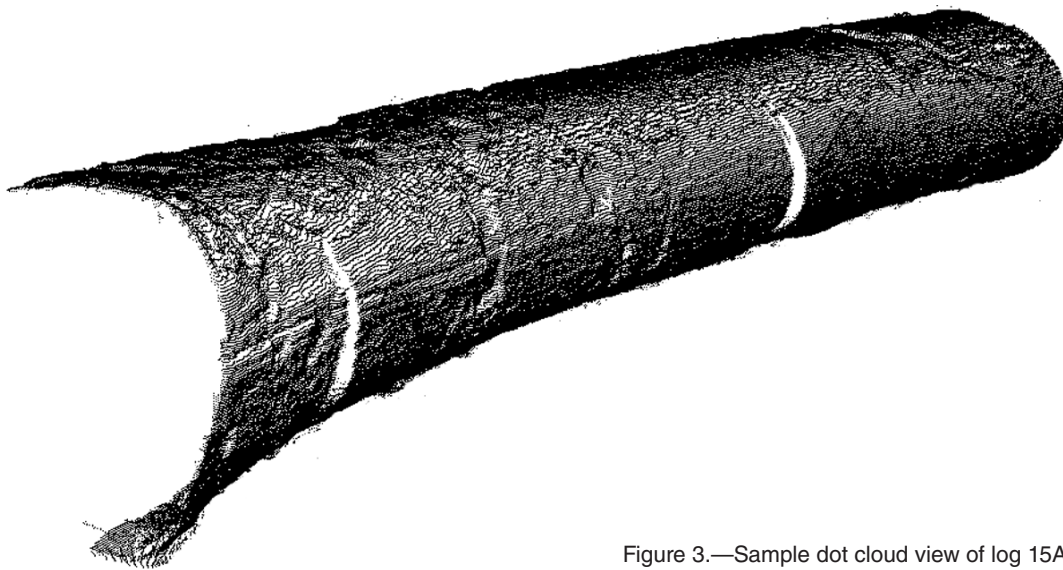


Figure 3.—Sample dot cloud view of log 15A.

Recall that each scan line represents a 1/16-inch thick slice of the log. Volume for a log is determined for each scan-line by calculating the volume of a series of triangles (Fig. 4). Using the center point as one point of a triangle and two adjacent edge points, the area for a portion of the scan is determined. Given the three points A, B, and C, the area of each triangle is calculated using Heron's equation (Page 2009):

$$area = 0.0625 * \sqrt{S * (S - AB) * (S - BC) * (S - CA)}$$

Where:

$$S = \frac{AB + BC + CA}{2}$$

and AB, BC, and CA are the lengths of the triangle's sides. Calculating the area for all triangles yields the total area for a single scan line or slice of the log. Adding the volume of all slices together yields the total volume of the log, bark included.

Recently, a series of yellow-poplar (*Liriodendron tulipifera* L.), red oak (*Quercus rubra* L.), white oak (*Quercus alba* L.), and sugar maple (*Acer saccharum* Marsh.) defects were collected and studied to determine the relationships among external log defect indicators and internal defect manifestations (Thomas 2008, Thomas 2009, Thomas 2012, Thomas²). In these studies, bark thicknesses and diameter outside bark (DOB) were recorded for each sample. Using this data, a series of linear regression analyses were performed to determine the correlation of DOB to bark thickness for each species (Thomas³). The bark thicknesses of red and white oak were estimated using the following equations:

Red oak: Bark Thickness = $0.267 + DOB * 0.005$

White Oak: Bark Thickness = $0.295 + DOB * 0.009$

² Thomas, R.E. [N.d.]. Predicting internal hard maple (*Acer saccharum*) log defect features using surface defect indicator measurements. Manuscript in preparation. On file with authors.

³ Thomas, R.E.; Bennett, N. [N.d.]. Estimating bark thicknesses of common Appalachian hardwoods. Manuscript in preparation. On file with authors.

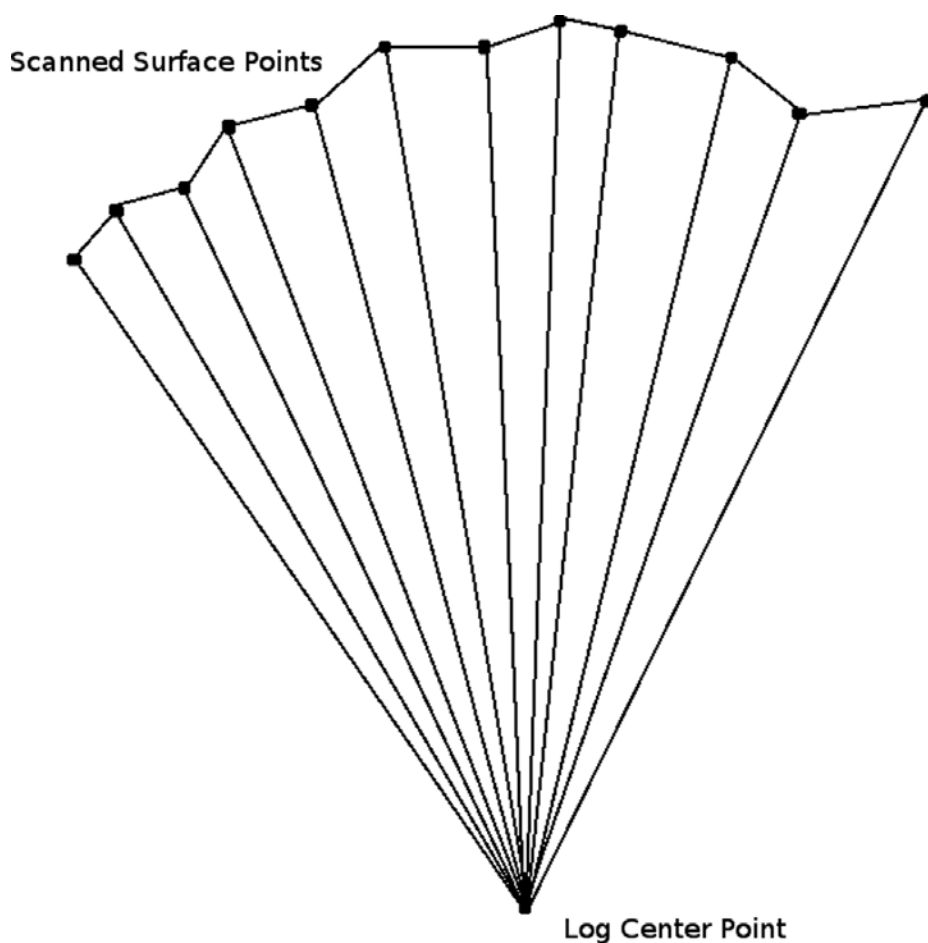


Figure 4.—Calculating the area of a series of triangles to determine log volume.

Using these bark thickness equations, the total bark volume as well as debarked log volume can be determined. To account for DOB changing along the length of the log, bark thicknesses are re-evaluated every foot along the log. To find debarked volume, the surface points are moved toward the center point a distance equivalent to the bark thickness. Debarked log volume is calculated using the modified surface point positions. Bark volume is the difference between total log volume and debarked volume.

Sixty-six white oak trees were randomly selected from three sites in West Virginia from which 249 logs were bucked. In addition, 32 red oak trees were randomly selected from an additional site in West Virginia from which 140 logs were bucked. All logs were scanned with the high-resolution laser scanner. From the total sample population of 369 logs, 20 white oak and 12 red oak logs were randomly selected for this volume study. During the analysis, one white oak log was identified as an outlier using Cook's distance ($\alpha = 0.05$) (Cook and Weisberg 1982). This log was bucked near a fork and had significant taper from either end to the center and was shaped much like a dog bone. This log was removed from the sample, leaving a total sample size of 31 logs.

Table 1 lists the measurements and characteristics of the sampled logs. All measurements were taken using the laser scanner measurement system. Using the laser scan data, the diameter can be determined at any point along the length of a log. To find the diameter, the geometric center of the

Table 1.—Measurements and characteristics of sample logs

Log	Species	Length	Large end diameter	Small end diameter	Sweep	Eccentricity	Average taper per foot
			<i>Inches</i>				<i>Inches</i>
F-12C	White Oak	125.5	18.6	17.5	1.91	1.91	0.11
F-36A	White Oak	174.7	26.8	23.8	2.12	2.12	0.22
F-15B	White Oak	193.8	11.6	10.5	2.71	2.71	0.07
GC-39C	White Oak	100.3	14.2	12.5	0.85	0.85	0.21
GC-20C	White Oak	132.9	14.0	12.3	1.72	1.72	0.17
FA-12B	White Oak	150.6	15.1	13.7	3.83	3.83	0.12
FA-12A	White Oak	143.1	22.9	15.6	1.38	1.38	0.73
FA-6B	White Oak	194.9	19.7	17.3	3.02	3.02	0.15
F-11B	White Oak	198.4	13.2	12.1	0.93	0.93	0.07
GC-31A	White Oak	207.9	16.3	12.9	1.32	1.32	0.21
GC-28A	White Oak	149.3	17.9	13.6	1.84	1.84	0.36
GC-23C	White Oak	107.7	11.4	10.5	1.26	1.26	0.11
GC-38C	White Oak	153.9	11.3	10.2	1.83	1.83	0.09
F-13A	White Oak	198.3	21.6	15.5	1.16	1.16	0.38
F-31C	White Oak	125.8	18.9	17.1	2.14	2.14	0.17
FA-11C	White Oak	132.0	19.1	18.3	1.83	1.83	0.08
FA-7A	White Oak	195.1	24.0	18.6	2.11	2.11	0.34
FA-9A	White Oak	144.3	24.6	19.1	2.68	2.68	0.45
FA-11B	White Oak	119.2	19.5	18.3	1.13	1.13	0.15
27C	Red Oak	187.1	19.5	19.0	0.59	0.59	0.03
32A	Red Oak	126.3	21.5	18.7	1.01	1.01	0.28
8D	Red Oak	125.7	17.8	16.6	1.40	1.40	0.12
28B	Red Oak	125.9	14.2	13.2	0.92	0.92	0.09
9C	Red Oak	156.8	10.2	8.4	3.14	3.14	0.15
29D	Red Oak	126.3	15.8	8.8	2.38	2.38	0.70
11C	Red Oak	158.2	14.4	13.2	2.92	2.92	0.10
15A	Red Oak	152.1	18.8	14.4	2.21	2.21	0.37
15B	Red Oak	104.7	14.1	13.2	0.96	0.96	0.12
17A	Red Oak	126.1	21.2	17.8	1.15	1.15	0.33
17B	Red Oak	126.81	17.682	17.3	0.50	0.50	0.03
17C	Red Oak	126.38	17.192	16.9	0.73	0.73	0.03

scanned log circumference for a single slice of the data is determined. Next, the distances from the center point to all circumference points are calculated and the average distance or radius determined. The diameter is then twice the average radius. Sweep is measured as the maximum distance of a straight line running between both ends of the log. Eccentricity is an index indicating how elliptical or out of round the log is. The eccentricity index is calculated using the following formula (Mason and Hazard 1947):

$$eccentricity = \frac{\sqrt{MajorDiameter^2 - MinorDiameter^2}}{MajorDiameter}$$

For example, a perfectly round log will have an index of 0.00, a log with a major diameter of 16 and a minor diameter of 14 would have an eccentricity index of 0.48. The greater the index value, the greater the degree of eccentricity or ellipticality. Taper per foot is the difference between large and small end diameter outside bark measurements divided by the scaled log length, and the greater the number, the greater the degree of taper present in the log.

Traditionally, log volumes have been estimated using one of several possible methodologies (Haygreen and Bowyer 1996), including Huber's (Eq. 1) and Smalian's (Eq. 2) geometric methods shown below:

$$\text{Volume} = L \times A \quad (1)$$

$$\text{Volume} = \left(\frac{A_1 + A_2}{2} \right) \times L \quad (2)$$

Where L is log length, A is the cross sectional area at the middle of the log, and A_1 and A_2 are the cross sectional areas of the large and small ends of the log, respectively.

Huber's and Smalian's formulas (Eqs. 1 and 2) find the volume of the log. To find the bark volume, the cross-sectional areas must be calculated twice, once with the bark and once excluding the bark. Subtracting the excluded bark volume from total volume yields bark volume. Another way to determine bark volume is to use Dobie and Wright's formula (Dobie and Wright 1975) listed below (Eq. 3) which assumes the ratio between double bark thickness and log diameter outside bark is the same ratio as bark volume to total log volume.

$$\text{Bark Volume} = \frac{DOB^2 - DIB^2}{DOB^2} \times 100 \quad (3)$$

DOB is the diameter measured outside the bark and DIB is diameter measured inside the bark. We used DOB and DIB measurements that were averages of the midpoint and small and large end diameters. To get bark volume from the Dobie and Wright equation (Eq. 3), you still have to determine total log volume by one of the previously described methods.

Using the 2011 R statistical package (R Foundation for Statistical Computing, Vienna, Austria), Anderson-Darling normality tests ($\alpha=0.05$) (Anderson and Darling 1952) were performed to determine if the variables and volumes involved in this study were normally distributed. The tests revealed that sweep, eccentricity, scanner determined volumes, Smalian calculated volumes, and the Huber calculated volumes were normally distributed. However, log taper and the differences between the scanner volumes and both the Huber and Smalian calculated volumes were not normally distributed.

Paired-t tests were used to compare Huber and Smalian calculated volumes to the scanner determined volumes. As these comparisons involved two tests, the significance level was adjusted for each individual test using the Bonferonni correction (Abdi 2007). An overall significance level of 0.05 was used with an adjusted significance of 0.025 for each individual test.

To determine if sweep, taper, or eccentricity had a significant relationship to the differences between scanner calculated volume and the Huber and Smalian volume estimation methods, a series of simple linear regression analyses were performed. Sweep, taper, and eccentricity were the independent variables and the dependent variable was the volume difference. The correlation for each independent

variable was analyzed separately. Cook's distance (Cook and Weisenberg 1982) was used to determine if any individual observation had a high influence over the regression. The residuals from each analyses were tested using the Anderson-Darling normality tests ($\alpha=0.05$) (Anderson and Darling 1952) to determine if the residuals were normally distributed. In all cases where a log feature had a significant correlation to a volume difference, the residuals had a normal distribution.

Due to the non-normal distributive nature of the bark volumes, Wilcoxon signed rank tests ($\alpha = 0.05$) were used to compare the results of the different bark volume estimation methods (Wilcoxon 1945). Five tests were performed comparing the scanner determined bark volume to the volumes calculated using the Smalian, Huber, and the Dobie and Wright equation variants. The significance level for the tests was adjusted using the Bonferroni correction (Abdi 2007).

RESULTS AND DISCUSSION

Table 2 lists the log volumes as determined by the scanner data and estimates using Smalian's and Huber's formulas. For whole log volumes, comparing the scanner volumes to Smalian estimated volumes shows a mean difference of -1.02 cubic feet with a maximum observed difference of -7.39 cubic feet. Volume estimates using Huber's formula were closer to the observed scanner volumes with a mean difference of 0.09 cubic feet and a maximum difference of 3.61 cubic feet. The Smalian maximum difference occurred on log F-36A which has a large degree of taper caused by butt swell. The maximum difference with Huber's equation occurred on log FA-7A which also has a large degree of taper. Using the R statistical package, two paired t-tests ($\alpha = 0.05$) were performed with the significance level for the tests being adjusted using the Bonferroni correction (Abdi 2007). In these tests, the Smalian and Huber estimated volumes were compared independently to the scanner volumes. The tests revealed that the means of the scanner and Smalian volumes were significantly different while those of the Huber were not significantly different from the scanner data.

Overall, results for debarked log volumes followed similar trends as whole log volumes (Table 2). Comparing scanner volumes to Smalian estimated volumes showed a mean difference of -0.94 cubic feet with a maximum observed difference of -7.12 cubic feet. Huber's debarked log volumes also were closer to the scanner volumes with a mean difference of 0.13 cubic feet and a maximum difference of 3.47 cubic feet. As before, the maximum differences for the Smalian and Huber volume methods occurred on the same logs that had the maximum whole log differences. Two paired t-tests ($\alpha = 0.05$) were performed to compare the Smalian and Huber estimated volumes to the scanner volumes. As before, the significance levels for the tests were adjusted using the Bonferroni correction (Abdi 2007). These tests revealed that the means of the scanner, Smalian, and Huber volumes were significantly different.

To determine what log features, if any, had a significant correlation to the difference between the scanner-based volume methodology and the Huber or Smalian volume estimations (rightmost columns Table 2), three simple linear regression analyses were performed. These analyses tested the relationship of log sweep, eccentricity, and taper to the differences between the scanner-based volumes and the Huber and Smalian volumes. It was found that sweep and eccentricity had no significant correlation to the differences. While Huber's volume error was not significantly correlated to taper, Smalian's volume error was weakly correlated to taper with an R^2 of 0.10. Thus, it appears that in general, Smalian volume estimation equations are slightly less accurate when used with tapered logs.

Table 2.—Volume measurements of sample logs using three different measurement methods

Log	Scanner whole log volume	Scanner debarked log volume	Smalian whole log volume	Smalian debarked log volume	Huber whole log volume	Huber debarked log volume	Scanner vs. Smalian whole log difference	Scanner vs. Huber whole log difference	Scanner vs. Smalian debarked difference	Scanner vs. Huber debarked difference
<i>----- Cubic feet -----</i>										
F-12C	20.3	18.7	18.5	17.0	18.6	17.1	1.72	1.62	1.67	1.58
F-36A	43.6	40.5	50.9	47.6	46.0	42.9	-7.39	-2.49	-7.12	-2.39
F-15B	10.5	9.4	10.8	9.6	10.7	9.5	-0.24	-0.21	-0.17	-0.14
GC-39C	7.4	6.8	8.2	7.4	6.6	5.8	-0.71	0.89	-0.54	0.98
GC-20C	9.2	8.3	10.4	9.4	9.2	8.2	-1.23	-0.03	-1.14	0.00
FA-12B	15.6	14.2	14.3	12.9	14.3	12.9	1.30	1.30	1.35	1.35
FA-12A	18.8	17.2	24.9	23.1	16.9	15.3	-6.05	1.99	-5.81	1.92
FA-6B	31.9	29.4	30.4	28.0	33.0	30.4	1.43	-1.12	1.38	-1.06
F-11B	14.5	13.0	14.5	13.0	15.0	13.5	-0.04	-0.54	-0.02	-0.49
GC-31A	18.8	17.2	20.4	18.5	17.8	16.1	-1.58	0.98	-1.39	1.06
GC-28A	14.6	13.2	17.1	15.6	13.9	12.5	-2.56	0.69	-2.43	0.68
GC-23C	5.8	5.2	5.9	5.2	5.8	5.2	-0.06	-0.01	-0.04	0.01
GC-38C	7.4	6.6	8.1	7.2	7.0	6.1	-0.72	0.44	-0.61	0.49
F-13A	24.8	22.9	31.9	29.4	25.1	22.8	-7.10	-0.27	-6.52	0.05
F-31C	20.3	18.7	18.6	17.0	21.3	19.7	1.69	-1.09	1.64	-1.03
FA-11C	19.6	18.0	21.0	19.3	18.8	17.2	-1.34	0.78	-1.25	0.78
FA-7A	34.5	31.8	40.9	38.0	30.9	28.3	-6.40	3.61	-6.17	3.47
FA-9A	32.2	29.9	31.8	29.5	34.8	32.3	0.39	-2.58	0.38	-2.47
FA-11B	19.6	18.1	19.4	17.8	21.5	19.9	0.20	-1.90	0.20	-1.81
27C	32.4	30.1	31.6	29.2	32.6	30.2	0.83	-0.18	0.85	-0.13
32A	22.8	21.1	23.2	21.6	22.7	21.1	-0.47	0.03	-0.45	0.04
8D	15.3	14.1	16.8	15.5	15.6	14.3	-1.51	-0.29	-1.42	-0.25
28B	10.7	9.7	10.8	9.7	10.7	9.7	-0.05	0.02	-0.03	0.04
9C	6.2	5.4	6.2	5.4	6.6	5.8	-0.05	-0.44	0.03	-0.33
29D	12.3	11.2	9.4	8.4	12.7	11.6	2.92	-0.41	2.79	-0.33
11C	13.1	11.9	13.7	12.4	12.6	11.3	-0.58	0.57	-0.48	0.61
15A	16.6	15.1	19.3	17.8	15.3	13.9	-2.75	1.23	-2.61	1.21
15B	9.2	8.3	8.9	8.0	9.2	8.3	0.30	-0.03	0.31	-0.01
17A	20.2	18.7	22.0	20.4	19.7	18.2	-1.74	0.54	-1.67	0.53
17B	17.1	15.7	17.7	16.3	17.1	15.7	-0.59	-0.04	-0.55	-0.02
17C	17.5	16.1	16.7	15.4	17.9	16.5	0.73	-0.40	0.74	-0.35
Mean							-1.02	0.09	-0.94	0.13
Maximum difference							-0.04	-0.01	-0.02	0.01
Minimum difference							-7.39	3.61	-7.12	3.47

In addition to the bark volume as determined by the scanner, we examined five different approaches to estimating bark volume. Two methods were simply the difference between whole log and debarked volume estimates calculated using the Smalian and Huber formulas. The other three methods used the Dobie and Wright bark estimation formula where bark volume is estimated directly using the scanner data and by using the Smalian and Huber whole log volume estimates.

Table 3 contains the bark volume estimates for these methods. The rightmost column contains the standard deviation of the different bark volume calculation methods for each log. Higher standard deviation values indicate logs that have the greatest volume discrepancies among the different volume estimation methods. Six of seven logs with the six highest standard deviation values (>0.10) were all butt logs. In addition, all seven logs had high degrees of taper (Table 1), indicating that one or more methods were more error prone when estimating the volume characteristics of butt logs, specifically those with taper or butt swell. This observation was consistent given the earlier finding that taper is significantly correlated with whole log volume estimation error. The results indicated that there were no significant differences between the means of the scanner methodology and the Dobie and Wright Huber-based and the Huber methods. The means of all other methods were significantly different from the scanner methodology.

SUMMARY AND CONCLUSION

Whole log and debarked volumes as determined by the scanner were most comparable to the volumes calculated using Huber's formula, with mean differences of 0.09 and 0.13 cubic feet, respectively. These results are similar to those found by Martin (1984), where Huber's formula more accurately estimated volume than Smalian's formula. Recall that Smalian's formula is based on an average of large and small end cross-sectional area, while Huber's formula is based on the midpoint cross-sectional area. Thus, the volume of logs with a large degree of taper, specifically butt logs, will be overestimated using Smalian's formula.

While the mean differences between the Huber and scanner volume methods were acceptably small, one must remember that the diameter and length measurements were from the laser scanning system. The laser scanner measures surface point locations accurate to 0.01 inch. As such, these measurements provide very accurate input data for Huber's formula. Field measurements for estimating volume will not likely be this accurate. All volume equations are dependent on accurate measurements of log diameter and length. Any error in measuring either is magnified when volume is calculated.

Calculating bark volume using the scanner-based methodology yields a solid volume. This removes the normal concerns of particle size and compaction when measuring bark volume. Using weight to measure residue also has problems. The moisture content of the bark varies among logs and from day to day. Similarly, there are specific gravity differences between species. Using the data from the laser scanner provides the most accurate way of determining bark and log volume. In mills where the logs are scanned at the headrig after debarking, it would be trivial, in terms of both cost and effort, to estimate bark thickness based on the log's diameter and calculate total bark volume or weight at a specific moisture content. Even in situations where DOB varied significantly along the log, the scanner-based method described here would still provide accurate bark and log volume estimates.

Table 3.—Bark volumes as calculated from scanner and estimation formulas

Log	Calculated bark thickness (Inches)	Scanner bark volume	Smalian bark volume	Huber bark volume	Dobie and Wright bark volume using scanner	Dobie and Wright bark volume using Smalians	Dobie and Wright bark volume using Hubers	Mean	Standard deviation
----- Cubic feet -----									
F-12C	0.383	1.61	1.54	1.55	1.71	1.56	1.55	1.59	0.06
F-36A	0.419	3.45	3.31	3.15	3.05	3.57	3.06	3.26	0.21
F-15B	0.319	1.14	1.20	1.20	1.18	1.21	1.21	1.19	0.03
GC-39C	0.335	0.65	0.80	0.71	0.76	0.83	0.67	0.73	0.07
GC-20C	0.334	0.95	1.03	0.97	0.94	1.06	0.94	0.98	0.05
FA-12B	0.357	1.35	1.37	1.38	1.51	1.38	1.38	1.39	0.06
FA-12A	0.375	1.60	1.84	1.53	1.57	2.07	1.40	1.67	0.24
FA-6B	0.384	2.53	2.46	2.57	2.59	2.48	2.65	2.55	0.07
F-11B	0.337	1.49	1.50	1.53	1.49	1.49	1.55	1.51	0.03
GC-31A	0.347	1.72	1.87	1.75	1.79	1.94	1.70	1.80	0.09
GC-28A	0.354	1.35	1.48	1.34	1.34	1.57	1.27	1.39	0.11
GC-23C	0.322	0.65	0.67	0.67	0.66	0.67	0.67	0.66	0.01
GC-38C	0.314	0.81	0.92	0.85	0.85	0.94	0.81	0.86	0.05
F-13A	0.379	2.07	2.49	2.23	2.08	2.68	2.11	2.28	0.25
F-31C	0.380	1.59	1.53	1.65	1.68	1.54	1.73	1.62	0.08
FA-11C	0.380	1.58	1.67	1.58	1.60	1.71	1.53	1.61	0.07
FA-7A	0.390	2.74	2.89	2.53	2.67	3.16	2.35	2.72	0.28
FA-9A	0.407	2.50	2.29	2.41	2.44	2.41	2.53	2.43	0.08
FA-11B	0.386	1.56	1.55	1.63	1.57	1.55	1.69	1.59	0.06
27C	0.363	2.33	2.34	2.37	2.40	2.34	2.40	2.36	0.03
32A	0.364	1.63	1.65	1.63	1.66	1.69	1.63	1.65	0.02
8D	0.350	1.25	1.34	1.29	1.24	1.36	1.26	1.29	0.05
28B	0.337	1.01	1.03	1.03	1.03	1.03	1.03	1.03	0.01
9C	0.317	0.74	0.81	0.84	0.80	0.80	0.87	0.81	0.04
29D	0.341	1.08	0.93	1.14	1.32	1.01	1.37	1.14	0.17
11C	0.336	1.20	1.30	1.24	1.27	1.32	1.21	1.26	0.05
15A	0.345	1.42	1.55	1.39	1.41	1.65	1.31	1.45	0.12
15B	0.337	0.86	0.85	0.87	0.88	0.85	0.88	0.87	0.01
17A	0.359	1.51	1.58	1.50	1.51	1.64	1.45	1.53	0.07
17B	0.353	1.36	1.40	1.37	1.36	1.41	1.36	1.38	0.02
17C	0.353	1.35	1.35	1.40	1.40	1.34	1.43	1.38	0.04
Mean	0.357	1.52	1.57	1.53	1.54	1.62	1.52		

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

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FOREST INDUSTRY & GROWTH

IMPACTS OF CONSTRUCTION MARKETS ON FIRM PERFORMANCE AND BEHAVIORS IN THE SECONDARY WOODWORKING INDUSTRY, 2010-2013

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Abstract.—Despite the severe downturn in the housing market that began in 2007, construction-related uses collectively remain the largest source of demand for appearance-grade hardwood lumber in the United States. Therefore, industry practitioners and researchers alike maintain an interest in housing and construction trends. This study investigated market conditions from the perspective of secondary woodworkers via 4 years of data from subscribers to "Wood & Wood Products" (now named "Wood Products") magazine. The series began in February 2010 (for the year 2009) and surveys were conducted in February/March 2011, 2012, and 2013. Year-over-year sales volume consistently improved for responding companies, with 58 percent of companies reporting an increase in sales from 2011 to 2012. However, this trend corresponded with a decline in the overall number of companies in operation. The residential remodeling market became more important to secondary manufacturers after 2009 as production associated with new construction declined. Overall economic conditions and development of new products were rated as factors that had helped companies increase sales during the period. The housing market, remodeling expenditures, offshore competition, and entry of domestic competitors all were rated as causes of sales declines during the period. As most companies pursued both cost reduction and revenue-seeking strategies to weather the housing downturn, referrals and following up on leads were rated by respondents as the most frequently used types of marketing communications; referrals were especially important for smaller firms.

INTRODUCTION

Given the importance of construction-based markets to demand for hardwood products, uncertainty regarding housing and related building activity remains an important issue for the industry (Bumgardner et al. 2011). Understanding the performance trends and behaviors of secondary woodworking manufacturers is important to researchers and practitioners alike when developing outreach and operating strategies to address current business conditions. For example, considering the prevalence of small firms in the secondary woodworking industry, it might be expected that actions aimed at finding new revenue sources to generate cash flow have been especially important to survive the downturn (Latham 2009). Buehlmann et al. (2013) recently found that marketing activities were rated as more important to business success by small woodworking firms than by large ones, even though small firms have been shown in other studies to face difficulties in carrying out many traditional marketing activities (Gilmore et al. 1999, Huang and Brown 1999). This paper presents analysis and discussion of 4 years of data collected from subscribers to a major trade publication involved in secondary wood products manufacturing to shed light on the actions firms have taken to remain profitable.

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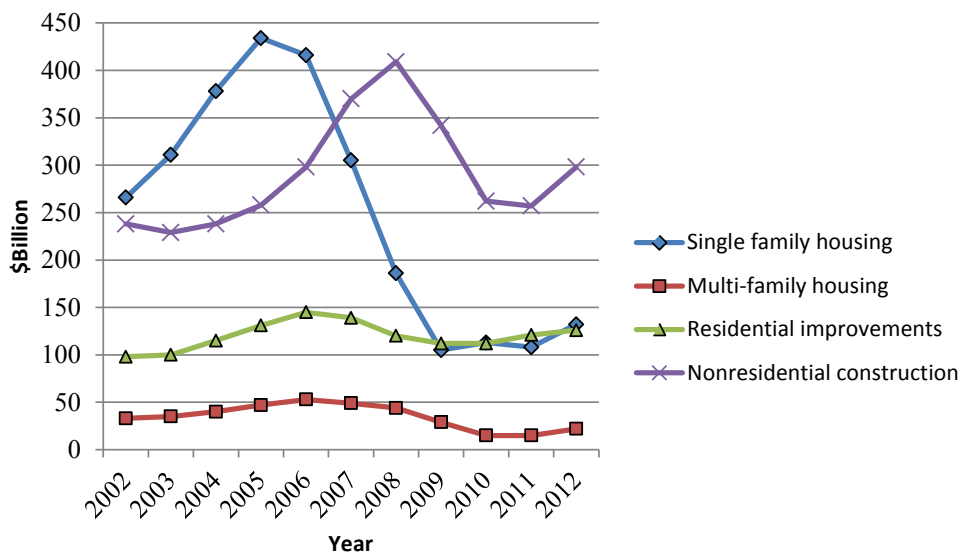


Figure 1.—Value of private U.S. construction put in place, 2002-2012 (U.S. Census Bureau 2013a).

Although construction markets in many areas remain challenging, national data showed improvements in 2012 (U.S. Census Bureau 2013a). Figure 1 displays the value of private construction in the United States, which increased in all major categories from 2011 to 2012, ranging from a 4-percent increase in residential improvements (or remodeling) to a 48-percent increase in multi-family housing. Despite the large proportional increase, multi-family remained by far the smallest market based on value. Single family construction and nonresidential construction also increased by 22 percent and 16 percent, respectively. For the fourth straight year, spending on remodeling was very similar to spending on single family housing construction. This result likely reflects, in part, an aging U.S. housing stock (Bumgardner et al. 2011) and plans by many homeowners to remain in their current homes for a longer time (or choosing not to move at all). Remodeling expenditures remained relatively stable throughout the past 10 years, including during the highest and lowest points of the housing market.

The trend also shows continued softness in the single family market compared to the early 2000s, when single family value surpassed that of the nonresidential construction market. Single family housing starts have seemingly stabilized since the precipitous fall that started after the peak in 2005; starts increased from about 431,000 in 2011 to about 535,000 starts in 2012 (U.S. Census Bureau 2013b), representing a 24-percent increase and the highest level in 4 years. Still, single family starts in 2012 were only about a third of the nearly 1.7 million starts in 2005 (Fig. 2). Correspondingly, total employment in both the wood kitchen cabinets and countertops (NAICS² 33711) and millwork (NAICS 32191) industries each declined by 46 percent from 2006 (a peak employment year for both industries) to 2012 (U.S. Bureau of Labor Statistics 2013).

Against this backdrop, the objective of this paper was to describe and analyze, from the perspective of active secondary woodworkers, what changed over the past 4 years in terms of market conditions, performance trends, and the actions taken to remain profitable.

² NAICS stands for the North American Industry Classification System.

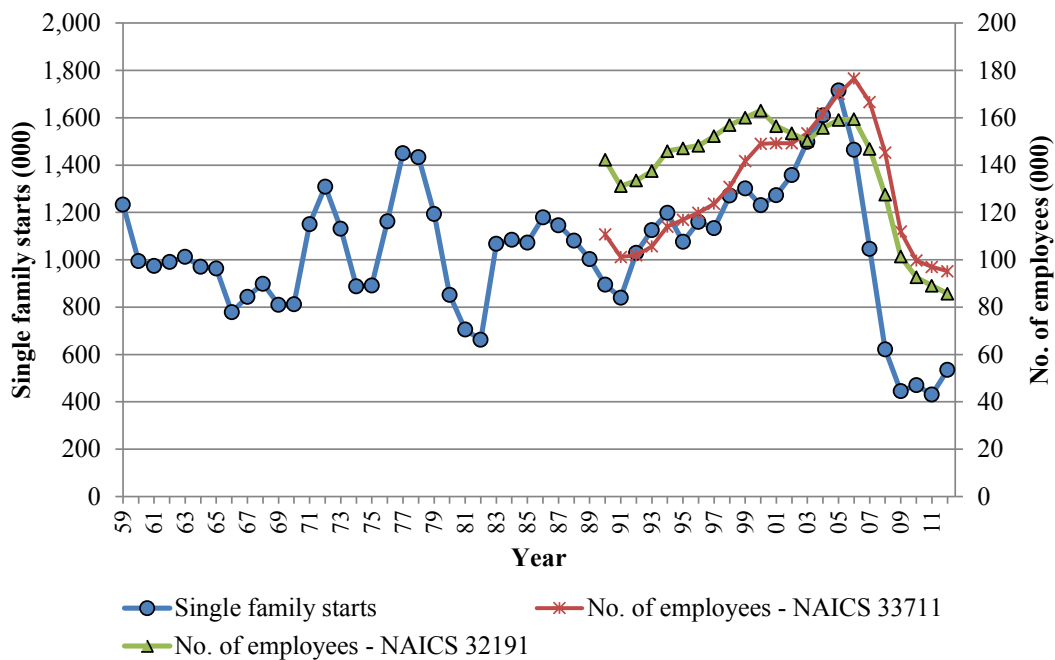


Figure 2.—Single family housing starts from 1959 to 2012 (U.S. Census Bureau 2013b), and number of employees in the wood kitchen cabinets and countertops industry (NAICS 33711) and millwork industry (NAICS 32191) from 1990 to 2012 (U.S. Bureau of Labor Statistics 2013).

METHODS

Beginning in February 2010 (assessing data for 2009), an annual online survey has been conducted by "Wood & Wood Products" (now named "Wood Products") magazine of its secondary woodworking subscribers in February/March to assess performance, behaviors, and perceptions of market conditions in housing and related construction markets. The number of responses from 2010 through 2013 is shown in Table 1, and ranged from 359 in 2010 to 244 in 2013. The number of invitations sent each year ranged from about 9,000 to 13,000, except for 2010, when it was 786 (Bumgardner et al. 2011). Most respondents each year were either company owners or in positions of corporate/operating management (ranging from 67 to 72 percent), and each year respondents represented firms in at least 41 states. More than 80 percent of responses each year were received from firms with only one manufacturing facility.

Statistical analyses were carried out by using SAS Enterprise Guide 4.3 (SAS Institute Inc., Cary, NC). An alpha level of 0.10 was used for all tests. For comparisons across study years for interval-level (i.e., scaled) responses, one-way analysis of variance (ANOVA) was used to determine if there were differences among means. Homogeneity of variance (HOV) was tested by using Levene's test and the Brown-Forsythe test, two common HOV tests available in SAS Enterprise Guide. When the null hypothesis of equal variances was rejected by at least one of the HOV tests, Welch's ANOVA was used. Each year was treated as an independent sample because the number of repeat respondents was somewhat low year-over-year (22 percent for 2011-2010, 19 percent for 2012-2011, and 16 percent for 2013-2012). Additionally, as shown in Table 1, the groups were generally similar in composition each year in terms of the main products produced, firm size, wood materials used, and price-points of primary products.

For frequency count data, chi-square tests were used. For two-group comparisons of interval-level data (i.e., comparing small and large firms on scaled responses), two-tailed *t* tests were used. When there was a difference in variances between the groups (*F* test), the Satterthwaite method was used.

Table 1.—Comparison of firm characteristics for respondents by study year (N=359 in 2010, 325 in 2011, 307 in 2012, and 244 in 2013)

Firm characteristics	2010	2011	2012	2013
Main products produced:	Percent			
Kitchen/bath cabinets	36	44	41	42
Household furniture	8	7	13	14
Architectural fixtures	8	7	10	8
Molding/millwork	13	11	11	11
Dimension or components	5	3	4	6
Office/hospitality/contract furniture	7	6	7	7
Other	23 ^a	21	14	12
Total sales:				
Less than \$1 million	45	52	56	56
\$1 to \$10 million	33	24	27	21
\$11 million or more	23	24	17	23
Wood material used:				
Mostly solid wood	29	26	28	36
Mostly wood composite or engineered products	12	9	12	8
A combination of solid and composites	57	62	59	54
Other	2	3	1	2
Price-point of primary product:				
Low to medium	8	11	10	6
Medium	28	24	20	25
Medium to high	54	54	56	56
High	10	11	16	11

^a Yearly totals may not sum to 100 percent due to rounding

RESULTS AND DISCUSSION

Changes in Sales Performance

Analysis of year-over-year sales performance across the 4 years of the study revealed improvement in terms of those firms reporting positive changes. In 2009, a full 60 percent of respondents indicated that their sales volume was “off by 20% or more” from 2008, and 80 percent indicated that sales volume had declined overall. For 2012, the number in the “off by 20% or more” category had decreased to just 15 percent (Fig. 3). Most respondents were in a category on the positive side of the ledger in 2012. Twenty-five percent of respondents realized somewhat better sales (“up by 10%”) from 2011 to 2012. Overall, 58 percent of respondents reported an increase in sales volume from 2011 to 2012, which corresponds with the mild improvements witnessed in construction at the national level (Fig. 1). Although these trends are generally positive, 31 percent of respondents still reported losing sales volume from 2011 to 2012, with 2011 having already been a challenging year for many manufacturers.

Another dynamic at work is a decline in the number of establishments in the overall secondary woodworking industry since the housing downturn began. For example, the number of establishments in the wood kitchen cabinets and countertops (NAICS 33711) and millwork (NAICS 32191) industries declined by 22 and 19 percent, respectively, from 2008 to 2012 (U.S. Bureau of

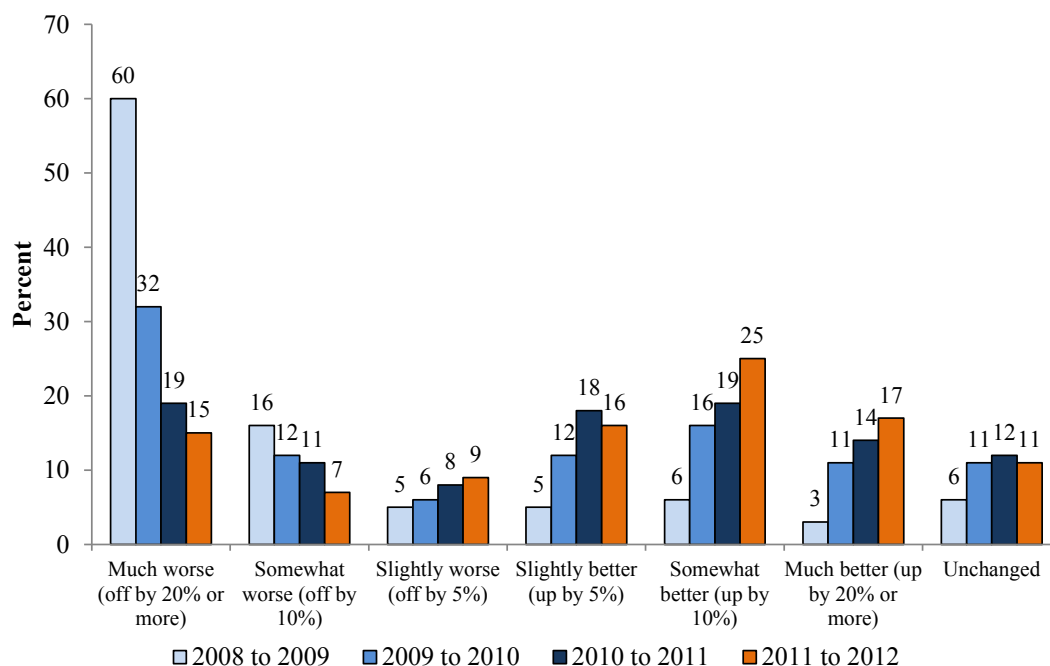


Figure 3.—Responses to the question, “Compared to the previous year, last year’s sales volume was . . .” by year (p -value < 0.01 for chi-square test of associated frequency counts).

Labor Statistics 2013). Having fewer competitors helps surviving firms increase their sales, even in a down market.

Changes in Markets Served

Respondents were asked to indicate the proportion of their production volume associated with several construction markets. Results are shown in Figures 4, 5, and 6. Although it appeared there was some movement back into single family residential construction in 2012 (as the relative number of companies in the “0%” category declined to near 2009 levels), the results were not significant (Fig. 4). For 2012, 26 percent of respondents indicated that “61-100%” of their production volume was directly associated with the new single family residential construction market. At the same time, the residential repair and remodeling market also maintained an important share in the “61-100%” production volume category in 2012, staying nearly level with 2011 at 27 percent (Fig. 5). Although the general trend was not significant for residential repair and remodeling across all four categories, the results became significant when combining categories to isolate the “61-100%” category, which was higher after 2010. This outcome is likely a reflection, in part, of some movement out of single family construction. As an indication of the importance of remodeling markets, only small percentages of respondents indicated no production dedicated to repair and remodeling, in contrast with single family construction, where nearly a quarter of respondents consistently reported having no production activity.

Last, nonresidential construction remained important for some woodworkers, but a majority continued to indicate this market accounted for no more than 20 percent of their production volume (Fig. 6). Still, for 2012, 77 percent of respondents reported at least some activity in nonresidential construction, which represents an alternative to residential construction. However, there was no discernible trend evident for 2012 in this market. A decline in the “1-20%” category was offset by small increases in the other categories, and the changes were not significant overall.

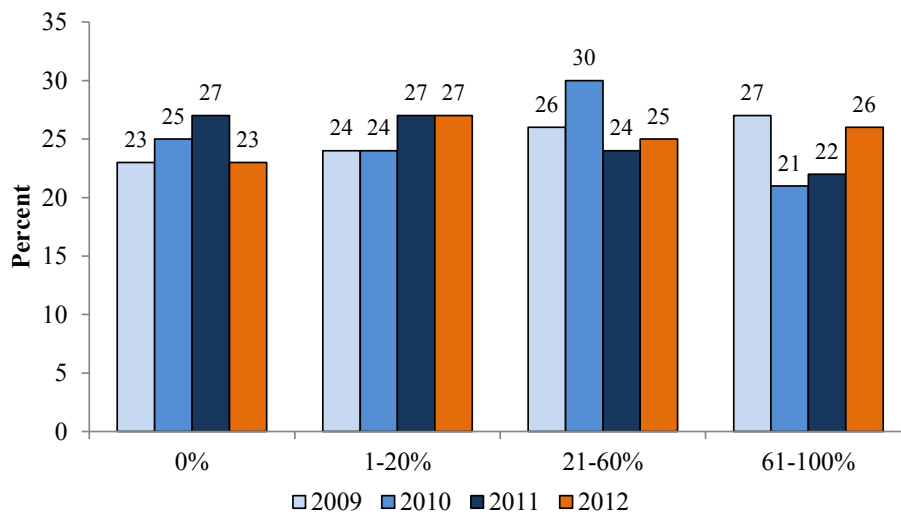


Figure 4.—Proportion of production volume associated with the new single family residential construction market (p -value = 0.519 for chi-square test of associated frequency counts).

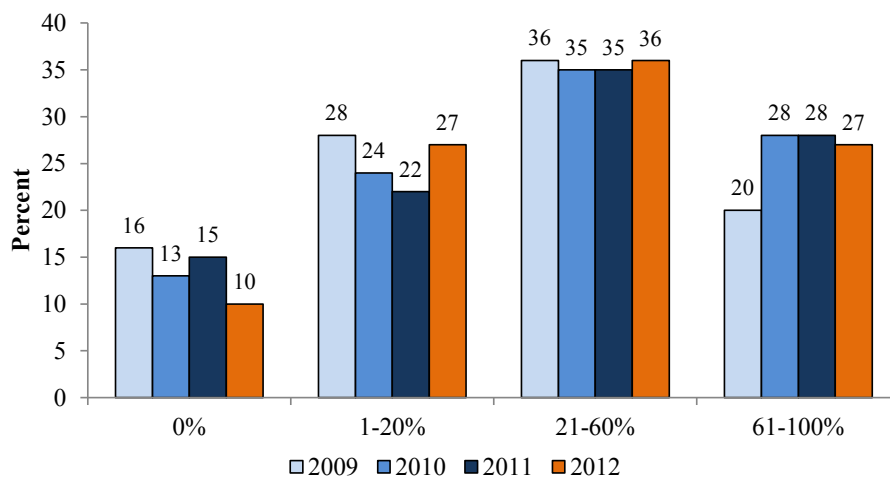


Figure 5.—Proportion of production volume associated with the residential repair and remodeling market (p -value = 0.141 for chi-square test of associated frequency counts; p -value = 0.036 for chi-square test when combining “0%-60%” into one category).

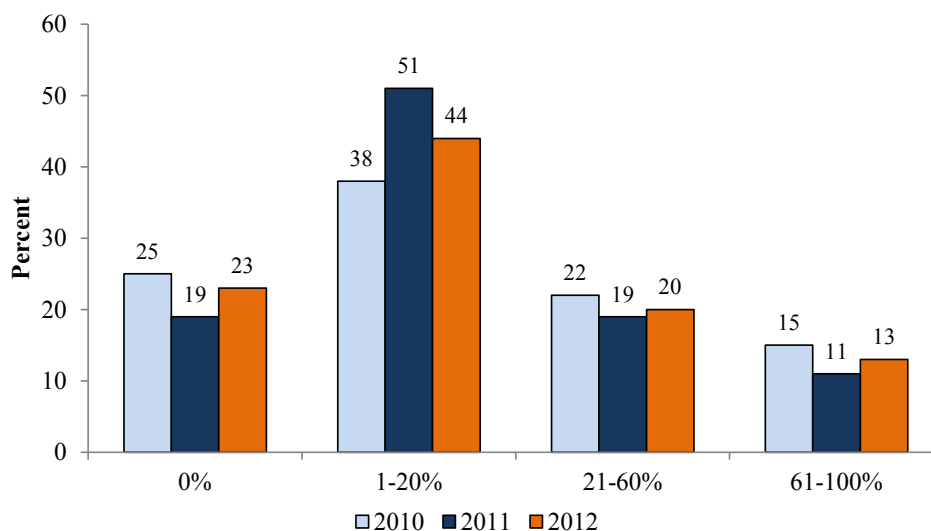


Figure 6.—Proportion of production volume associated with the nonresidential/business construction market (p -value = 0.137 for chi-square test of associated frequency counts).

Perceived Reasons for Sales Volume Increases or Declines

Each year, respondents were asked about the perceived reasons for their increased sales volume (for those firms indicating an increase; see Figure 3). Generally, respondents did not rate highly any of the possible reasons offered in the studies (Fig. 7). One of the highest rated reasons, especially in 2012, was that companies simply grew in proportion with the overall economy. This item was significantly different across years, suggesting some perceived improvement in economic conditions for 2012. Most of the actions or behaviors for increasing sale volume at the individual firm level (i.e., offering new services, entering new markets, developing new products, and productivity improvements) were rated similarly to slightly lower than in previous years, but only development of new products was significant across years. New product development was rated relatively high in 2011 but declined in 2012, perhaps as a response to improving demand conditions for surviving companies.

Generally, respondents had stronger views regarding causes for sales volume declines (for those firms indicating a decline; see Figure 3). Downturns in the housing market and in remodeling expenditures consistently were rated as the most substantial reasons for sales volume declines, and both were significant across years (Fig. 8). A declining trend was evident more recently, suggesting

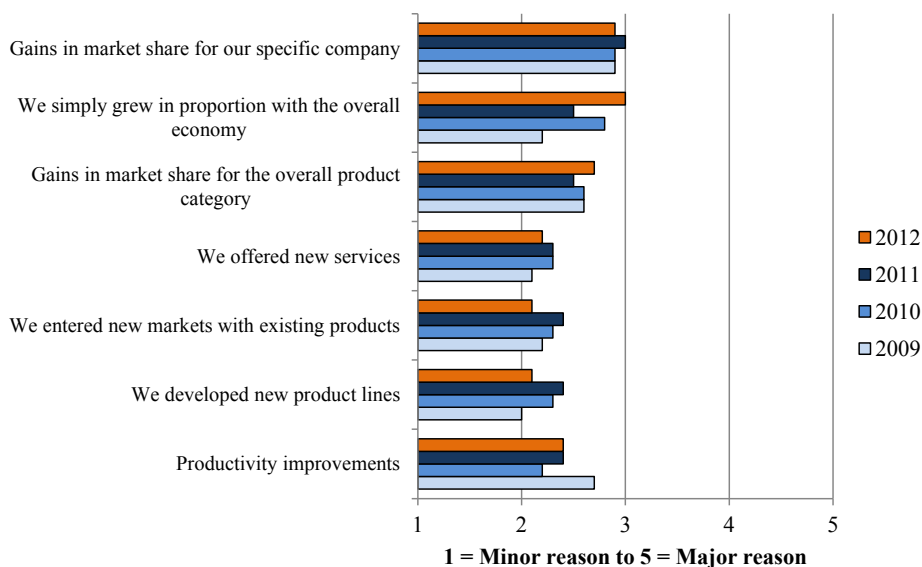


Figure 7.—Perceived reasons for sales volume increases (for those firms indicating an increase in sales, by year). The following were significant at $\alpha = 0.10$ based on a one-way ANOVA across years: “We simply grew in proportion with the overall economy”; and “We developed new product lines.”

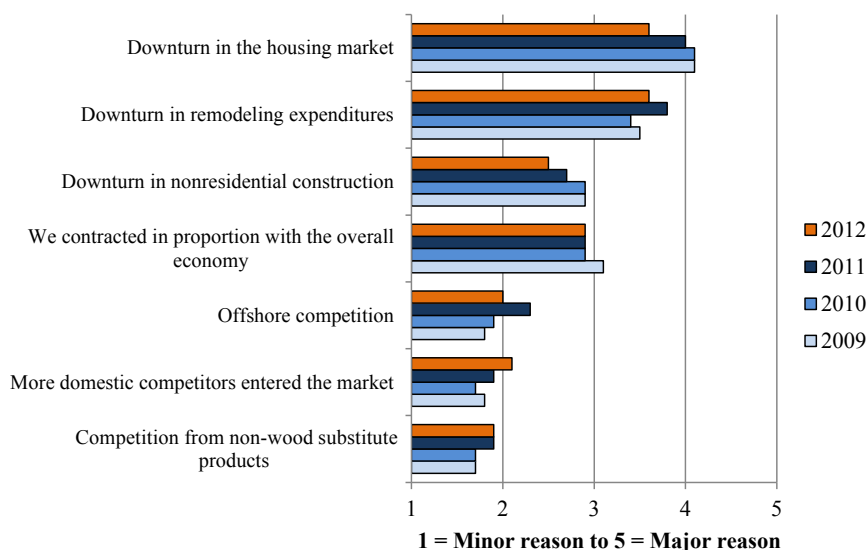


Figure 8.—Perceived reasons for sales volume declines (for those firms indicating a decrease in sales, by year). The following were significant at $\alpha = 0.10$ based on a one-way ANOVA across years: “Downturn in the housing market”; “Downturn in remodeling expenditures”; “Offshore competition”; and “More domestic competitors entered the market.”

Table 2.—Mean responses to question: “How frequently did your company use each of the following types of marketing communications (over the last three years)?” for small firms (n=136) and large firms (n=68), and t-test results

Type of marketing communication	Small firms (mean) ^a	Small firms (sd)	Large firms (mean)	Large firms (sd)	t	p
Referrals	4.2 ^b	(1.0)	3.7	(1.0)	3.11	0.002^c
Contacting initial leads to follow up	3.3	(1.3)	3.3	(1.1)	-0.25 ^d	0.801
Calling past customers	2.6	(1.3)	3.5	(1.2)	-4.88	<0.001
Visiting potential or past customers	2.5	(1.2)	3.6	(1.2)	-6.37	<0.001
Social networking	2.4	(1.4)	2.7	(1.2)	-1.52	0.130
Updates/postings to company Web site	2.4	(1.5)	3.5	(1.2)	-5.69 ^d	<0.001
Emails sent to past customers	2.1	(1.3)	2.9	(1.3)	-4.27	<0.001
Print advertising	2.0	(1.2)	2.1	(1.1)	-0.76	0.449
Emails sent to potential new customers	2.0	(1.2)	2.9	(1.2)	-5.19	<0.001
Cold-calling potential new customers	1.8	(1.2)	2.8	(1.3)	-5.76	<0.001
Mailings sent to potential new customers	1.5	(0.9)	2.3	(1.2)	-4.59 ^d	<0.001
Trade show exhibits	1.5	(0.9)	2.5	(1.4)	-5.52 ^d	<0.001
Mailings sent to past customers	1.5	(1.0)	2.5	(1.4)	-5.06 ^d	<0.001

^a Means are ordered from the most to least frequently used by small firms.

^b Scale ranged from 1 = Never to 5 = Frequently.

^c Boldface values indicate significant differences in responses between small firms and large firms.

^d Based on the Satterthwaite method (for unequal variances)

some perceived improvement in residential markets (especially the housing market) in 2012. There also was a decrease in “downturn in nonresidential construction” as a reason for declining sales, but the difference across years was not significant. Interesting, too, was the significant uptick in answers suggesting that the presence of more domestic and offshore competition were causes of respondents’ sales declines. These factors were rated relatively low overall, but the trend could indicate that woodworking markets are being viewed by more firms as potentially profitable.

Marketing Communications

It was noted through qualitative analysis in the early years of the study (Bumgardner et al. 2011) that revenue-generating actions seemingly were being taken more frequently than cost-reduction activities in the height of the downturn to help firms maintain or grow their sales volume. To investigate this observation further, a line of questions in the 2012 study asked about the types of marketing communications secondary woodworkers had used over the last 3 years. Furthermore, we were interested in comparing smaller firms (1-19 employees) with larger firms (20 or more employees) to discern any possible differences, given the importance of revenue generation/cash flow to smaller firms during economic downturns (Latham 2009). Results are shown in Table 2. Across nearly all items, large firms engaged in marketing communications significantly more often than small firms. Among the highest rated communication types, large firms were especially more likely than small firms to visit potential or past customers, call past customers, and post new information to their respective Web sites. In addition, large firms tended to use email more often than small firms.

Conversely, referrals and contacting initial leads (e.g., walk-ins, calls received) to follow up were especially important to small firms. Referrals were the only item rated significantly higher by small firms than large firms. Overall, past customers appeared to be especially important to woodworkers in

the current business environment, whether being called or visited directly (especially by larger firms) or more indirectly as a source of referrals.

A related question asked respondents to indicate whether their respective firms were relying more on cost reductions or the seeking out of new revenue sources as the “primary” strategy to weather the housing downturn. The results indicated that most believed they were pursuing both equally (57 percent), whereas 18 percent were primarily seeking cost reductions and 12 percent were primarily seeking new revenue sources. The remaining 13 percent of firms indicated they were doing neither. No differences were detected across these categories between smaller and larger firms (as defined above) based on a chi-square test (p -value = 0.254).

SUMMARY AND CONCLUSIONS

Sales volume performance improved over the last 3 years for the secondary woodworking manufacturers that survived the housing downturn. Relatedly, conditions in housing and remodeling markets, as well as the overall economy, were perceived to have improved from previous years. Still, a substantial proportion of firms (about a third) reported losing sales volume in 2012 compared to 2011, suggesting that improvements were uneven and that challenges remain. For example, secondary data suggest that the number of U.S. cabinet and millwork establishments in operation has declined by nearly a quarter since 2006. Companies reported adapting to lower demand from housing markets by developing new products and remaining in close contact with past and potential new customers. Nearly all respondents reported that at least some of their production volume each year was geared toward remodeling markets, more so than for single family housing or nonresidential construction.

Referrals, following up on leads, and keeping in contact with past customers appeared to be especially important modes of marketing communication for secondary woodworkers in the current business environment. Keeping a current Web site also was relatively important for larger firms. Overall, small firms used most types of marketing communications less frequently than did larger firms, consistent with the notion that smaller firms face unique challenges in carrying out many marketing activities (Huang and Brown 1999). Thus, even though marketing activities are perceived to be especially important to success by small firms (Buehlmann et al. 2013), it seems that such firms often are relying on a limited set of basic marketing activities, i.e., referrals and following up on leads, to help generate sales and cash flow. Staying close to customers is an important advantage for smaller firms, and such focus is seen as a way to help them garner referrals to other potential customers (Gilmore et al. 1999). Cost reductions also were important to all secondary manufacturers as a means for surviving the downturn.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

SEASONAL INFLUENCE ON OHIO HARDWOOD STUMPAGE PRICE TRENDS

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Abstract.—The average annual percentage rates of change in real sawtimber stumpage prices from 1978 through 2012 (dollars per thousand board feet, Doyle) for the 10 commercial hardwood species of Ohio were determined. Each species was then further examined for differing trend lines between the spring and fall reporting periods. Annual real rates of change ranged from -1.10 percent for basswood to 2.13 percent for hard maple. Across species, 37 percent of the annual price highs occurred in spring; 59 percent of the annual highs occurred in fall. Approximately 75 percent of all price cycles lasted three seasons or less. However, no spring/fall seasonality effect in reported prices was observed for any of the 10 species. Although the relatively low rates of change suggest Ohio has historically been a buyer's market, the seasonal variation characteristic of a buyer's market was not present in Ohio's hardwood stumpage price trends.

INTRODUCTION

Stumpage price is the residual of lumber price less milling, harvesting, and hauling costs and the profits derived from each process. Stumpage prices are fundamentally driven by supply and demand, which can exhibit seasonal influences. Mead (1964) found demand for specific grades of lumber varied due to the seasonal needs of downstream purchasing sectors. Dahal and Mehmood (2005) stated one determinant of a timber tract's bid price was seasonality. For example, increasing procurement activities in fall before wetter and colder weather can help guard against potential emergency purchases in periods of low inventories (Gallagher 2003). Prudent supply management can also protect sawmills from competing against pulp mills for marginal quality sawtimber in times of reduced timber availabilities (Luppold 1996).

Price trends provide production forestry clientele with a measure of timber's economic performance. The Ohio Timber Price Report, which dates to 1960, has provided semiannual sawtimber price survey results for 10 hardwood species since 1978: ash (*Fraxinus americana*), basswood (*Tilia americana*), black cherry (*Prunus serotina*), hard and soft maples (*Acer saccharum* and *A. rubrum*), hickory (e.g., *Carya cordiformis*), black walnut (*Juglans nigra*), red and white oaks (*Quercus rubra* and *Q. alba*), and yellow-poplar (*Liriodendron tulipifera*). Participants in the surveys include foresters, loggers, mills, and timber buyers, with pricing data collected for the spring (May) and fall (November) seasons. Gathering prices from several sources helps provide an overall picture of the marketplace that reflects differing perspectives.

How the Ohio timber market has been affected by past seasonal influences has yet to be determined. The objective of this study was to determine the 1978 through 2012 sawtimber stumpage price trends of Ohio's commercial hardwood species. Price trends for the species were then evaluated for seasonality by using both descriptive statistics and regression models that contained seasonal indicator variables.

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METHODS

Statewide average sawtimber stumpage prices (dollars per thousand board feet, Doyle) reported for each species were used in this study. Nominal prices were adjusted for inflation to 2012 constant dollars by using the Producer Price Index for all commodities (U.S. Bureau of Labor Statistics 2013). Real prices for each species were averaged, with the amount of variation occurring about the means recorded.

Trend analyses were conducted for each species across their entire time series as well as for the spring and fall seasons from 1978 through 2012 at a significance level of $\alpha = 0.05$. Determining price trends first required transforming price data for each species to their natural logarithms. The following equation describes the simple linear regression model used in SAS version 9.2 (SAS Institute Inc., Cary, NC) to find the rate of price change:

$$Y = \beta_0 + \beta_1 X_t + \varepsilon$$

where

$Y = \ln(P_t)$, with P_t being the average price of a species paid at time t ;

β_0 = the intercept of the line, represents the initial price in a series;

β_1 = the slope, or the continuous rate of change in price;

X_t = time t in a series, which was numbered sequentially (1978/spring = 1.25, 1978/fall = 1.75, 1979/spring = 2.25...2012/spring = 35.25, 2012/fall = 35.75); and

ε = the error of the model.

The average annual percentage rate of change (APR) was then found by using the following equation (Wagner and Sendak 2005):

$$APR = (e^{\beta_1} - 1) * 100$$

Time-series data are often autocorrelated, which means the residuals of the data are similar to adjacent points (Moineddin et al. 2003). Linear regression, though, requires residuals of observations to be independent of one another (Kutner et al. 2004). The Durbin-Watson test statistic, which tests the assumption of independence of the residuals of a linear regression (Albright et al. 1999), found significant autocorrelation existed across price data for each species.

Maximum Likelihood stepwise autoregression was used to account for the autocorrelation. Five lag variables were assigned by using a backward stepwise approach, and nonsignificant variables were removed one at a time. Any remaining variable(s) were those significantly contributing to the model. One lag variable accounted for the autocorrelated data of each price series. Autoregression model errors were reported as percent root mean square error (% RMSE) by using the equation below (Linehan et al. 2003):

$$\%RMSE = (e^{RMSE} - 1) * 100$$

Each species was then examined for seasonal price differences. First, annual price highs occurring in both spring and fall were counted for each species, with their frequency percentage determined. Price

cycle lengths, which were defined here as the elapsed time between adjacent price peaks, were also established. A seasonality effect in the price trends of each species was then investigated by testing for trend line intercept (initial price) and slope (APR) differences. This was done by adding an indicator variable to a regression function to differentiate between seasons by using the following equation:

$$Y = \beta_0 + \beta_1 x_{1t} + \beta_2 x_{2t} + \beta_3 x_{1t} x_{2t} + \varepsilon$$

where

$Y = \ln(P_t)$, with P_t being the average price of a species paid at time t ;

X_{1t} = time t in a series;

X_{2t} = the indicator variable (1 for spring, 0 for fall);

$X_{1t}X_{2t}$ = the interaction term;

β_0 , β_1 , β_2 , and β_3 were model coefficients; and

ε = the error of the model.

The indicator variable coefficient, β_2 , tested whether the initial prices between spring and fall were different. The interaction coefficient, β_3 , tested for an APR difference between seasons. Autocorrelation was again examined and accounted for as described previously.

RESULTS

Overall Prices and Price Trends

The average real prices (constant 2012 dollars) for each species from 1978 through 2012 are provided in Figure 1. Average prices ranged from \$175 for hickory to \$769 for walnut. Variability about the means ranged from 19.7 percent to 44.7 percent. Overall, five species had price variations of less than 25.0 percent from their means, and five had greater than 25.0 percent. Prices of hickory were the most stable, varying 19.7 percent from its mean price. Cherry and hard maple each exhibited variations of at least 40.0 percent from their respective means.

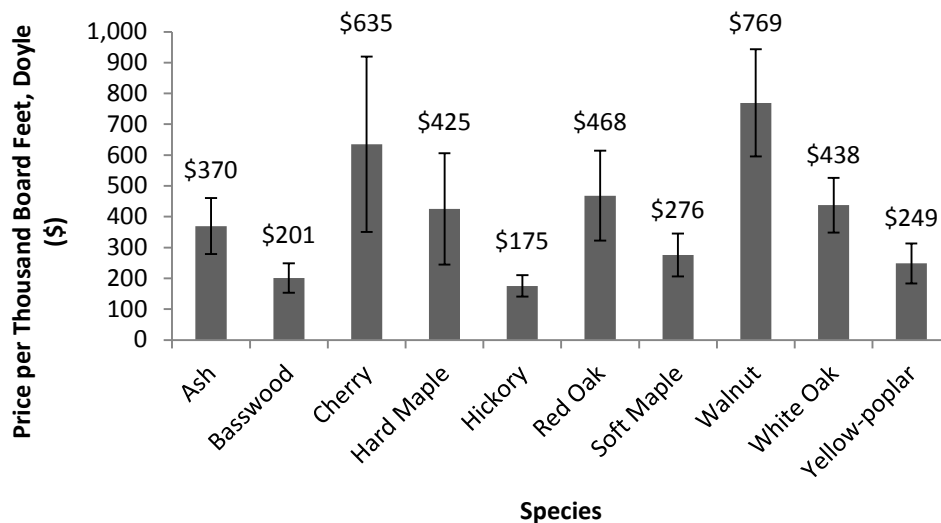


Figure 1.—Average real prices (constant 2012 dollars) from 1978 through 2012 for 10 Ohio hardwood species. Error bars are the standard deviation.

The APRs of real prices from 1978 through 2012 ranged from -1.10 percent for basswood to 2.13 percent for hard maple and are presented in Figure 2. Two species, ash and basswood, had overall negative APRs. Three species had APRs of at least 1 percent: hard maple, cherry, and white oak. Prices of only two species, though, were found to be changing at significant annual rates: basswood ($p = 0.05$) and white oak ($p = 0.04$).

Spring Price Trends

Real price APRs in spring from 1978 through 2012 ranged from -0.81 percent for basswood to 2.37 percent for hard maple (Fig. 3). Ash and basswood had negative APRs. Hard maple, cherry, and white oak showed price increases of at least 1 percent annually. Hard maple ($p = 0.04$) was the only species with a significant APR for the spring price series.

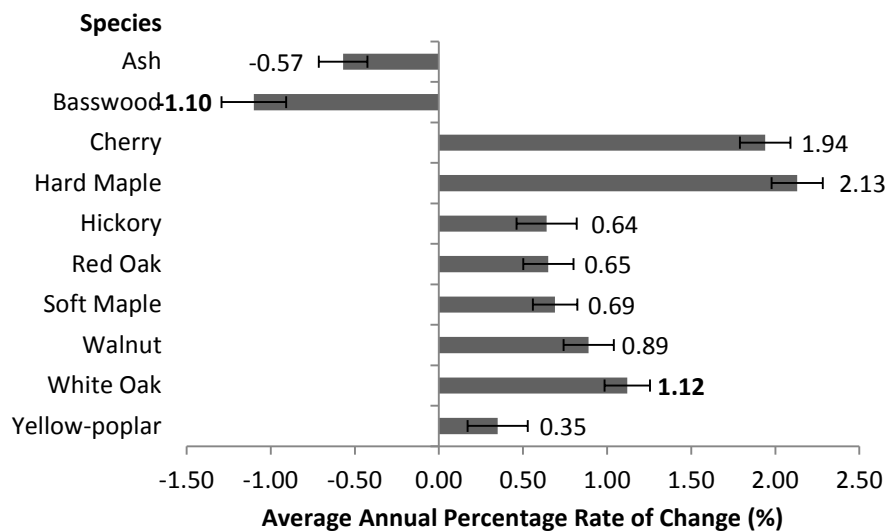


Figure 2.—Overall annual percentage rates of change (APR) in real prices from 1978 through 2012 for 10 Ohio hardwood species. Boldface APRs were significantly different from zero at $\alpha = 0.05$. Error bars are percent root mean square error.

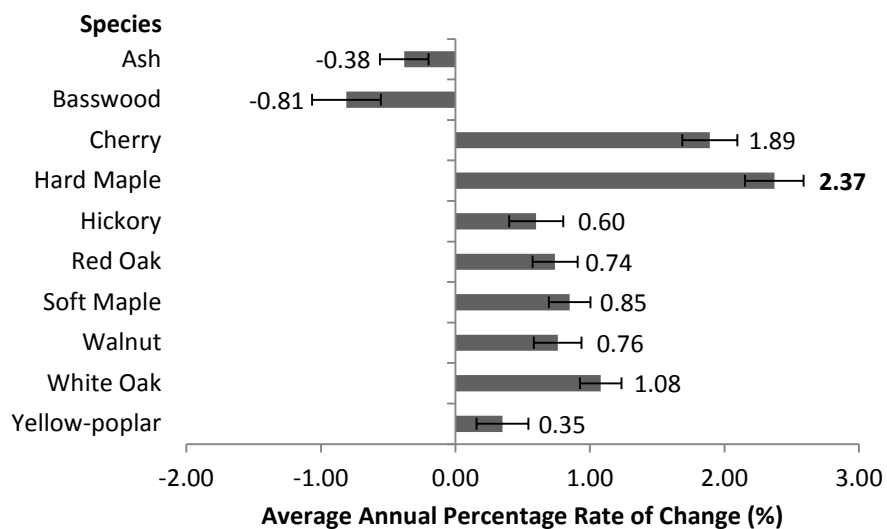


Figure 3.—Annual percentage rates of change (APR) in spring real prices from 1978 to 2012 for 10 Ohio hardwood species. Boldface APRs were significantly different from zero at $\alpha = 0.05$. Error bars are percent root mean square error.

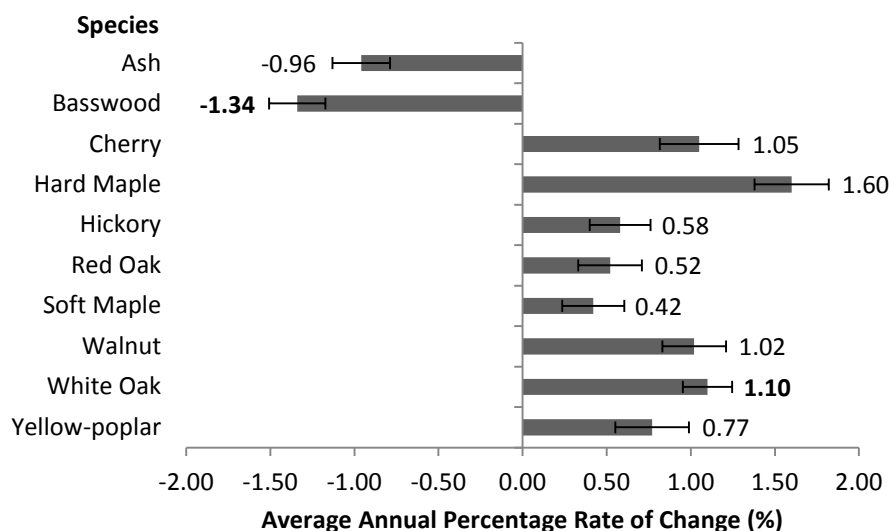


Figure 4.—Annual percentage rates of change (APR) in fall real prices from 1978 to 2012 for 10 Ohio hardwood species. Boldface APRs were significantly different from zero at $\alpha = 0.05$. Error bars are percent root mean square error.

Fall Price Trends

Real prices in fall from 1978 through 2012 changed at annual rates of between -1.34 percent for basswood to 1.60 percent for hard maple (Fig. 4). Ash and basswood again had negative APRs. Hard maple, white oak, cherry, and walnut had fall APRs of at least 1 percent. For two species, basswood ($p = 0.04$) and white oak ($p = 0.03$), fall prices changed at significant annual rates.

Seasonality in Stumpage Price Trends

Tables 1 and 2 provide descriptions of any seasonal volatility occurring in hardwood stumpage prices from 1978 through 2012. Across species, 37.2 percent of the annual price highs occurred in spring (Table 1). No individual species had price highs occur in spring more than 50.0 percent of the time. Annual price highs occurred in fall at least 60.0 percent of the time for seven species. Overall, annual price highs occurred in fall 59.4 percent of the time.

Table 2 describes the price cycle lengths observed in Ohio hardwoods. The most common cycle observed in six species was the shortest possible, 1 year. Most of the cycle lengths for walnut, ash, hard maple, hickory, and yellow-poplar were only 1 year. Fifty percent of white oak's price cycles were intervals of at least 2 years. Across species, 44.3 percent of all price cycles lasted only 1 year, and 75.4 percent were three seasons or less in length.

Within each species, the seasonal trend lines were tested to determine if potential differences existed between the initial prices or APRs of each time series (Table 3). Initial prices did not differ between seasons ($0.37 \leq p \leq 0.94$), nor did APRs ($0.46 \leq p \leq 0.97$). Given the lack of intercept and slope differences, seasonal timber prices in Ohio have not historically exhibited any significant spreads between price levels, nor have the trends been diverging or converging.

Table 1.—Frequency of annual price highs occurring in spring and fall from 1978 through 2012 for 10 hardwood species in Ohio

Species	High price in spring	High price in fall	Prices equal
<i>Frequency, %</i>			
Ash	34.4	65.6	0.0
Basswood	37.5	62.5	0.0
Cherry	28.1	71.9	0.0
Hard maple	50.0	46.9	3.1
Hickory	34.4	62.5	3.1
Red oak	31.3	65.6	3.1
Soft maple	31.3	68.8	0.0
Walnut	43.8	40.6	15.6
White oak	50.0	43.8	6.3
Yellow-poplar	31.3	65.6	3.1
All species	37.2	59.4	3.4

Table 2.—Frequency of cycle lengths between price highs for 10 Ohio hardwood species

Species	Cycle length (and equivalent number of seasons)				
	1 year (2 seasons)	1.5 years (3 seasons)	2 years (4 seasons)	2.5 years (5 seasons)	3 years (6 seasons)
<i>Frequency, %</i>					
Ash	54.5	9.1	27.3	9.1	0.0
Basswood	20.0	50.0	30.0	0.0	0.0
Cherry	45.5	36.4	18.2	0.0	0.0
Hard maple	54.5	27.3	9.1	9.1	0.0
Hickory	54.5	18.2	9.1	9.1	9.1
Red oak	40.0	40.0	20.0	0.0	0.0
Soft maple	36.4	54.5	9.1	0.0	0.0
Walnut	58.3	25.0	16.7	0.0	0.0
White oak	12.5	37.5	37.5	12.5	0.0
Yellow-poplar	54.5	18.2	18.2	9.1	0.0
All species	44.3	31.1	18.9	4.7	1.0

Table 3.—Within-species seasonal price comparisons of trend line intercepts (initial price levels) and slopes (average annual percentage rates of change [APR]). Model errors are presented as percent root mean square error (%RMSE).

Species	Initial price, p-value	APR, p-value	%RMSE
Ash	0.50	0.56	17.41
Basswood	0.94	0.75	21.39
Cherry	0.37	0.46	21.67
Hard maple	0.40	0.56	21.80
Hickory	0.82	0.84	18.71
Red oak	0.58	0.72	17.09
Soft maple	0.62	0.65	16.97
Walnut	0.65	0.47	17.30
White oak	0.56	0.92	14.98
Yellow-poplar	0.77	0.97	18.71

DISCUSSION

Since 1978, real prices have not changed significantly across species. Overall price trends of only two species, white oak at 1.12 percent and basswood at -1.10 percent, were significantly different from zero. White oak is distributed widely throughout the Central Hardwood region and has a long-held preferred status in consumer and industrial products. Basswood has had limited demand in Ohio for some time. No species contained both significant spring and fall APRs. Though not overly encouraging, the rates of price change were at or near the average inflation rate.

Potential seasonality was first investigated by using descriptive methods. Measuring the variation about the means provided some measure of the volatility in prices. Five species had coefficients of variation (CVs) of at least 25.0 percent, and five others had CVs less than 25.0 percent. However, the presence of year-to-year variation may have masked any seasonal differences in the price data for the species. The frequency of seasonal price highs occurring in spring and fall were then determined. Annual prices were generally found to be lower in spring and higher in fall. Additionally, price cycle lengths were commonly three seasons or less, with 1 year being the most common. This pattern was similar to the volatility seen in Illinois's timber price trends (Campbell and White 1989).

By these criteria, Ohio timber market participants perhaps have attempted to purchase or market stumpage more aggressively in fall. For all species, though, the lack of trend line intercept and slope differences across the entire price ranges of both seasons indicated spring/fall seasonality was not a factor in determining sawtimber market price. A seasonal effect was also absent in Vermont's stumpage price trends (Sendak and McEvoy 1989).

However, the lack of seasonality found in Ohio may have been locally driven and not indicative of the greater timber market. Timber prices in the southern United States, for example, have been found to be higher in fall than other seasons in anticipation of weather-induced harvesting restrictions (Dahal and Mehmood 2005). Mead (1964) concluded a significant negative correlation existed between seasonal price variation and lumber grade, which in turn could indicate the potential for a seasonal effect within the price trends of one or more individual log or tree grades of a species. Future research should address this issue.

The relatively low rates of price change suggest a historical buyer's market for Ohio timber. A sale in this type of marketplace is typically one of a smaller private tract, where price is negotiated between the seller and an individual buyer. Formal bidding by multiple parties, which is advantageous to the landowner, is often the exception rather than the norm. An environment is thereby created where prices favorable to the buyer can endure (Campbell and White 1989). Another indicator of a buyer's market is price volatility, where local conditions can result in wider price fluctuations between reporting periods than those found in the greater region. Annual price highs occurred in Ohio more frequently in fall and cycle lengths often lasted no more than three seasons. However, the trend lines for Ohio's spring and fall price series were not distinct for any species. It was concluded Ohio's hardwood stumpage price trends were not exhibiting a significant seasonal effect.

ACKNOWLEDGMENTS

The Ohio Timber Price Report is financially supported with federal funding provided by the Renewable Resources Extension Act. I thank the anonymous reviewer for his or her constructive comments.

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NORTHERN RED OAK VOLUME GROWTH ON FOUR NORTHERN WISCONSIN HABITAT TYPES

Michael Demchik, Kevin M. Schwartz, Rory Braun, and Eric Scharenbrock¹

Abstract.—Northern red oak (*Quercus rubra*) grows across much of Wisconsin. Using site factors to aid in prediction of volume and basal area increment facilitates management of red oak and other species of interest. Currently, habitat type (Wisconsin Habitat Type Classification System) is often determined when stands are inventoried. If habitat type were strongly related to annual volume and basal area increment, it would be a valuable tool in making management decisions. The objective of this study was to determine if individual tree annual volume and basal area increment (last 20 years) of northern red oak was related to habitat type. Four common habitat types were selected: AAt (*Acer saccharum*/*Athyrium filix-femina*; 10 sites), ATM (*Acer saccharum*-*Tsuga canadensis*/*Maianthemum canadense*; 8 sites), AVb (*Acer saccharum*/*Viburnum acerifolium*; 7 sites), AVDe (*Acer saccharum*/*Vaccinium angustifolium*—*Desmodium glutinosum*; 8 sites). On each site, increment cores from 10 northern red oak trees were used to determine individual tree basal area and volume increment. Site index was also determined for each site. Generally, ATM grouped with AVb, and AAT grouped with AVDe. Of the four habitat types, three (AAt, AVDe, and ATM) had quite predictable basal area and volume increment (tight confidence intervals); however, AVb was much more variable. Habitat type, by itself, may be adequate for planning purposes on some habitat types, but on other habitat types, additional site factors may be necessary.

INTRODUCTION

Oak (*Quercus* spp.) is very common in Wisconsin, covering more than 3 million acres. It represents >2 billion cubic feet of growing stock (Perry et al. 2008) and 30 percent of the total saw log harvest (Reading and Whipple 2007). Although oak is of great current importance on many sites, regeneration on high quality sites is unpredictable (Beck and Hooper 1986, Nowacki and Abrams 2008). Consequently, this cover type has declined on high quality sites (Perry et al. 2008). Because this decline has significant ecological consequences, the Wisconsin Department of Natural Resources (WI DNR) has listed oak regeneration as one of its statewide objectives (WI DNR 2004).

The habitat type classification system (HTCS; Kotar et al. 2002) has been used by foresters across Wisconsin as a method of classifying sites. It is based on using understory plants to predict climax vegetation communities. The system was a logical outgrowth of work by Daubenmire and Daubenmire (1968) and Daubenmire (1976, 1981), which used vegetation to predict productivity and other site factors. The system itself is easy to use, but its basic ecological foundation is Clementsian successional theory, which has been the subject of numerous revisions in the last 40 years (Cook 1996). Even with this constraint, much of the Wisconsin state forest land is already classified, as is some private land; thus, this system could be of great use as an indirect way to predict other parameters. Schwartz (2012) used habitat type as a predictor for oak advance regeneration under two canopy stocking conditions. Both overstory stocking and habitat type had a significant

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Table 1.—Volume growth and basal area increment for northern red oak growing on four habitat types in northern Wisconsin (with means presented as $\pm$ 90-percent confidence intervals, and significant differences between habitat types at $\alpha = 0.1$ indicated by superscripts a, b, c)

Habitat type	n	Annual increment (most recent 20 years)	
		Volume (ft ³ /tree)	Basal area (ft ² /tree)
AAAt	10	0.48 $\pm$ 0.05 ^a	0.019 $\pm$ 0.002 ^a
AVDe	8	0.52 $\pm$ 0.12 ^{ab}	0.020 $\pm$ 0.004 ^a
AVb	7	0.79 $\pm$ 0.21 ^{bc}	0.030 $\pm$ 0.007 ^b
ATM	8	0.83 $\pm$ 0.09 ^c	0.031 $\pm$ 0.003 ^b

impact on oak advance regeneration. In contrast, Bakken and Cook (1998) found habitat types to be very poor at predicting overall regeneration potential (due in part to large background variability in regeneration). Of note is that these two studies addressed completely different habitat types with no overlap, so direct comparison is impossible.

These contrasting studies suggest that HTCS may have variable utility in classifying sites depending on the parameter of interest. The objective of our study was to determine if annual volume and basal area increment (last 20 years) of northern red oak was related to HTCS.

METHODS

See Schwartz (2012) for specifics on site selection. Generally, WI DNR provided sites that were “at least 50 years old, five acres or larger and supported at least 40-50% oak in the overstory” (Schwartz 2012). We selected a subset of these sites which comprised four of the northern habitat types: AAAt (*Acer saccharum*/*Athyrium filix-femina*; 10 sites), ATM (*Acer saccharum*-*Tsuga canadensis*/*Maianthemum canadense*; 8 sites), AVb (*Acer saccharum*/*Viburnum acerifolium*; 7 sites), and AVDe (*Acer saccharum*/*Vaccinium angustifolium* – *Desmodium glutinosum*; 8 sites; Kotar et al. 2002). At each site, 10 dominant or codominant northern red oak (*Q. rubra*) trees were increment cored and their height was measured with a clinometer. Cores were mounted to wooden core blocks and measured with a digital caliper in 5-year intervals for 0-40 years and from 40 years to pith for the remaining rings. To estimate cubic foot volume, equations from Hahn and Hansen (1992) were used. Analysis of variance (ANOVA) with habitat type as the main factor and measured site index (SI) as the response variable was used to determine whether SI varied consistently with habitat type. To determine impact of SI, SI was used as the predictor in a linear regression with annual basal area increment as the response variable. Unbalanced ANOVA with main factor of habitat type and a covariate of proportion canopy cover was planned, but Levene’s test for equality of variance demonstrated unequal variance for both response variables of interest: annual total cubic foot increment and annual basal area increment. The AVb habitat type had a much higher variance than the others (see Table 1 for confidence intervals). As a consequence, means were compared by using 90-percent confidence intervals.

RESULTS

Site index explained 26 percent of the variance in annual basal area increment ($p = 0.003$, $r^2 = 0.26$). Because SI has been consistently used as a predictor of growth, this result is not surprising. Site

Table 2.—Moisture and nutrient regime of four habitat types in northern Wisconsin (information from Kotar et al. 2002)

Habitat type	Moisture regime	Nutrient regime
AAAt	Dry-mesic	Medium to rich
AVDe	Dry-mesic	Medium
AVb	Dry-mesic	Medium to rich
ATM	Mesic to dry-mesic	Medium

index was significantly different between habitat types ($p = 0.013$, $r^2 = 0.23$) with ATM having a significantly greater SI than all other habitat types; however, none of the other three habitat types was significantly different in SI.

For annual volume increment, AAAt and AVDe were not significantly different from each other although AAAt produced less volume growth than either AVb or ATM. AVDe produced less volume than ATM but was not significantly different from AVb. AVb and ATM were not significantly different in volume production (Table 1).

For annual basal area increment, AAAt and AVDe had the lowest basal area increment during the last 20 years of growth, but they were not different from each other (Table 1). AVb and ATM were not significantly different from each other (Table 1).

DISCUSSION

Because the habitat type classification system is so widely known in Wisconsin and the data have already been collected on a considerable percentage of the land base, using HTCS to indirectly determine other parameters would be beneficial. Bakken and Cook (1998) demonstrated that despite extremely large background variance in numbers of both large and small seedlings, HTCS could be broadly used to predict dominant species in the regeneration. Schwartz (2012) showed that the presence of oak advance regeneration was influenced by habitat type, with AAAt having the least and AVDe, AVb, and ATM grouping at a moderate level of advance regeneration. The ability to use this existing site information to predict other parameters such as annual basal area or volume increment would be valuable.

Generally, annual volume and basal area increment were quite well predicted by HTCS. AAAt and AVDe seemed to group as lower productivity sites and AVb and ATM seemed to group as higher productivity sites (although this result was clear for annual basal area increment, it was somewhat more complicated for annual volume increment). Kotar et al. (2002) listed ATM as moister than the other three habitat types (Table 2), so its grouping as more productive than AAAt and AVDe is not unexpected. The relationship of growth rate to habitat type seems logical. Fassnacht and Gower (1998) showed annual net primary productivity (ANPP) to be strongly related to habitat type. Their mean ANPP by habitat type tended to increase as soil moisture regime became more mesic and as soil nutrient regime became richer. Additionally, the higher variability (greater confidence intervals) of AVb could in part be due to the variable nutrient regimes (from medium to rich) that are listed as characteristic for that habitat type. Whereas the means for the other habitat types had relatively narrow confidence intervals (most likely adequate for predicting growth rates relative to management activities), AVb was significantly more variable with 90-percent confidence intervals of annual basal

area increment of 0.023 to 0.037 square feet per tree compared to 0.017 to 0.021, 0.016 to 0.024, and 0.028 to 0.034 square feet per tree for AAt, AVDe, and ATM, respectively. It is quite possible that habitat type may be adequate, by itself, to predict annual basal area and volume increment on some habitat types but that for sites with other habitat types (in this case AVb), other parameters may be necessary to get an adequate prediction. What those parameters might be is only speculation with our current data set.

Overall, habitat type appears to have potential use in the prediction of annual basal area increment and volume growth for individual dominant and codominant trees on some habitat types. Refining this information for broader use seems warranted based on this exploratory project.

ACKNOWLEDGMENTS

This project was funded as part of a McIntire Stennis Cooperative Research Grant. Thanks to the many DNR, county, and private forest managers and private landowners who provided site access for this project. Thanks to Rob Michitsch and Ron Zimmerman. Thanks to Sophie Demchik for assistance with the paper.

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FOREST DISTURBANCE

STAND DYNAMICS FOLLOWING GAP-SCALE EXOGENOUS DISTURBANCE IN A SINGLE COHORT MIXED SPECIES STAND IN MORGAN COUNTY, TENNESSEE

Brian S. Hughett and Wayne K. Clatterbuck¹

Abstract.—Differences in composition, structure, and growth under canopy gaps created by the mortality of a single stem were analyzed using analysis of variance under two scenarios, with stem removed or with stem left as a standing snag. There were no significant differences in composition and structure of large diameter residual stems within upper canopy strata. Some preexisting advance regeneration was recruited as a new cohort following the disturbance. On cut plots, the recruitment consisted of eastern white pine (*Pinus strobus*), yellow-poplar (*Liriodendron tulipifera*), and red maple (*Acer rubrum*). On no cut plots, the advance regeneration recruited as a new cohort was comprised of red maple, American beech (*Fagus grandifolia*), and a few oaks (*Quercus* spp.). The removal of the gap maker provided a pathway to recruit suppressed stems into larger diameter and crown classes. Conversely, plots where the gap maker was left as a standing snag tended to result in larger radial increases by the closest major competitors.

INTRODUCTION

Forests of southern Appalachia are subject to both anthropogenic and nonanthropogenic disturbance. Disturbances affect the composition, structure, and future development of forests at varying scales and frequencies. Differences in disturbance types and characteristics can influence species composition, age, geographic location, time since previous disturbance, and developmental stage of the forest (Oliver and Larson 1996). The ability of trees to respond following a gap-scale disturbance depends on species characteristics, age, and the gap environment (Wilder et. al. 1999). Gap-scale disturbances are characterized by small openings within the canopy that occur due to the loss of one or a few trees. Gap-scale disturbances can affect species composition and stand structure by facilitating the establishment of new germinates, recruiting subcanopy trees to larger size classes, and lateral crown expansion of overstory stems (Hart and Grissino-Mayer 2009, Hart et. al. 2010, Wilder et. al. 1999).

Canopy gaps formed in secondary growth forests of the eastern United States are typically very small because of the smaller tree size and shorter distances between trees of forests in the precomplex stage of development (Hart and Grissino-Mayer 2009, Hart et. al. 2010, Wilder et. al. 1999). These small canopy gaps usually close by lateral crown expansion of adjacent overstory stems rather than height growth of understory trees. However, even small canopy gaps can alter stand composition and structure by establishing new sources of regeneration and recruiting understory stems into larger size classes (Hart and Grissino-Mayer 2009, Hart et. al. 2010, Runkle 1981). Successive gap-scale disturbances may allow understory trees to reach the canopy (Hart and Grissino-Mayer 2009, Hart et. al. 2010). Light availability on the forest floor can be two or more times greater under single tree canopy gaps compared to closed canopy conditions (Krasny and Whitmore 1992). Numerous studies

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have demonstrated that this light increase is sufficient enough to release only shade-tolerant species (Hart and Grissino-Mayer 2009, Hart and Kupfer 2011, Hart et. al. 2010, Hix and Helfrich 2003). However, some studies have suggested that species with intermediate shade tolerance will respond similarly to shade-tolerant species (Canham 1989, Naidu and DeLucia 1998).

In mixed species stands, disturbances caused by senescence, forest pests, and pathogens typically result in the mortality of single stems or single species dispersed throughout the stand (Hart and Kupfer 2011, Krasny and Whitmore 1992). These types of disturbances cause the gradual formation of canopy gaps over a period of years rather than suddenly (Krasny and Whitmore 1992). Researchers have hypothesized that forest response to gradual canopy gaps differs from sudden canopy gaps. Gradually formed canopy gaps that retain standing snags for a number of years are believed to differ from suddenly formed canopy gaps in the following manners: gaps with standing snags are smaller (Hart and Grissino-Mayer 2009, Hart and Kupfer 2011, Hart et. al. 2010, Krasny and Whitmore 1992); light and possibly belowground resources become available more gradually (Krasny and Whitmore 1992); gradual tree death is less destructive to advanced regeneration (Krasny and Whitmore 1992); and the bole and branches of standing dead trees may inhibit sunlight availability to the understory (Hart and Kupfer 2011, Krasny and Whitmore 1992). Despite numerous studies focusing on forest gap dynamics in mixed species stands, no research has directly compared the changes in species composition and growth response under canopy gaps formed by the mortality of a single stem where the stem was either cut or left as a standing snag.

JUSTIFICATION AND OBJECTIVES

Southeastern mixed species stands vary greatly in terms of composition, structure, and disturbance. Limited research has been conducted on the successional processes occurring after small scale exogenous disturbance within upland mixed species forests of the Southeast. The goal of this study was to investigate stand dynamics following gap-scale exogenous disturbance. Specifically, the objectives were to determine differences in forest composition, structure, and growth response under canopy gaps created by the mortality of a single stem where the stem was either removed or left as a standing snag. We hypothesized that removing the stems responsible for the creation of canopy gaps would result in greater growth by intermediate and suppressed stems and increased variation of forest composition and structure.

STUDY SITE

The study was conducted in the Cumberland Forest field research unit of the University of Tennessee Forest Resources Research and Education Center located in Morgan County, Tennessee. This area is located within the Cumberland Mountain physiographic region, subregion two, land type association G, type 24 and 25, which is the Wartburg Basin and Jellico Mountains (Smalley 1984). Regionally, the topography is characterized by elevations of 1,200 to 3,000 feet above sea level, steep slopes of 20 to 60 percent, narrow crests, and narrow, winding valleys (Smalley 1984). Within the study site, elevation ranged from 1,200 to 1,500 feet above sea level with slopes of 5 to 45 percent. Regional soils are deep sandy-silt loams derived from weathered colluvial sandstone and shale. They are described as acidic, well to excessively drained, and are of moderate to moderately low productivity (Smalley 1984). Soils within the study area were predominantly of the Gilpin-Petros complex or the Lonewood series, which reflects the regional description.

The climate is classified as humid mesothermal, with long, moderately hot summers and short, moderately cold winters. The mean annual temperature is 55 °F (Thorntwaite 1948). The frost free period is typically 180 to 190 days, with the first freeze in mid to late October and the last freeze occurring in mid-April. The region receives an average annual precipitation of 49 inches, which is usually well distributed throughout the year. However, the region is prone to short periods of intense precipitation or drought (Smalley 1984).

Braun (1950) included the Wartburg Basin as part of the Cumberland Mountain region and classified it as the “outlying area” of the Mixed Mesophytic Forest region. True mesophytic species dominate only protected lowland areas. Within the Wartburg Basin, upland pine and pine-oak communities are prevalent (Braun 1950). Despite the patchy old-growth, mixed mesophytic forest remaining today, much of the region’s forest structure and composition has changed (Hinkle 1989). Regionally, disturbance caused by detrimental logging practices (1800 to present), coal mining (1915 to present), wildfire, forest pests, and pathogens, have varied the community types located within the Wartburg Basin (Deselm et al. 1978).

The study area was heavily cut over in the years from 1915 to 1937 prior to being deeded to the University of Tennessee in 1937. Between 1998 and 2002, southern pine beetle (*Dendroctonus frontalis* Zimmerman) populations reached epidemic proportions, affecting forests throughout the southeastern United States. Prior to the pine beetle epidemic, eastern white pine (*Pinus strobus*) accounted for an average of 18 percent of the total basal area of overstory trees on our sites. The southern pine beetle outbreak resulted in the mortality of nearly all overstory eastern white pine within the study area. In 2003, the management activities conducted on our site involved the salvage cutting of the easily accessible overstory eastern white pine stems that the southern pine beetles killed. The salvage cutting was limited to eastern white pine stems that were easily accessible, thereby minimizing the damage to residual vegetation. Due to the low basal area and relatively even distribution of eastern white pine on our site, many small canopy gaps were formed.

METHODS

Data Collection

Ten 0.2-acre research plots were established in 2010. Plots were restricted to canopy gaps created by the mortality of single stems, which, prior to the pine beetle outbreak, were in dominant or codominant positions. Plots were separated into two treatment categories according to whether the pine was harvested in 2003 (salvage cut) or left as a standing snag (not cut). Five 0.2-acre plots were sampled for each treatment. For each 0.2-acre plot of a given treatment, a 0.2-acre plot of the opposite treatment was established on similar site conditions, e.g., aspect, landscape position, slope, and concavity (Table 1).

Table 1.—Description of site conditions for paired 0.2-acre plots

Site type	Aspect	Landscape position	Slope (%)	Concavity
1	west	mid slope	22-26	convex
2	west	ridge	5-9	concave
3	southwest	ridge	12	convex
4	southwest	low slope	18-20	concave
5	south	mid slope	24-27	concave

In each 0.2-acre plot, species, diameter at breast height (d.b.h.) and crown class were recorded for all stems with a d.b.h. ≥ 5 inches. Annual radial growth is directly proportional to annual height growth (Hart et. al. 2010, Kariuki 2002). To evaluate annual diameter growth from 1998 to 2010, three trees on each plot were cored at breast height with an increment borer. Thirteen measurements of annual radial increase, one for each year from 1998 to 2010, were taken on each tree core. Each cored tree fell into one of three competitor classes: the closest major competitor (CMC) to the dead or removed eastern white pine, a tree other than the CMC within a dominant or codominant crown class, or a tree within the intermediate or suppressed crown class. Cored trees in each competitor class across all plots had a similar shade tolerance and age. For example, each CMC tree cored had an intermediate shade tolerance and was approximately 98 years of age. The “Silvics of North America” by Burns and Honkala (1990a, 1990b) describes five shade tolerance classes: very intolerant, intolerant, intermediate, tolerant, and very tolerant. This study used only three shade tolerance classes: intolerant, which includes very intolerant and intolerant species; intermediate, including only intermediate species; and tolerant, which is composed of very tolerant and tolerant species.

Within each plot, two 0.001-acre regeneration plots were established at a distance of 15 feet from plot center at azimuths of 0 degrees and 180 degrees. Density, cumulative height, and shade tolerance of advance regeneration were recorded for each species. Cumulative height was defined as the total height of all the trees of a certain species or species group per unit area.

Statistical Methods

Basal area, density, and species richness were determined for all trees with a d.b.h. ≥ 5 inches on the five 0.2-acre plots for each treatment. A randomized complete block design was used, blocking on site (Table 1). Analysis of variance was run, and means separation with the Tukey method of experimentwise error control was used to test for differences between treatments for all plots and by canopy class for each treatment. SAS Version 9.2 was used for all statistical analyses (SAS Institute Inc., Cary, NC).

Density and cumulative height were calculated for all sources of advanced regeneration on 20 of the 0.001-acre subplots. A randomized complete block design with sampling was used, blocking on site (Table 1). Analysis of variance was run and mean separation with the Tukey method of experimentwise error control was used to test for differences between treatments for all plots and by shade tolerance for each treatment.

Annual radial increase, as a measure of growth response, was measured on three tree cores (one for each competitor class) from five 0.2-acre plots for each treatment for 13 time periods between 1998 and 2010. A randomized complete block design with repeated measures was used, blocking on site (Table 1). Analysis of variance was run, and mean separation with the Tukey method of experimentwise error control was used to test for differences between treatments, time, and the interaction effect of treatment x time by each competitor class. A conventional type one error rate of five percent was chosen for all tests of statistical difference. However, trends ($0.05 \leq P \leq 0.1$) were also reported for tests utilizing repeated measures treatment design. Trends were reported because the Tukey method of experimentwise error control and repeated measures treatment design resulted in an unacceptable level of statistical power when testing at the 0.05 alpha level. Testing at a type I error rate of 10 percent raised the power of these tests to an acceptable level of greater than 78 percent for all whole plot main effects. Power analysis was conducted using Proc power.

RESULTS

Significantly more trees occupied a dominant crown position on plots where the gap maker was left as a standing snag, but no significant differences were found between treatments for basal area, density, and species richness (Table 2). White oak (*Quercus alba*), composing 30 percent of the total basal area on each type, was the species of greatest dominance for both the cut and no cut treatment types. White oak and red maple (*Acer rubrum*) were present in greatest densities on both treatment types. However, red maple made up a larger proportion of the basal area on the no cut treatment plots. Cut plots were mostly dominated by three species, white oak, northern red oak (*Quercus rubra*), and scarlet oak (*Quercus coccinea*), with oaks (*Quercus* spp.) constituting over 75 percent of the basal area. The basal area of the no cut treatment plots was more dispersed across species and included white oak, scarlet oak, red maple, yellow-poplar (*Liriodendron tulipifera*), northern red oak, and mockernut hickory (*Carya tomentosa*) (Table 3).

Table 2.—Means and standard errors for diversity, structural, and compositional measures on plots where the gap maker was salvage cut (Cut) and plots where the gap maker was left as a standing snag (No cut)

Parameter	Cut	No Cut
Basal area (ft ² /acre)	61.24 ± 5.89 a	70.18 ± 5.89 a
Density (trees/acre)	49.00 ± 3.24 a	55.00 ± 3.24 a
Richness (# of species)	5.20 ± 0.45 a	5.60 ± 0.45 a
Dominant (trees/acre)	10.00 ± 1.32 b	16.00 ± 1.32 a
Codominant (trees/acre)	19.00 ± 3.50 a	21.00 ± 3.50 a
Intermediate (trees/acre)	6.00 ± 2.69 a	11.00 ± 2.69 a
Suppressed (trees/acre)	14.00 ± 3.16 a	7.00 ± 3.16 a

Means within a row that are followed by the same letter are not significantly different ($p < 0.05$).

Table 3.—Average basal area and average density by species for plots where the gap maker was salvage cut (Cut) and plots where the gap maker was left as a standing snag (No Cut)

Species	Basal area (ft ² /acre)		Density (trees/acre)	
	Cut	No cut	Cut	No cut
<i>Fagus grandifolia</i>	.	1.8	.	4
<i>Quercus prinus</i>	2.1	5.5	1	3
<i>Carya tomentosa</i>	1.2	8.3	1	5
<i>Quercus rubra</i>	25.4	8.4	9	4
<i>Acer rubrum</i>	5.4	15.0	12	24
<i>Quercus coccinea</i>	8.5	17.0	3	9
<i>Quercus falcata</i>	3.3		1	.
<i>Liquidambar styraciflua</i>	.	1.6	.	1
<i>Oxydendrum arboretum</i>	0.5	.	1	.
<i>Quercus alba</i>	28.2	27.4	20	16
<i>Liriodendron tulipifera</i>	2.3	8.7	4	7
<i>Pinus strobus</i>	4.8	.	12	.
Sum	81.7	93.6	65	73

Cut plots had a reverse J-shaped diameter distribution curve, which is representative of a single cohort stratified mixture (Fig. 1). No cut plots had a bi-modal diameter distribution, which is indicative of a stand recovering from disturbance (Fig. 1). Vertical structure differed in composition between treatment types in all crown classes except the dominant crown class which was dominated by oaks for both treatment types (Fig. 2). The codominant crown class of cut plots was almost entirely dominated by oaks while the same crown class of no cut plots, including both shade-intolerant and shade-tolerant species, was more heterogeneous (Fig. 2). The intermediate and suppressed crown classes of the no cut plots were dominated almost entirely by shade-tolerant species (Fig. 2).

Fourteen species of advance regeneration were found on the 20 subplots taken across the two treatment types. There were no significant differences in density between the two treatments (Table 4). The species with the highest densities on plots of each treatment type were red maple, mockernut hickory, white oak, northern red oak, black cherry (*Prunus serotina*), sourwood (*Oxydendrum arboretum*), eastern white pine, and yellow-poplar. Plots of the cut treatment supported a significantly higher cumulative height of advance regeneration (Table 4). Cumulative height of shade-tolerant species of advanced regeneration was significantly larger on cut plots (Table 5).

Analysis of variance revealed no significant differences in growth response between the main effects of treatment and time; these factors did not interact. However, a trend was evident for the main effect of treatment for both the CMC and suppressed tree competitor classes. Radial increase of the CMC competitor class tended to be higher on plots where the gap maker was left as a standing snag (Fig. 3). Radial increase of the suppressed tree competitor class tended to be higher on plots that were salvage cut (Fig. 3).

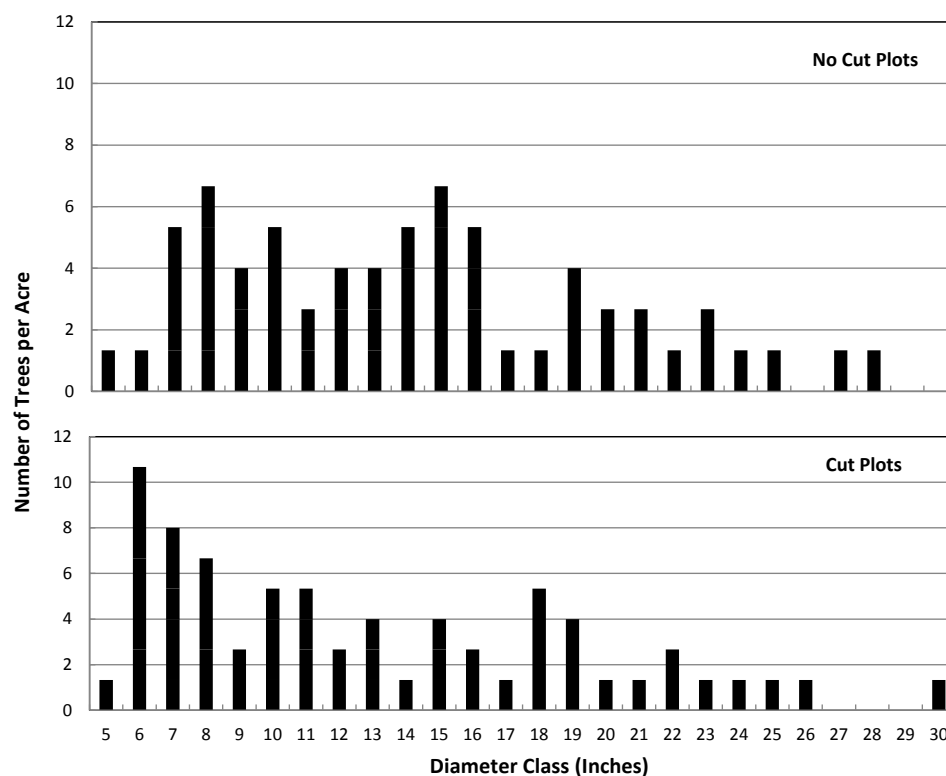


Figure 1.—Number of trees (with d.b.h. ≥ 5 inches) per acre by 1-inch diameter classes for all species on both cut and no cut plots.

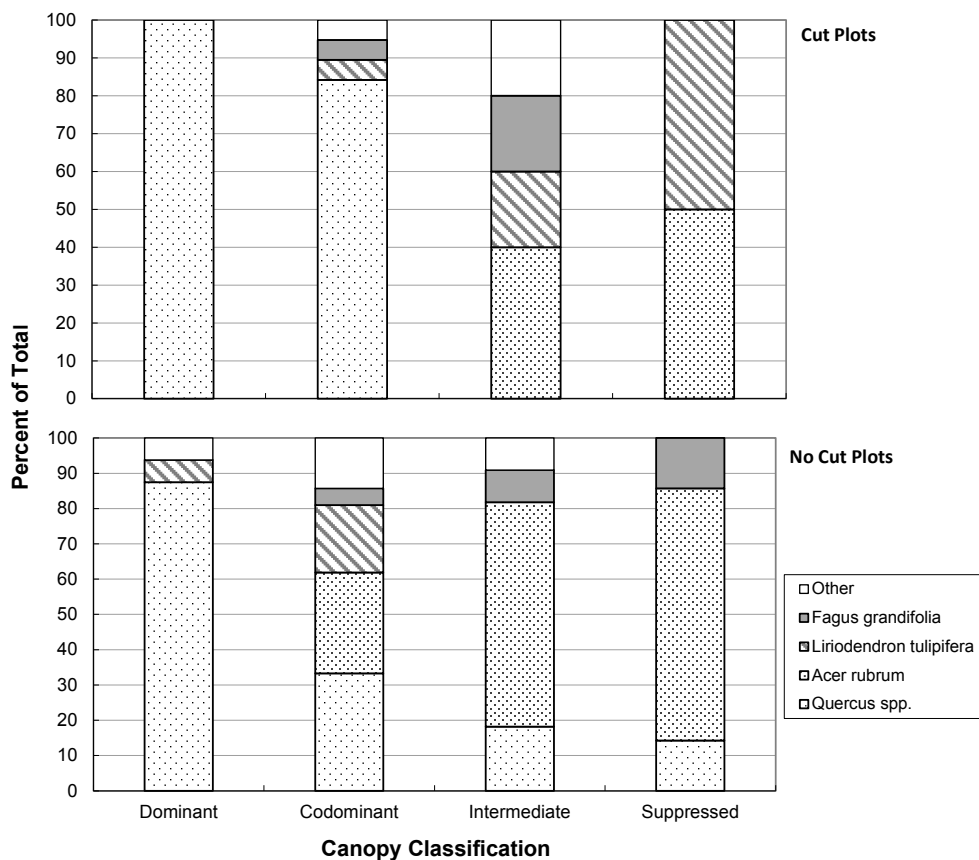


Figure 2.—Canopy class distributions for plots of each treatment. Categories are based on the amount and direction of intercepted light (Oliver and Larson 1996).

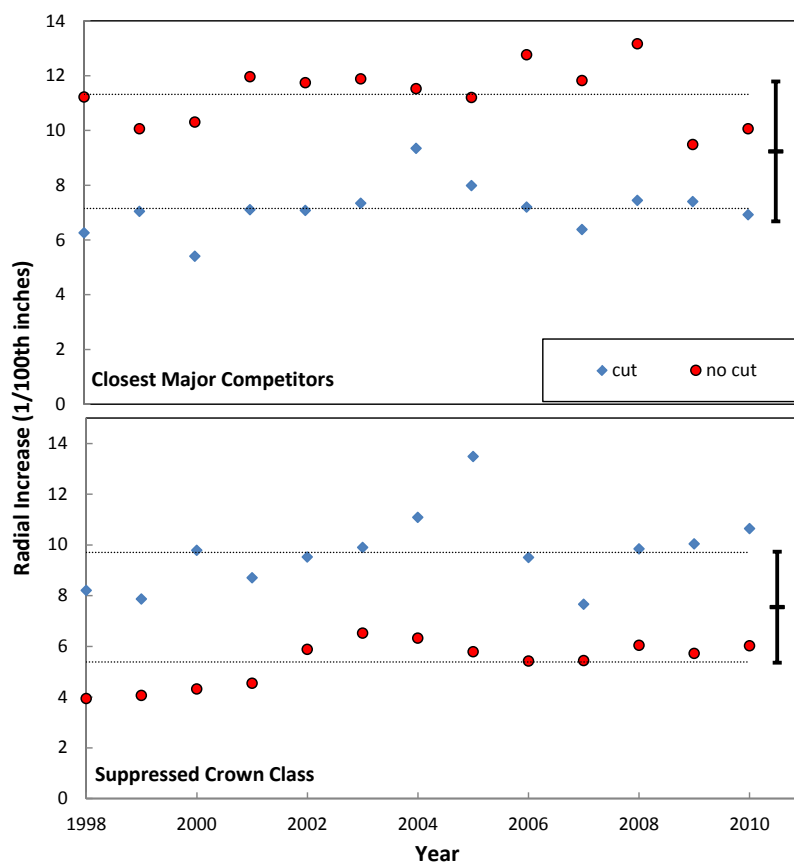


Figure 3.—Mean values of radial increase for each year of measure by treatment type (interaction effect). Dashed lines indicate mean radial increase for each treatment (main effect). Error bars represent Tukey mean separation values used to test for significant ($p < 0.05$) differences between treatments (main effect) for each competitor class.

Table 4.—Means and standard errors for measures of advanced regeneration on plots where the gap maker was salvage cut (Cut) and plots where the gap maker was left as a standing snag (No cut)

Parameter	Plots	
	Cut	No cut
Density (stems/acre)	24,600 ± 2505 a	18,700 ± 2505 a
Cumulative height (ft/acre)	52,700 ± 6,290 a	25,850 ± 6,290 b

Means within a row that are followed by the same letter are not significantly different ($p < 0.05$).

Table 5.—Means and standard errors for measures of advanced regeneration by shade tolerance on plots where the gap maker was salvage cut (Cut) and plots where the gap maker was left as a standing snag (No cut)

Parameter	Shade tolerant		Intermediate tolerance		Shade intolerant	
	Cut	No cut	Cut	No cut	Cut	No cut
Density (stems/acre)	10,200 ± 1,885 a	6300 ± 1885 a	8,500 ± 2,012 a	7,500 ± 2,012 a	5,900 ± 1,639 a	4,900 ± 1,639 a
Cumulative height (ft/acre)	16,900 ± 3,191 a	9450 ± 3191 b	19,050 ± 4,860 a	10,850 ± 4,860 a	16,750 ± 4,915 a	5,550 ± 4,915 a

Means within a row that are followed by the same letter are not significantly different ($p < 0.05$).

DISCUSSION

Previous research has shown that high basal area of hardwoods relative to that of pine species limits the spread of southern pine beetle (Schowalter and Turchin 1993), but stands with low pine densities can become infested if they are overstocked (Lorio 1980). The mortality of the pine component, which comprised roughly 18 percent of the total basal area prior to the southern pine beetle disturbance, altered the stand structure and composition. This result corroborates the findings of similar studies which demonstrated that small canopy gaps within secondary forests can influence stand structure and successional pathways (Hart and Grissino-Mayer 2009, Hart and Kupfer 2011, Hart et. al. 2010). The southern pine beetle disturbance transformed the stand from a fully to overstocked, single cohort, mixed species stand in the understory reinitiation stage of stand development into a moderately full to fully stocked, two cohort, mixed species stand in the understory reinitiation stage of development.

Despite changes in basal area and density resulting from the loss of the overstory pine, the southern pine beetle disturbance had no effect on the species composition of trees within the larger diameter classes and upper crown classes. However, small diameter, lower canopy stratum trees were released from competition. The changes caused by small canopy disturbances are often most prevalent in lower canopy stratum and the regeneration layer (Hart and Kupfer 2011). Response by stems of the lower canopy strata represent the pool of species that can be recruited to larger size classes and potentially the canopy (Hart and Kupfer 2011, Wilder et. al. 1999). Some preexisting advance regeneration was recruited as a new cohort following the disturbance. On cut plots, the recruitment consisted of eastern white pine, yellow-poplar, and red maple. On no cut plots, the advance regeneration recruited as a new cohort was comprised of red maple, American beech (*Fagus grandifolia*), and a few oaks. Unlike the results of Hart and Kupfer (2010), oak saplings were released under the same canopy gaps as red maple and American beech saplings. In their study, the gaps under which oaks were released existed on very dry sites where red maple and American beech are not competitive. None of the plots in our study were severely moisture deficient. We speculate that

no oaks were released under gaps where the gap maker was removed because oaks were quickly outcompeted by the numerous shade-intolerant stems that responded to the cut treatment. For the most part, shade-intolerant species were recruited on cut plots. Canopy gaps created suddenly, as is the case with harvesting, are typically larger and transmit more light to the understory than gaps that retain a standing snag (Hart and Kupfer 2011, Krasny and Whitmore 1992). The density and growth response of shade-intolerant species under canopy gaps is proportional to the intensity of the disturbance (Canham 1989, Hart et. al. 2010, Hart and Kupfer 2011, Hix and Helfrich 2003). In contrast to the cut plots, mostly shade-tolerant species responded on plots where the gap maker was left as a standing snag. We speculate that prior to the disturbance, shade-tolerant saplings existed throughout the stand because of closed canopy conditions and were released under both treatments (Hart et. al. 2010, Hart and Kupfer 2011, Hix and Helfrich 2003).

Predictably, plots from the cut treatment supported greater densities and significantly larger cumulative height of regeneration. The differences in density and dominance can be attributed to the particular growth habits of each species. The significantly larger cumulative height and high density of shade-tolerant species of advance regeneration on cut plots indicate light increases to the forest floor that are favorable to the establishment of high numbers of relatively small shade-tolerant species. However, although not significantly different, the magnitude of difference in cumulative height between cut and uncut treatments for shade-intolerant and intermediate species exceeds that of shade-tolerant species. This result along with the relatively low densities indicates that the light increase to the forest floor on cut plots is also favorable to the establishment of few relatively tall stems of shade-intolerant and intermediately tolerant species of regeneration. The lack of statistical difference between cut and no cut plots can likely be attributed to the high level of variability between plots resulting in large standard errors for both shade-intolerant and intermediate species. Without continued disturbance, it is likely that these sources of regeneration will only persist as seedlings in the understory (Hart and Kupfer 2011).

Trends in radial increase indicated that smaller diameter trees within the intermediate or suppressed crown classes responded more vigorously than did other competitor classes when the gap maker was removed by salvage cutting. This response led to the recruitment of smaller diameter trees into successively greater diameter and crown classes. These results indicate that disturbances that do not leave a standing snag are more likely to foster crown closure as a result of vertical height growth than disturbances that do retain a standing snag. The removal of the gap maker may provide a pathway to recruit suppressed stems into the canopy; this is especially likely should future canopy disturbances occur (Hart and Kupfer 2011). Conversely, plots where the gap maker was left as a standing snag tended to result in larger radial increases by the closest major competitors. Standing snags in this study will eventually fall, likely as the result of a wind event (Hart and Kupfer 2011). How this delayed second disturbance event will affect future composition and structure is unknown.

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The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

INITIAL OBSERVATIONS ON TREE MORTALITY FOLLOWING A SEVERE DROUGHT IN 2012 IN TWO INDIANA STATE FORESTS AND IMPLICATIONS FOR LONG-TERM COMPOSITIONAL DYNAMICS

Andrew R. Meier and Mike R. Saunders¹

ABSTRACT

Introduction

Compositional and structural changes in response to silvicultural treatments in forest stands are well documented (e.g., Saunders and Wagner 2008), but the stochastic nature of natural disturbance events often precludes direct observation of their impacts on stand dynamics. Though the current dominance of oak-hickory forest types in the Central Hardwoods Forest region (CHFR) has been largely attributed to anthropogenic disturbance, some have postulated that periodic severe droughts may also have limited the dominance of more vigorous competitors. Morrissey et al. (2008), for example, found in a study of 70 clearcut stands on highly productive sites in the Hoosier National Forest that oak (*Quercus* spp.) persisted, and occasionally increased in dominance, through the stem exclusion phase of stand development; they attributed this to differential mortality among oaks and more site-sensitive yellow-poplar (*Liriodendron tulipifera* L.), particularly on more exposed slope positions and xeric sites (Beck 1990). Several other researchers (e.g., Hilt 1985, Parker and Swank 1982) have made similar observations of yellow-poplar's relative susceptibility to drought events.

Methods

We used a subset of forest inventory data collected as part of the hardwood ecosystem experiment in southern Indiana to assess the immediate impacts of a severe drought in 2012 on tree mortality. Initial sampling of the study area was conducted between 2008 and 2010 (hereafter Inventory 1). All standing living and dead stems were measured and categorized by condition and decay class (1-5) in 1/4-acre overstory (>4.5-inch diameter at breast height [d.b.h.]) and nested 1/20-acre sapling (2- to 5-inch d.b.h.) plots. Remeasurement of two-thirds of these plots was completed in 2013 (hereafter Inventory 2). This presentation is a preliminary report of a partial inventory conducted one year following the drought of 2012.

Proportions of dead trees were summarized by species, size class (sawtimber [>12-inch d.b.h.] or pole [2- to 12-inch d.b.h.]), and ecological land type (ELT, Van Kley and Parker 1993). A binomial proportions test (prop.test) in the R version 2.15.0 statistical software package (R Foundation for Statistical Computing, Vienna, Austria) was used to determine the significance of differences in the proportion of dead to living trees. Due to inconsistencies in data collection in the first inventory, we were unable to directly compare proportions for individual decay classes between Inventory 1 and Inventory 2.

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Results and Discussion

Overall, about 5.2 percent of stems were dead in Inventory 1, which is significantly lower ($p < 0.001$) than in Inventory 2 taken in 2013 when nearly 10 percent of stems were dead. The number of dead trees increased by 2.5 percent for sawtimber-sized trees and by 5.8 percent for poles between Inventories 1 and 2. All species considered in this analysis showed significantly higher proportions of dead trees in 2013, though the magnitude of the increase was greatest for yellow-poplar.

Detailed observations from Inventory 2 showed about 3.2 percent of all trees recorded had a decay class of 1, indicating recent mortality. This was substantially higher than background annual mortality rates (McWilliams et al. 1997). A smaller percentage of sawtimber-sized trees (2.1 percent) compared to poles (3.6 percent) was recorded as decay class 1. Mortality rates of all yellow-poplar stems (11.6 percent) were significantly higher ($p < 0.001$) than that of the main oak (*Quercus*) and hickory (*Carya*) species (2.4 percent). There was some variation by ELT in the number of dead and declining yellow-poplar trees, with mesic slopes (ELT 4) and dry slopes (ELT 2) having the highest proportion of newly dead trees (Figs. 1A and 1B). White oak (*Quercus alba* L.), the most widely distributed oak species on the sites, showed little variation in mortality across the different ELTs (Figs. 1C and 1D).

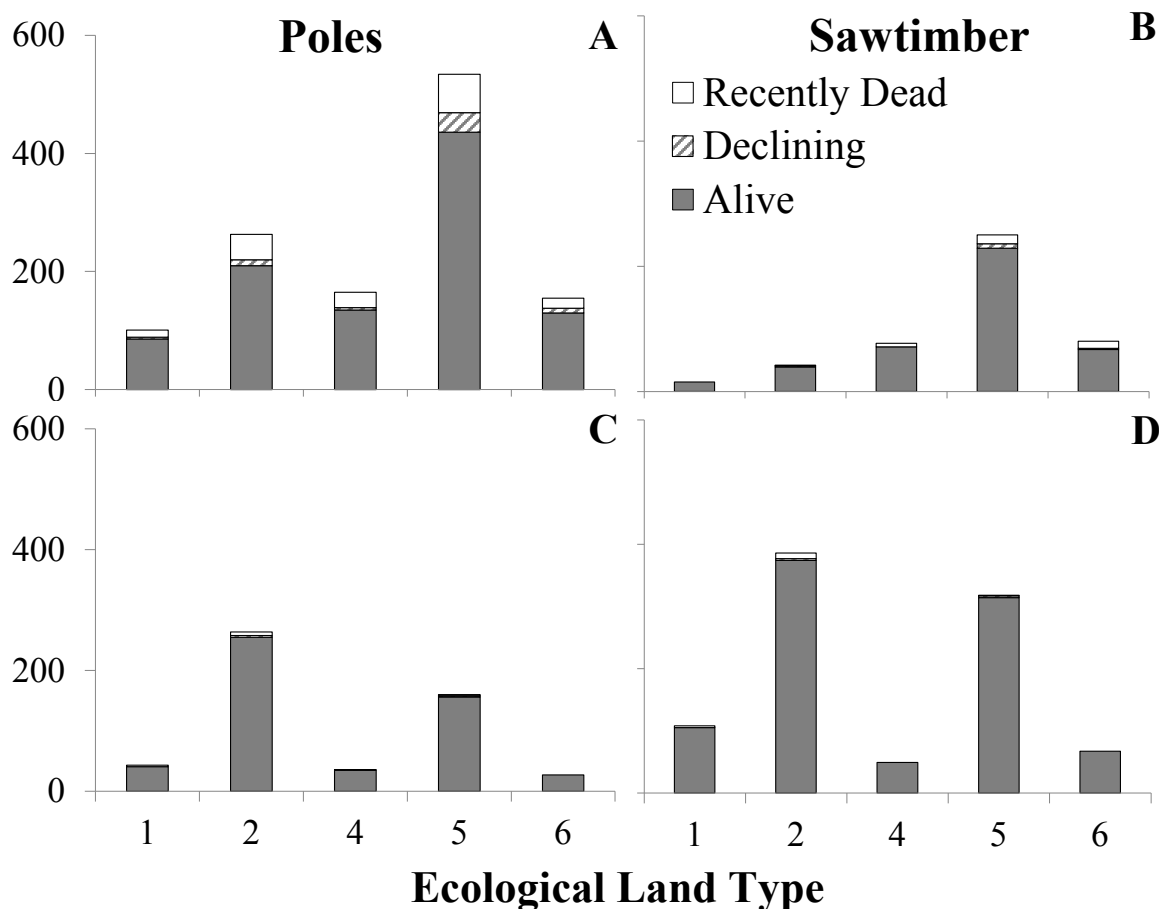


Figure 1.—Relative proportion of recently dead (decay class 1) and declining trees by size class for yellow-poplar (A, B) and white oak (C, D). Ecological land types (ELTs) are: dry ridges (1), dry slopes (2), mesic broad summits (4), mesic slopes (5), and bottomlands (6).

These preliminary results provided limited quantitative evidence that mortality rates for yellow-poplar immediately following a severe drought event were significantly higher than those for co-occurring oaks of the same size class, thus supporting the supposition that drought promotes shifts in species dominance in even-aged stands in the central hardwoods. Since drought-related mortality often occurs over a period of several years, we expect that overall mortality from the 2012 drought will be higher than the estimates in this analysis. It was evident from this data that pole-sized yellow-poplar stands were particularly susceptible to drought. However, if oak species are to capitalize on the growing space vacated by drought-killed yellow-poplar, it is imperative that oaks persist in a competitive state until a severe drought occurs. Given the growing evidence that noncompetitive oaks in pole-sized stands likely require release as early as 10 years of age (Zenner et al. 2012), it seems doubtful that managers can rely exclusively on stochastic drought events to maintain oak in even-aged central hardwood stands.

Acknowledgments

This paper is a contribution of the Hardwood Ecosystem Experiment, a partnership of the Indiana Department of Natural Resources, Purdue University, Ball State University, Indiana State University, Drake University, and The Nature Conservancy. Funding for the project was provided by the Indiana Division of Forestry. The authors thank the numerous technicians who collected, entered, and proofed the data.

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SUSCEPTIBILITY OF CENTRAL HARDWOOD TREES TO STEM BREAKAGE DUE TO ICE GLAZING

KaDonna C. Randolph¹

ABSTRACT

Introduction

During January 26-28, 2009, a winter storm dropped a mix of rain, ice, and snow from Texas across the Ohio River Valley and into New England. The storm caused multiple fatalities and millions of dollars of property damage and was called “the biggest natural disaster in modern Kentucky history” (Brammer and Funk 2009: 13). The storm disturbed an estimated 2.4 million acres of forest land across the central United States, including 2.1 million acres in Arkansas and Kentucky (Miles 2013). Ice accumulations up to 2.0 inches thick caused extensive damage to trees throughout the region.

Methods

Susceptibility to damage from ice storms, also known as glaze events, varies among tree species; however, there are some generally consistent trends (Hauer et al. 2006, Kraemer and Nyland 2010). For example, elm (*Ulmus* spp.) and black cherry (*Prunus serotina*) tend to be susceptible to damage, whereas hickory (*Carya* spp.) and black walnut (*Juglans nigra*) tend to be resistant. Maple (*Acer* spp.) and oak (*Quercus* spp.) are generally moderately susceptible to ice damage. The consistency of these trends in Arkansas and Kentucky following the January 2009 ice storm were examined with data collected between February 1, 2009, and January 31, 2012, by the U.S. Forest Service Forest Inventory and Analysis (FIA) program.

Each live tree with a diameter at breast height (d.b.h.) of at least 1.0 inch measured by the FIA program prior to the storm was matched with its post-storm assessment and designated as broken if after the storm the crown was broken and completely detached from the main stem. Broken and unbroken trees were grouped by the FIA disturbance variable² into two categories, disturbed only by ice in 2009 or undisturbed. Trees on undisturbed plots in counties with an ice-disturbed FIA plot were excluded. Hardwood species with at least 50 observations across all of the disturbed plots were included in the analysis. Species susceptibility to glazing was ranked according to the percentage of broken trees on the disturbed plots and by the odds ratio comparing the odds of a tree sustaining a broken top on a disturbed plot to the odds of a tree sustaining a broken top on an undisturbed plot. Trees were placed into categories of high, moderate, and low susceptibility based on the percentage of broken trees (greater than 20 percent, 10 to 20 percent, and less than 10 percent, respectively) and on the odds ratio (greater than 6.0, 3.0 to 6.0, and less than 3.0, respectively).

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² U.S. Department of Agriculture Forest Service. 2010. Forest inventory and analysis national core field guide: field data collection procedures for phase 2 plots. Version 5.0. Unpublished report. Available at <http://www.fia.fs.fed.us/fia/data-collection/>. (Accessed January 31, 2014).

Results and Discussion

Susceptibility rankings based on the percent broken criterion agreed with the susceptibility rankings based on the odds ratio criterion for 20 of 31 species. Susceptibility ranked high for black locust (*Robinia pseudoacacia*) and sassafras (*Sassafras albidum*) according to the percent broken criterion but low according to the odds ratio criterion. This was because both species sustained a high percentage of broken stems in the undisturbed conditions, suggesting that glazing only exacerbated the tendency of these species to sustain broken stems. Species such as bitternut hickory (*Carya cordiformis*), chinkapin oak (*Quercus muehlenbergii*), black walnut, and flowering dogwood (*Cornus florida*), that ranked relatively low in terms of percent broken on the undisturbed plots but relatively high on the disturbed plots, had high odds ratios suggesting that these species may be particularly susceptible to glazing.

Based on the odds ratio criterion, the susceptibility rankings for black cherry (high) and maple (moderate) agreed with general trends reported for previous storms; however the rankings for other species varied from previously reported rankings. Oak species ranked in all three susceptibility categories. Blackjack oak (*Quercus marilandica*), southern red oak (*Q. falcata*), and chinkapin oak ranked high. White oak (*Q. alba*), black oak (*Q. velutina*), and northern red oak (*Q. rubra*) ranked moderate, and post oak (*Q. stellata*) and chestnut oak (*Q. prinus*) ranked low. Bitternut hickory ranked high, whereas the other hickory species (*C. alba*, *C. glabra*, *C. ovata*, and *C. texana*) ranked moderate. Among the elms, American elm (*Ulmus americana*) ranked moderate and slippery (*U. rubra*) and winged (*U. alata*) elm ranked low.

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POSTER ABSTRACTS

EFFECTS OF FIRE AND FERTILIZATION ON GIANT CANE: DEVELOPING MANAGEMENT TOOLS FOR CANEBRAKE

Margaret M. Anderson, James J. Zaczek, Jon E. Schoonover, and Sara Baer¹

Giant cane (*Arundinaria gigantea* (Walt) Muhl.), a native bamboo, is an integral component of bottomland forests in the southeastern United States. Cane occurs as monodominant stands, also known as canebrakes, which historically covered vast areas of land. As a result of land conversion, overgrazing, and altered fire regimes, a 98 percent reduction of canebrakes has occurred. Interest in giant cane restoration has increased due to its ecological significance as wildlife habitat, a riparian buffer, its role in soil stabilization, and its potential as woody biomass. Research with planted cane indicates fertilization and burning have interacting effects on cane growth, however in remnant natural stands, the influence of burning and fertilization on canebrake growth and spread is unknown. This study examines the survival and growth response of cane to burning and fertilization in remnant stands to provide guidance for rehabilitation, restoration, and management.

Four treatment plots were replicated eight times across six sites in canebrakes growing in riparian zones adjacent to agricultural fields in the Cache River Watershed, Illinois. The four treatments were randomized factorial design of: (1) burning, (2) fertilization, (3) burning/fertilization, or (4) control. Within treatment plots, two interior and three exterior 1 m² sample plots were randomly established to measure culm density, height, diameter, and spread prior to treatment and after one and two growing seasons. Fertilization and burning/fertilization plots were treated in summers of 2011 and 2012 with a half corn rate of nitrogen (56 kg/ha), phosphorus (22 k/gha), and potassium (37 kg/ha). Prescribed burning took place in March 2012.

Data were analyzed using a repeated measures analysis ($\alpha = 0.05$) (SAS Institute Inc., Cary, NC). At year 0 (2011), culm density, height, and diameter were not significantly different among treatments. By year 2, live density in interior plots slightly increased, however density in exterior plots generally more than doubled, indicating canebrake expansion over time. Fertilization generally increased height but had little effect on cane diameter. Research suggests that cane typically increases in both height and diameter simultaneously, suggesting that fertilization only partially provides the resources needed to stimulate growth. Further analysis on fertilization application may be necessary to ascertain the efficiency of its role in culm growth and development.

Prescribed burning resulted in a decrease in height growth and diameter and consumed a portion of the existing culms. However, vigorous postfire resprouting of the canebrake resulted in an increase of culm density and appears to demonstrate the possible utility of fire as a tool for land managers to reduce competition and increase canebrake health and expansion.

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LAUREL WILT DISEASE AND FIRE: SHOULD THE CENTRAL HARDWOOD REGION TAKE NOTICE?

Cody D. Bailey and J. Morgan Varner¹

Invasions of nonnative insects and pathogens into forest ecosystems have historically generated large economic and ecological impacts. Nonindigenous insects are often more problematic than native insects because host tree species have not had the opportunity to evolve natural resistance. The rate of introductions into naïve ecosystems has coincided with a near global increase in the frequency of a behavioral shift from targeting dead to targeting live trees. A recent nonnative species introduction is the redbay ambrosia beetle (*Xyleborus glabratus*), which transmits a fungal pathogen *Raffaelea lauricola* responsible for laurel wilt disease (LWD). *R. lauricola* blocks the xylem of redbay (*Persea borbonia*) and many other taxa in the Lauraceae family including avocado (*Persea americana*), sassafras (*Sassafras albidum*), swamp bay (*Persea palustris*), silk bay (*Persea humilis*), pondspice (*Litsea aestivalis*), pondberry (also known as southern spicebush; *Lindera melissifolia*), northern spicebush (*Lindera benzoin*), camphortree (*Cinnamomum camphora*), and California bay laurel (*Umbellularia californica*), causing wilt symptoms that eventually lead to 90 to 100 percent mortality. Following mortality, dead redbay leaves are marcescent for 1 year or more; this phenomenon has been linked to increased crown ignition in several North American tree species. Accompanying leaf senescence, surface fire hazard may increase as the leaves, branches, and stems fall to the forest floor. Sassafras is capable of being the sole host of LWD in stands north of the originally predicted range of the redbay ambrosia beetle and may affect fuel loading in this area. The Central Hardwood Forest region may be subject to similar shifts in fire behavior as new nonnative pests continue to accumulate and alter the ecosystems at an ever-increasing rate.

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LANDSCAPE SCALE RESTORATION WITHIN THE CACHE RIVER JOINT VENTURE PARTNERSHIP

Jennifer A. Behnken, John W. Groninger, Erin L. Seekamp, and James J. Zaczek¹

Landscape-scale restoration measures in the Midwest typically require the formation of collaborative partnerships. The Cache River wetlands in southern Illinois, a designated Ramsar site², have been undergoing ecological restoration across multiple state, federal, and private ownerships. Although some aspects are coordinated under the Cache River Joint Venture Partnership (CRJVP), individual agencies and entities pursue specific ownership priorities and approaches to management. This case study explores the dynamics among land managers employed by federal agencies, state agencies, and nongovernmental organizations managing land and water resources in a hydrologically linked wetlands system within the CRJVP. Semi-structured interviews were conducted with 25 managers, including staff members who maintain active participation in restoration activities, and individuals who have worked closely with CRJVP. Triangulation of interview transcriptions, meeting observations, management plans, and other relevant agency/organization documents revealed emerging themes and patterns within the data. Grounded theory was applied to better understand how differences in institutional cultures, missions, and resources impact management practices across the landscape.

Results suggest that administrative processes, funding sources, policy and regulations, mission statements, specified objectives, and management goals within and between agencies and organizations determine how institutional priorities and capacity impact management decisions and on-the-ground implementation. Institutional structures influence decisionmaking power and grassroots capabilities. Incompletely defined management and decisionmaking criteria challenge compatibility among partners and the central mission of the CRJVP itself. There are implications of this research in the function of existing or future partnerships facing similar challenges.

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² Ramsar sites are wetlands of international importance, designated under the Ramsar Convention of 1971.

ALLOMETRIC EQUATIONS FOR ABOVEGROUND AND BELOWGROUND BIOMASS ALLOCATION OF AMERICAN CHESTNUT AND NORTHERN RED OAK REGENERATION

Ethan P. Belair and Mike R. Saunders¹

Prior to the importation of chestnut blight (*Cryphonectria parasitica* [Murr.] Barr) in the early 1900s, American chestnut (*Castanea dentata*) was a dominant species in many hardwood forest types in the eastern United States. Based on their co-occurrence in many habitats, it is often assumed that chestnut and oak species (*Quercus* spp.) are similarly adapted to resist abiotic stresses. Specifically, it is well known that oak's preferential allocation of resources to root systems increases their tolerance of both drought and fire. However, the similarity of chestnut's biomass allocation to co-occurring oak species has not been formally tested using the seedlings and saplings that may be deployed during the impending American chestnut restoration efforts. Furthermore, to the author's knowledge, belowground structures of chestnut seedlings in natural conditions remain unstudied. We harvested aboveground structures and excavated roots from 29 American chestnuts and 47 northern red oaks (*Quercus rubra*) with ground line diameters between 0.5 and 5 cm at three sites in north central Indiana. All individuals were divided into four component parts: stem, branches, foliage, and coarse roots (>2 mm). Additive biomass equations were developed using nonlinear seemingly unrelated regressions with ground line diameter, total height, live crown length, number of first order lateral branches, crown diameter, and overstory canopy openness as independent variables. Leaf area allocation and specific leaf area were investigated using digital scans of a subset of the foliage from each individual. This information is of interest as it elucidates chestnut's stress tolerance adaptations and likely performance on various sites, as well as its suitability for carbon sequestration.

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DENDROHYDROLOGICAL ANALYSIS OF MISSISSIPPI RIVER FLOOD EVENTS IN A MIXED BOTTOMLAND HARDWOOD FOREST

Margaret B. Bialecki and Matthew D. Therrell¹

Destructive flooding in 2011 highlights the dynamic nature of the Mississippi River, and the bottomland hardwood forest riparian ecosystem presents a unique opportunity to examine the history of the Mississippi River flood pulse and provide insight into the ecological effects of long-term hydrologic alterations in the river-floodplain system.

We collected tree ring samples in 2009 from 33 living and 2 dead oak (*Quercus* spp.) trees from Big Oak Tree State Park in Mississippi County, MO, to evaluate long-term yearly growth and response in floodplain trees. This site represents one of the few remaining stands of virgin wet-mesic bottomland hardwood forests within the lower Mississippi River (LMR) alluvial valley. We developed an annually resolved tree ring record of high magnitude flooding on the LMR based on anatomical signatures evident in the wood (flood rings). Flood ring years were determined by examining each tree series for evidence of flood injury and abnormalities consistent with previous flood ring studies. We found that the most pronounced characteristic of flood rings in the oaks sampled was a reduction in the cross-sectional area of the earlywood (EW) vessels during the year of inundation. Additional characteristics used for identification included narrow rings, irregular EW vessel distribution, reduced latewood fiber, and disorganized flame parenchyma.

The resulting flood ring record identified spring flood events on the LMR from 1694-2009 and included virtually all of the observed high magnitude spring floods of the 20th century occurring on the LMR adjacent to the Birds Point-New Madrid floodway, as well as similar flood events in prior centuries. A response index analysis for years 1770-2009 indicated that more than half of the floods identified caused anatomical injury to more than 50 percent of the sampled trees and many of the greatest flood events were recorded by 80 to 100 percent of the trees at the study site. A comparison of the response index with average daily river stage height values at New Madrid, MO (1879-2009) indicated that the flood ring record can explain significant portions of the variance in both stage height (30 percent) and number of days in flood (40 percent) during spring flood events. Preliminary analysis of EW vessel size revealed that median vessel diameter can also be a proxy for large-scale flooding. Vessel size appeared to be highly responsive to stream flow measures, in particular duration of spring flood events ($r^2=0.82$). The flood ring record also suggested that high-magnitude spring flooding is linked to regional climate variability.

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ANALYSIS AND IMPACT OF ENERGY RECOMMENDATIONS IN THE WOOD PRODUCTS INDUSTRY

Brian Bond, Henry Quesada-Pineda, and Janice K. Wiedenbeck¹

Increasing manufacturing costs have contributed significantly to the decline of the forest products industry in the United States; increasing costs limit the ability of manufacturers to flourish against global competitors. While forest products companies in the United States are continually improving their products, processes, finances, and business practices, many have not seized upon opportunities to reduce energy consumption. Cutting energy costs remains a way that many wood products firms can trim operating expenses while developing new business prospects and serving existing customers. The goal of this project is to increase the competitiveness of the wood products industry by providing current information about the most beneficial energy saving opportunities. Methods include (1) data mining the implemented energy saving recommendations from the Industrial Assessment Center (IAC) collection for U.S. manufacturers, and (2) identifying lean management practices or principles that can be used to not only decrease energy consumption but also to increase productivity. Energy reduction recommendations based on lean principles can lead to greater savings and relative shorter payback times than will other common energy reduction recommendations.

Cluster analysis and statistical techniques will be used to identify the best energy recommendations for wood products sectors (primary and secondary). Those recommendations that have been successfully implemented by the industry will be classified as technical, administrative, or process improvement by using clustering techniques. Energy saving recommendations will be compared by different implementation criteria such as cost savings, payback period, and capital cost across different North American industry classification codes (NAICS) within the wood products industry. An example of this data for the pallet and skid-manufacturing sector includes an average savings per implementation of an estimated \$60.40/employee or \$0.09/ft², with an average payback period per implementation of 0.84 year. The most common implemented energy recommendation for this industry sector was the elimination of leaks in inert gas and compressed air lines/valves with an average payback of 0.23 year and annual savings of \$19.10 per employee or \$0.025/ft².

Work completed to date demonstrates that recommendations based on lean principles can lead to greater savings and relatively shorter payback times than other common energy reduction recommendations. For example, preliminary data analysis indicates that for 8 out of 10 cases, the payback period is less than 1 year. Data also indicate that for lean-based recommendations, the savings, based on the area (size) of the plant, ranges from \$0.08/ft² to \$1.92/ft². The underlying strength of lean-based recommendations is that while energy consumption is decreased, productivity is simultaneously improved.

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MAPPING TEMPORAL CHANGE IN OAK-DOMINATED ECOSYSTEMS IN THE CHICAGO REGION

Matthew Casali and Robert T. Fahey¹

Oaks are a keystone species in northeastern Illinois, driving much of the biodiversity in the region. Oak ecosystems are in decline because of landscape-level changes such as alteration of disturbance regimes, habitat fragmentation, and urban development, as well as stand-level changes such as competition from shade-tolerant species, encroachment of invasive plant species, and lack of management or disturbance-related canopy openings—all of which have led to widespread reproductive failures. The goal of this project was to identify existing oak dominated communities throughout the seven northeastern Illinois counties that make up the Chicago metropolitan region: Cook, DuPage, Kane, Kendall, Lake, McHenry, and Will Counties. The current distribution of oak dominated ecosystems was compared spatially and temporally with records from multiple time periods dating back to the 1800s. The project provides spatially explicit information regarding the quantity and parcel size of remaining oak ecosystems, their ownership and conservation status, and relationship to drivers of change from original distribution.

Using ArcGIS software (Esri, Redlands, CA), oak dominated ecosystems were identified from a combination of presettlement vegetation information from 1800s public land survey notes, 1939 aerial photography, modern soils data, and orthoimagery from 2010. Using the 1939 aerial imagery, oak dominated parcels were located and digitized. Probable oak dominance for each parcel was evaluated based on species data from public land survey notes, landscape position, soil data, and user interpretation of the imagery based on tone, shape, size, texture, and association. These parcels were then compared to the 2010 orthoimagery and were reshaped or removed to include only the previously existing oak communities.

Our findings indicated a significant decline in the extent of oak ecosystems from presettlement baselines across the region, but also high spatial variability in this landscape transition and its drivers. Across the seven-county region, the estimated area of oak dominated ecosystems declined from 782,709 acres in the 1830s to 209,951 acres in 1939 (27 percent remaining), and 111,518 in 2010 (14 percent). The two highest levels of remnant oak ecosystems in 2010 were found in exurban Kendall County (27 percent) and highly urbanized Cook County (19 percent). The lowest levels were found in exurban McHenry County (10 percent) where oak ecosystems were especially dominant in the presettlement landscape. Conservation status also differed greatly across the region; the highest percentage of protected ecosystems was found in Cook County (82 percent) and the lowest in Kendall County (14 percent). Overall, there have been large declines in the number of remaining large oak parcels. In 1939, there were 11 parcels 500 acres or greater, 83 parcels 200 acres or greater, and 271 parcels 100 acres or greater. In 2010, there were 3 parcels 500 acres or greater (27 percent), 37 parcels 200 acres or greater (45 percent), and 119 parcels 100 acres or greater (44 percent).

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These findings indicate patterns of decline in the oak ecosystems of the Chicago metropolitan region. Further data analysis from this project will illustrate the effects of different urbanization patterns on landscape composition and will be essential in regional oak conservation efforts, including illustrating the importance of private lands and connections between existing large parcels.

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INTERACTIONS AMONG INSECT DEFOLIATION, INSECTICIDE TREATMENTS, AND GROWTH RATE IN AMERICAN AND BACKCROSS GENERATION BC₃F₃ CHESTNUTS

Ashley E. Case, Albert E. Mayfield III, Scott E. Schlarbaum, Stacy L. Clark,
and Arnold Saxton¹

The American chestnut, *Castanea dentata* (Marsh.) Borkh, was once one of the most useful and abundant canopy trees in eastern North American forests. During the last 200 years, the species has been decimated by two exotic pathogens. *Phytophthora cinnamomi* (Rands) (Oomycetes: Pythiaceae) causes ink disease on wet or poorly drained sites, and *Cryphonectria parasitica* (Murrill) Barr (Diaporthales: Cryphonectriaceae) causes chestnut blight throughout the species' former range. Chestnut blight has been responsible for reducing millions of trees to short-lived understory sprouts. Breeding programs have aimed to transfer blight resistance from Asian chestnut species to American chestnut via a backcross breeding approach.

Seedlings from the third intercross of a third backcross generation (BC₃F₃) were planted in a series of field tests in southern Appalachian national forests. In these tests, the Asiatic oak weevil, *Cyrtepidomus castaneus* (Roelofs) (Coleoptera: Curculionidae), was identified as a primary late-season defoliator of chestnut seedlings. To better understand the impact Asiatic oak weevil has on seedling growth, we initiated a study using 1-0 bareroot American and BC₃F₃ chestnut seedlings. The seedlings were planted around the perimeter of a 40-year-old northern red oak, *Quercus rubra* L., seed orchard in eastern Tennessee where previous insect surveys detected an abundance of Asiatic oak weevils. In addition to quantifying the impact of *C. castaneus* defoliation on seedling growth, the study evaluated the efficacy of various insecticide treatments (imidacloprid, acephate, dinotefuran², and a water control) on phytophagous insects.

Ten seedlings of American and BC₃F₃ chestnuts, respectively, were randomly chosen from a pool of the healthiest individuals for each of four treatments in the study. Visual-categorical method and a digital-numerical method were compared in assessments of defoliation once a month from August through October. Height and root-collar diameter were also measured throughout the growing season to assess the impact of defoliation on seedling growth. Temporal and spatial fluctuations in weevil emergence was observed using 31 conical wire mesh insect traps placed on top of existing leaf deposits about 40 meters apart on the inside of the perimeter of the study site and beneath the drip line of the orchard's oaks.

Preliminary results showed a trend of greater defoliation on American chestnut than on BC₃F₃ chestnut seedlings. It is important to note the defoliation patterns observed cannot be solely attributed to *C. castaneus*. Future research will enclose *C. castaneus* on chestnut seedlings to further

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² Imidacloprid (BayerCrop Science, Monheim, Germany); acephate (Sigma-Aldrich, St. Louis, MO); dinotefuran (Mitsui Chemicals America, Rye Brook, NY).

understand the impact the weevil may have on seedling growth. In general, higher defoliation ratings were produced using the digital-numerical assessment method than with the visual-categorical method. Both methods suggested that imidacloprid and dinotefuran were effective in protecting American chestnut from defoliation. Defoliation ratings were less conclusive on BC₃F₃ seedlings, although seedlings that received imidacloprid treatments had the highest defoliation rates for both digital and visual ratings by October. *Cyrtopistomus castaneus* emergence peaked in late July and varied spatially within the study area, suggesting possible insect microsite preferences. Results from this research will aid in understanding the relationship that defoliation has on growth with American and BC₃F₃ chestnut seedlings and may ultimately contribute to the successful restoration of blight resistant American chestnut to eastern forests.

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ASSESSMENT OF THE COMPOSITION AND STRUCTURE OF A SUSPECTED OLD-GROWTH, MESIC HARDWOOD FOREST IN THE SOUTHERN OZARKS

James Crawshaw¹

This study assesses the quality of a small tract of mature forest in the Boston Mountains of the Southern Ozarks that exhibits characteristics of virgin, old-growth forest, and species typical of mixed mesophytic forests. The study site is about 200 m by 40 m and follows a northeast facing slope in the southernmost branch of Boen Gulf, a hollow in the headwaters of the Buffalo River. We hope to determine if this stand of mesic forest is remnant old-growth forest by assessing the age structure, species composition, density, frequency, and basal area of trees within the approximately 0.8 ha study site. All stems greater than 10 cm d.b.h. were inventoried. Twenty-one tree species were identified in the site with no clear dominant species based on frequency and basal area. Cucumbertree (*Magnolia acuminata*) represented 19.6 percent of the basal area present, more than any other species, followed in order by American basswood (*Tilia americana*; 15.1 percent), blackgum (*Nyssa sylvatica*; 12.5 percent), sugar maple (*Acer saccharum*; 11.1 percent), American beech (*Fagus grandifolia*; 7.8 percent), and umbrella magnolia (*Magnolia tripetala*; 5.6 percent). The diameter distribution for all species followed a reverse-J pattern with only a few individuals greater than 70 cm diameter. More data are being collected on the age structure of the stand by taking core samples from all the inventoried trees greater than 10 cm d.b.h. This study will serve as a baseline for future research in forest ecology and succession in Boen Gulf.

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SURVEY OF NORTHERN AND CENTRAL WISCONSIN FORESTERS REGARDING OAK SILVICULTURE

Michael Demchik, Kristine F. Kurszewski, Kaitlin N. Johnson, and Kevin M. Schwartz¹

Difficulties in regenerating oak (*Quercus* spp.) have been encountered over much of the eastern United States. These difficulties have been partially explained by a number of factors that include small or limited advanced regeneration, deer browsing, altered disturbance regimes, and competing vegetation. We surveyed foresters working in Wisconsin to determine the successes and barriers that they have experienced with regenerating oak.

Forty-three foresters working in 49 counties of northern and central Wisconsin were the subjects of this survey. Our goal was to survey at least one forester that worked in each northern and central Wisconsin county (noting that some foresters work regularly across more than one county). Nonrespondent bias was relatively low. One person stated that he had no time, several forwarded us to someone else in their office, three had retired, and we were unable to reach six for various reasons. Overall, we heard from foresters working in all but three counties in the northern two-thirds of Wisconsin. The survey was completed between September 2011 and May 2012.

Lower quality oak (scrub oak) sites presented a smaller regeneration barrier. For foresters that managed scrub oak (21 of the foresters included this as part of the areas they managed), 52 percent said that overstory removal was adequate for regeneration while 19 percent had encountered some issues. Other techniques that were tried with success on low quality sites included shelterwood, patch clear cut, group selection, and scarification. In Wisconsin, coppice is a generally accepted practice on scrub oak sites. These sites are usually cut while young (45-70 years) as a pulp and low grade log harvest. Nineteen percent of the foresters surveyed still encountered issues with regeneration on scrub oak sites; these issues included poor acorn crop, hazelnut competition, history of site degradation, or too small size (area) of cuts that encouraged deer browsing.

Higher quality sites presented more of a barrier. Only seven foresters found high quality sites easy to regenerate. The primary methods used, according to their responses, were shelterwood with site preparation (burning, chemical, chaining, or the combination of chemical and chaining; 71 percent), overstory removal over existing regeneration (29 percent). Group selection also was mentioned. Three of these foresters mentioned that oak regeneration was easier now that the deer population was lower in their area. For the 24 foresters that experienced problems with the success of oak regeneration on high quality sites, 63 percent had tried some form of shelterwood, with 21 percent also scarifying or chemically releasing, and 13 percent mentioned having done supplemental planting in the shelterwoods. One forester had given up hope that oak can be successfully regenerated under current conditions. A few other techniques were mentioned: releasing oak poles, single tree selection, and group selection. Two of the barriers that were mentioned were deer browsing and competing vegetation. While questions relating to deer were not asked in the survey, almost half of those

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surveyed still mentioned deer problems. Competing vegetation was also mentioned as a possible barrier. The combination of these two factors (existing competing vegetation and selective deer browsing) can present a rather significant barrier for oak regeneration. From these results, it appears that in some counties, shelterwood with site preparation may be adequate. In other counties, there may either be a more complex combination of barriers to successful regeneration or the local foresters have not devised a successful combination of treatments yet.

Overall, on sites where oak has a strong competitive advantage (dry, nutrient-poor sites), oak regeneration is easier to achieve. On high quality sites, foresters encounter more problems. The likelihood of success appears to improve either by using existing advanced regeneration or using a shelterwood with some site preparation.

Acknowledgments

This project was funded as part of a McIntire Stennis Cooperative Research Grant. We thank all foresters that participated in the survey and the Internal Review Board at University of Wisconsin Stevens Point which approved the sampling protocols. Thanks also to Sophie Demchik for assistance.

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LEAF STRUCTURE AND PHYSIOLOGICAL ATTRIBUTES OF *AILANTHUS ALTISSIMA* IN COOL AND WARM TEMPERATE REGIONS

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and Kyoichi Otsuki¹

To understand key attributes associated with the successful establishment and invasion of *Ailanthus altissima* (hereafter referred to as *Ailanthus*), we examined its leaf structure and ecophysiological characteristics from trees grown on sites in two different climatic regions: cool (Glenville, WV, USA) and warm (Fukuoka, Japan) temperate regions. *Ailanthus* was introduced in both countries from China and is considered an invasive species that threatens natural forests. Although leaf size was the same in both sites, specific leaf area, an indicator of photosynthetic capacity, was found to be larger in trees located in Glenville ($297.2 \pm 23.7 \text{ cm}^2/\text{g}$) compared to those in Fukuoka ($237.0 \pm 33.5 \text{ cm}^2/\text{g}$). Relative water content (RWC) was lower in Glenville (63.6 ± 2.6 percent) than in Fukuoka (79.9 ± 2.0 percent). Low RWC may indicate the ability of the plants to sustain excessive water loss without desiccation ($\text{RWC} < 40$ percent). There were also leaf structural differences between the two sites; those from Glenville exhibited light-adapted leaf characteristics with shorter stomatal length ($22.03 \pm 0.46 \text{ mm}$) and higher stomatal density ($232 \pm 8 \text{ per mm}^2$) than in Fukuoka ($28.6 \pm 0.87 \text{ mm}$; $196 \pm 12 \text{ per mm}^2$, respectively). Trees from Fukuoka sustained higher stomatal conductance ($205.0 \pm 14.7 \text{ mmol per m}^2 \text{ per s}$) throughout the day compared to those trees in Glenville ($135.6 \pm 14.5 \text{ mmol per m}^2 \text{ per s}$). Result of chlorophyll fluorescence analysis showed that *Ailanthus* trees in Glenville had a higher mean F_v/F_m of 0.80 than in Fukuoka (0.78). A lower value of F_v/F_m (< 0.80) may indicate photoinhibition, which can result in a decline in photosynthetic capacity due to high light intensity. The leaf structure and ecophysiological parameters measured in this study revealed the key attributes of *Ailanthus* that are associated with its invasiveness. Although *Ailanthus* in Fukuoka may still be in its early stage of invasion, its successful establishment where it was originally planted and aggressive physiological characteristics showed its potential to continuously invade natural forest ecosystems of Japan.

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

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IMPORTANCE OF OAK FOREST ECOSYSTEMS TO WOOD WARBLERS IN THE WESTERN HEMISPHERE

Paul B. Hamel, David A. Buehler, David King, Claudia Macias-Caballero, Kathryn Purcell, Scott H. Stoleson, and Carl G. Smith III¹

Oak forest regeneration and management are important economic and ecological concerns in the Central Hardwood region and elsewhere. Similar concern exists in the nearctic-neotropical migratory avifauna. Wood warblers (Parulidae) are prominent among this avifauna. Extensive harvest of North American oak forests in the late 19th and early 20th centuries imposed great changes in occurrence, distribution, and age structure of oak forests. Oak species diversity is a point of pride for Mexico and its persistence is a point of concern. Current use of oak forest habitats by warblers and other avian species reflects a response to past wholesale changes in these forests and may guide future restoration and management activities. Our curiosity about the apparent co-occurrence of oaks and warblers led us to review the natural history of these birds and of oak forests in order to develop a base of information to suggest hypotheses and support the needs of managers and conservationists alike. This report indicates progress to date on this project. The avian family Parulidae comprises approximately 115 neotropical and north temperate species. We investigated the extent of their documented use of oak forest ecosystems and the overlap of their geographic range with that of oak forests at different stages of the life cycle. First, we searched existing literature for references to use of oak trees, oak forests, or ecosystems including oak. Second, we compiled electronic maps of distribution of oak species into a composite western hemisphere oak forest distribution map and compared that map with publicly available electronic maps of distribution of wood warbler species. Third, we compared the observed map overlaps between distribution of oaks and individual warbler species to the literature record of use of oaks by that species. We identified 446 papers during our initial literature review, of which 404 included a total of 2,542 references to use of oak forests by one or more warbler species. Of this large number of references, typically more than 90 percent refer to general use of forest types; only a small proportion specifically places warblers in oak trees. All 51 migratory warbler species use oak forest for some portion of the life cycle: 49 during the breeding season, 47 during the nonbreeding season, and 42 during migration. References to use of oak forest during the breeding season include all of the migratory species except Connecticut warbler, *Oporornis agilis*, and Cape May warbler, *Setophaga tigrina*. By contrast, the record of oak use by nonmigratory species is poorly documented, including 61 references to use of oak by 10 species. The mapped ranges of 38 of 60 nonmigratory species were found not to overlap oak forests in this review. However, three of these nonmigratory species, flame-throated warbler, *Oreothlypis gutturalis*; golden-browed warbler, *Basileuterus belli*; and collared redstart, *Myioborus torquatus*, show no range overlap but are listed in references as using oak forests. Furthermore, these migratory birds were shown to use, or are believed to use oak forest for a portion of the life cycle: 51 species during the breeding season, 46 during the nonbreeding season, and 45 during the migratory period. Forty-one migratory and 22

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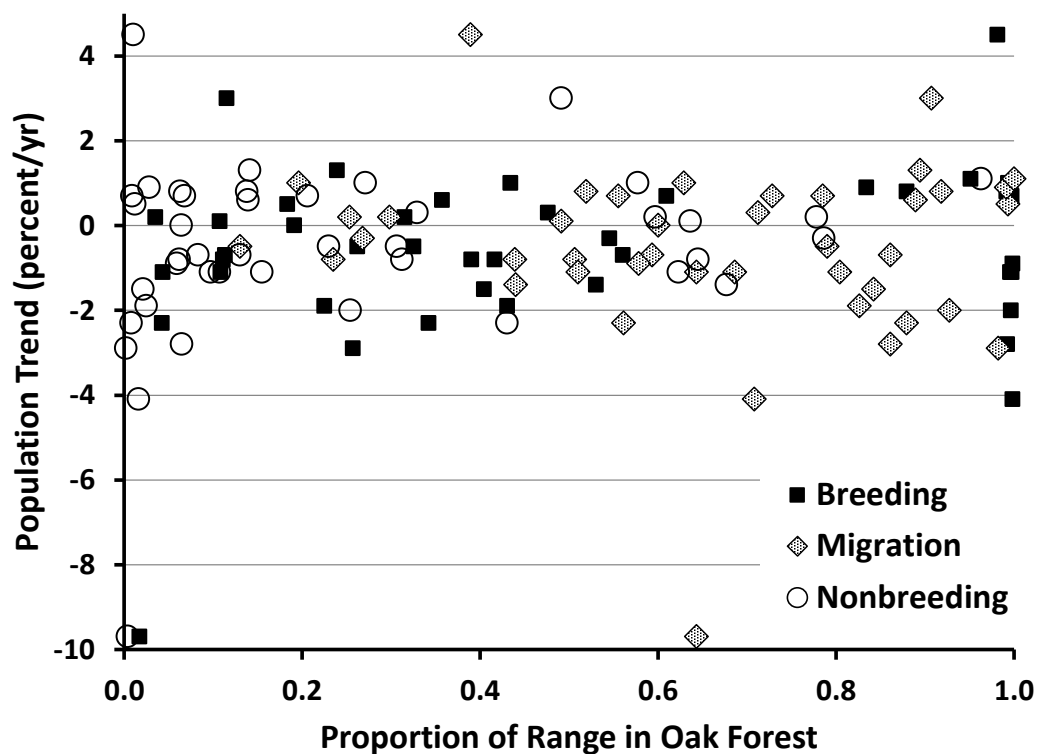


Figure1.—Relationship between population trends of warblers and proportion of range in oak forests during breeding, migration, and nonbreeding periods.

nonmigratory species use oak forest ecosystems throughout the year, of which 15 migratory and 10 nonmigratory species include more than 20 percent range overlap with oak forests. Additional species may do so, but insufficient data exist in our review to confirm this. Population trends of warblers are uncorrelated with proportion of range in oak forest, for any of the 43 species for which sufficient data exist to plot the relationship (Fig. 1). Among three warblers listed as endangered by the U.S. Fish and Wildlife Service, golden-cheeked warbler (*Setophaga chrysoparia*) depends upon oak forest ecosystems for nonbreeding habitat and Bachman's warbler (*Vermivora bachmanii*) is associated with cane (*Arundinaria gigantea*) stands in mixed oak bottomland forests. Two of 18 additional species also listed by International Union for Conservation of Nature (IUCN) in some category of vulnerability are long distance migrants, golden-winged warbler (*Vermivora chrysoptera*, IUCN Near Threatened) and cerulean warbler (*Setophaga cerulea*, IUCN Vulnerable). Central American resident pink-headed warbler (*Cardellina versicolor*, IUCN Vulnerable) and several migratory warbler species depend heavily upon oak forests and associated ecosystems. Resident warblers of the northern Andes Mountains also use oak forests. These birds, and other wildlife species, have a substantial stake in the successful restoration, sustainable management, and dependable regeneration of oak forests. (Citations available on request to phamel@fs.fed.us.)

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AUTUMN BAT ACTIVITY IN SELECTION HARVESTS AND INTACT FOREST STANDS LOCATED NEAR HIBERNACULA

Scott Haulton¹

Indiana's Harrison-Crawford State Forest (HCSF) is home to several major bat hibernacula, including the well-known Wyandotte Cave complex. HCSF is also a 24,000 acre actively managed forest which provides roosting and foraging habitat for bats of many species during the nonhibernation seasons. The autumn prehibernation period is particularly important at HCSF since high concentrations of bats use the caves and surrounding forest to prepare for hibernation or as a temporary stopover during migration. To determine if habitat use near hibernacula was affected by timber harvesting, bat activity in forest stands that had recently received selection harvests was compared with activity levels observed in "intact" mature forest stands (i.e., >15 years since single-tree selection harvest). Anabat SD2 acoustic detectors (Titely Scientific, Columbia, MO) were used to monitor bat activity at randomly located sites within 22 paired harvested and intact stands on HCSF during September-November 2012. All sampling sites were within 2 miles of a cave entrance serving a known bat hibernaculum. Mean distances were similar between harvested (0.9 miles) and intact (0.92 miles) sampling sites and the entrance of the closest known hibernaculum. Bat calls were identified to species group based on call characteristics using three automated software packages, EchoClass (U.S. Army Engineer Research and Development Center, Vicksburg, MS), Kaleidoscope (Wildlife Acoustics, Inc., Concord, MA), and BCID (Bat Call Identification, Inc., Kansas City, MO). For all species groups, activity levels were greater ($P < 0.05$) in recently harvested stands. Results indicate that recent selection harvests may be an important resource to foraging bats during migration and the prehibernation period.

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STATEWIDE PRESCRIBED FIRE NEEDS ASSESSMENT FOR ILLINOIS

Bruce M. Henry and Charles M. Ruffner¹

Extensive efforts are underway to increase the use of prescribed fire across many areas of Illinois. Prescribed fire use has certainly increased over the last decade but no entity has tried to assess these efforts using any standardized criteria nor have they made an attempt to disseminate the results to constituency groups. Many entities using prescribed fire to manage natural areas generally wish to increase their effectiveness and scope across the landscape. However, to do so would mean overcoming several obstacles to expansion and development of all fire programs including funding requests and training activities for a largely volunteer force of burn crew members. Before any coordinated expansion in the prescribed fire community occurs, it would aid planning efforts if the fire community were to conduct a frank and timely assessment of our capacity to plan, conduct, and monitor prescribed burns on lands within our state. A survey document has been developed and disseminated to all prescribed fire practitioners statewide to collect specific information about prescribed fire use. The data gathered through this survey along with the use of LANDFIRE spatial data layers will help quantify the amount and types of lands currently under prescribed fire management as well as qualify the current fire return intervals and FRCC (fire regime condition class) that the lands are in. This information will allow researchers to determine if prescribed fire managers in Illinois are meeting their burn objectives and specify the particular needs and limitations of practitioner burn programs.

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GAP AND UNDERSTORY LIGHT REGIMES ALTER STAND SIZE INEQUALITIES IN GARLIC MUSTARD

Amey Libman, Matthew Myers, Brittany E. Pham, and Kelly D. McConnaughay¹

ABSTRACT

Garlic mustard (*Alliaria petiolata*) is a biennial herb native to Europe that has become invasive to North American forests. Monospecific populations of first and second year garlic mustard plants were collected from both understory and gap locations in a temperate forest in central Illinois to determine how light regime influenced the formation of population size inequalities over the course of a full growing season. Aboveground biomass was measured for individuals within populations consisting of first or second year plants, and size inequalities were calculated using Gini coefficients. Greater size inequalities occurred in understory locations for first year populations at the end of the growing season, whereas second year populations exhibited larger size inequalities in gaps throughout the growing season. Results indicate that age classes of garlic mustard respond differently to varying light regimes, which could have meaningful implications on the genetic makeup of subsequent generations and future management strategies.

Introduction

Garlic mustard (*Alliaria petiolata*), a biennial herb introduced to North America in the late 1800s, has become invasive throughout temperate forests in North America (Clapham et al. 1952, Nuzzo 1991). Garlic mustard's competitive advantage may be related to its ability to successfully grow and reproduce under light conditions ranging from undisturbed canopy to areas of moderate disturbance (Myers and Anderson 2003). Garlic mustard grows in dense monospecific clusters, which can exhibit strong intraspecific competition resulting in alternating-aged cohorts within populations (Bauer et al. 2010). We predict that more pronounced size inequalities will develop within garlic mustard populations growing in forest canopy gaps and less developed size inequalities will be found in populations growing in intact understory (Weiner 1985). Our objective was to survey first and second year garlic mustard plants within gap and closed understory populations to evaluate if size inequalities exist in these populations and to consider the consequences of garlic mustard population size structure for forest management.

Methods

Gap and understory locations were determined based on visual inspection and confirmed by quantum flux measurements in a temperate, mixed hardwood forest in Peoria, IL. Sample populations of first and second year garlic mustard were harvested every 2 to 3 weeks from April through July 1999. Individual (aboveground) plants were dried, biomass was determined, and population size inequalities were calculated using Gini coefficients (Weiner 1986). Data were analyzed using a three-way ANOVA with harvest date, stand age, and light regime as fixed main effects.

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Results

- Understory light levels declined rapidly early in the season and remained low throughout the growing season while light levels in gaps were higher and more variable. Gaps had more overhead light and understories had more side light.
- Mean plant size within garlic mustard populations increased throughout the growing season for both first and second year plants, but did not differ by gap vs. understory location within either age group.
- Size inequalities in first year populations were not well-developed early in the growing season, but became more pronounced as the season progressed. Conversely, size inequalities for second year populations were initially high, and declined through the season.
- Size inequalities for second year populations were more pronounced in gap than in understory environments, particularly early in the growing season.

Discussion

Lack of appreciable size inequalities in first year populations early in the season is not unexpected, as the initially very small plants would not likely compete with each other for light; size inequalities typically develop over time as individuals grow larger and neighboring plants' canopies come in proximity to each other such that intraspecific competition for light becomes more pronounced (Weiner 1986). Size inequalities in first year populations increased as expected throughout the growing season.

Less expected were the large size inequalities in second year populations at the start of the growing season. If these populations exhibited similar population size structure to the first year populations in the current study, it appears that size inequalities became more pronounced during the overwintering months. This would occur if the smaller individuals within the population at the end of the first growing season were at a competitive disadvantage as growth recommenced in early spring. The reduction in size inequalities in second year stands is consistent with trends in size structure in populations undergoing size-specific mortality such that the smallest individuals are being "thinned" out of the population (Weiner and Thomas 1986).

The greater size inequalities of second year populations in gaps relative to those in the understory that are apparent early in the growing season may indicate that overwintering mortality was more pronounced in gaps. This is perhaps not surprising, as gaps are found to exhibit more extreme temperatures, humidities, and wind levels, and are thus likely to present a more hostile overwintering environment to these biennial herbs.

Although gap and understory environments differed in light conditions (e.g., directionality) and thus the potential for asymmetric competition, size inequalities were not appreciably different in these environments for first year populations throughout the growing season. The apparently greater overwinter thinning experienced in gap populations was more likely due to factors other than light availability, as it is assumed that these plants are largely photosynthetically inactive, and indeed are under leaf and snow pack during the winter months. Nonetheless, gap environments did result in greater size inequalities, and thus possibly greater genetic bottlenecks, for these garlic mustard populations. Since vegetative biomass predicts reproductive potential, greater vegetative biomass size inequalities in a population predict greater inequalities in reproductive fitness.

A better understanding of the controls on population size inequality in garlic mustard populations may help in the management of this invasive weed. Our data suggest that selective logging or stand thinning in forests could increase overwintering mortality and decrease genetic diversity in garlic mustard populations.

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FACTORS LIMITING OAK REGENERATION IN SOUTH CENTRAL PENNSYLVANIA

Robert P. Long, Aaron D. Stottlemeyer, Patrick H. Brose, and Stephen Wacker¹

Oak (*Quercus* spp.) regeneration on the 94,000+ acre Tuscarora State Forest has been problematic for decades. Early work in the 1970s implicated acorn insects, rodents, and deer as the major factors limiting oak regeneration. Foresters have used deer exclosure fencing, scarification, shelterwood harvest followed by prescribed burning, herbicides, and other methods, but with only limited success. The forest lies within the Ridge and Valley physiographic province, and soils derived from bedrock in the Tuscarora formation (Tf) can readily regenerate oaks. However, oak regeneration is problematic on the older Juniata formation (Jf) and on the younger lower Clinton formation. In 2013, three plots, each with 230 northern red oak (*Q. rubra* L.) (NRO) acorns, were planted on eight sites, four sites on soils that developed on the Tf and four sites on soils developed on the Jf. Preliminary results indicate heavy losses due to both deer browsing and small mammal clipping of emerging seedlings. In early July, 32 percent of NRO seedlings had germinated and survived on the Tf and 34 percent on the Jf. By August, only 25 percent of NRO seedlings survived on the Tf and 29 percent on the Jf. Duff samples from all 24 planted plots indicate duff thickness and mass are significantly ($P \leq 0.05$) greater for planted sites on the Tf compared with the Jf. Duff mass averaged 92 Mg/ha for stands on the Tf while it was only 61 Mg/ha on the Jf. Duff thickness appears highly variable and a possible impediment to acorn germination and establishment.

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CHARACTERIZATION OF MATERNITY ROOSTS OF INDIANA BAT IN SOUTHERN ILLINOIS

Karen E. Mangan, Casey J. Bryan, and Margaret M. Anderson¹

Indiana bat (*Myotis sodalis*), a federally endangered species, uses bottomland forests for both summer foraging and roosting habitat. A limiting factor for this species is the availability of roost trees, which provide day resting and maternity habitat. Most female Indiana bats form summer roosting colonies of 25 to 100 individuals. An individual Indiana bat maternity colony may use several different roosts to provide a range of environmental conditions needed for raising young. Colonies generally use both primary and secondary roost trees. Primary trees are typically larger snags with solar exposure that provide proper roosting conditions (cover and correct temperature) most of the time (Carter and Feldhamer 2005). Secondary roosts are generally used by fewer bats. Live trees that have exfoliating bark, such as shagbark hickory (*Carya ovata*), are known to have a relatively high value as roosts but are generally used as secondary roosts because bark structure limits the number of individuals that are able to roost together under the bark in one area of the tree. To manage for the Indiana bat, a sustained supply of suitable roost trees is critical. Our objectives were to characterize maternity roosts of Indiana bats and determine the optimal density of suitable roost trees necessary to support a healthy maternity colony.

Within Cypress Creek National Wildlife Refuge, seven 100-acre bottomland hardwood stands were surveyed using a point transect method. This habitat inventory covered bottomland hardwood forest with known Indiana bat maternity colonies. Within each stand, four transects consisting of five plots each were surveyed. Within the 0.2-acre sample plots, all snags and hickories greater than 5 inches in diameter were measured for height, diameter at breast height (d.b.h.), tree condition, decay class, and percentage bark remaining. Percentage canopy cover was also measured in the four cardinal directions at each tree. Known roosts (bats present), random snags (no bats confirmed), and hickories (both known roosts and random) were compared. Known roosts were located by tracking female Indiana bats using radio telemetry. SAS statistical software (SAS Institute Inc., Cary, NC) was used to generate parameter estimates for each tree and stand as well as combined stands where individual maternity colonies were located. A general linear model analysis ($\alpha = 0.05$) was used to compare diameter and height of known Indiana bat roosts to random snags and hickories. A chi-squared statistical analysis was used to compare tree condition, decay class, percentage bark remaining, and percentage canopy cover of known roosts to random snags and hickories.

Thirty-seven different known roosts of 11 different tree species were located. Average snag density per stand ranged from 3.3 snags/acre to 10.5 snags/acre with an average of 4.4 snags/acre across all stands. Hickory density ranged from 0.5 hickories/acre to 5.0 hickories/acre with an average of 1.75/acre across all stands. Initial results indicate known Indiana bat roost trees tend to have larger diameters and heights than random snags and hickories. Compared to known roosts, random snags were typically in a more degraded condition. Most were degraded to the point that only the bole was

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remaining whereas snags used as roost trees retained many twigs, or at least the large limbs. For the stage of decay, random snags covered several stages of decay whereas roost trees tended to be weakly decayed, with the wood still fairly hard and with loose bark. Trees used by roosting bats (snags and hickories) tended to have greater than 50 percent of the bark remaining. Unlike hickories, most snags (both known roosts and random snags) in the study had less than 50 percent canopy cover.

While the general roosting ecology of Indiana bats has been described, few studies have looked at actual roost availability and made recommendations on snag densities to assist land managers in managing for this species. Existing recommendations suggest optimal conditions for Indiana bat roosting occurs at a density of 16 to 17 trees/acre with a tree d.b.h. greater than 9 inches and retaining more than 25 percent of the bark (Garner and Gardner 1992). While we found general roost characteristics similar to those in other studies, the densities of potential roosts on Cypress Creek National Wildlife Refuge were lower than existing recommendations. We recommend further research into the snag densities necessary to sustain healthy maternity colonies.

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DISPERSAL AND SEED VECTOR TRANSPORT OF JAPANESE CHAFF FLOWER

Travis Neal and David J. Gibson¹

Nonnative invasives are species that are introduced to a new environment and they frequently cause ecological problems within otherwise functional ecosystems. Invasive species have been noted to cause local extirpations and change community structure and function. An experiment was conducted to assess dispersal of a relatively recent invasive species in order to gain insight on the transport of its seeds. Unintentional dispersal of seeds attached to the clothes and shoes of humans has been recognized, but only recently have efforts been made to quantify the species dispersed. The nonnative invasive plant Japanese chaff flower (*Achyranthes japonica*), has morphological adaptations that appear to aid in seed dispersal that may allow it to spread over large distances. A previous study has noted mortality of young birds when ensnared by the fruiting stalks of Japanese chaff flower. When seeds are mature and ready to disperse, it is likely that they can become attached to the fur and feathers of game, such as white-tailed deer (*Odocoileus virginianus*) and turkey (*Meleagris gallopavo*), or the clothing of people, thus actively dispersing the seed. Few empirical investigations have occurred to monitor the dispersal by human aided vectors. Seeds are commonly observed on the surface clothing of people but there is currently limited empirical data on clothing as a seed vector. Most importantly, this experiment was conducted to gain an understanding of the movement and dispersal of seeds to plan control practices to reduce the spread of particular invasive species.

A 2-year experiment was conducted to assess dispersal of Japanese chaff flower, to gain insight on the transport of its seeds, and to investigate the efficacy of potential seed dispersal vectors at Chestnut Hills Nature Preserve, in southern Illinois. Japanese chaff flower dispersal data were recorded from 50 randomly located 1-m² plots from 2012-13. We also collected data on plant height and cover, seed rain, seed production, slope, canopy cover, and species richness within each plot. Soil samples were collected from the plots in autumn 2012 and analyzed for pH, conductivity, total nitrogen, and total carbon.

Deer fur, turkey feathers, and cotton fabric were systematically moved across the plants in each plot to “collect” seed. The materials were combed to remove and count the seeds that had become attached. Seed rain estimates were determined from counts in seed traps constructed from pie pans coated with Tanglefoot® (ConTech Enterprises, Victoria, BC, Canada) and placed on the ground to collect seeds that fell from mature adult plants. The field experiment showed that seeds are readily dispersed by deer, birds, and humans. The fewest number of seeds were collected by turkey feathers (28.88 ± 2.52 seeds/m²), while twice as many seeds were collected on deer fur and cotton fabric (60.44 ± 8.21 and 56.61 ± 6.53 seeds/m², respectively). Deer and turkey are common in the forests that Japanese chaff flower is invading, as are humans (principally wild game hunters). These vectors collect large numbers of seed through brushing against the Japanese chaff flower. This ‘hitch-hiking’ mode of secondary dispersal is enabling Japanese chaff flower to readily invade plant communities of southern Illinois as well as the surrounding Ohio and Mississippi River Valleys.

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EVALUATING THE SUCCESS OF OAK AFFORESTATION ON FORMER AGRICULTURAL LANDS IN SOUTHERN ILLINOIS

Joshua B. Nickelson, Eric J. Holzmüller, and John W. Groninger¹

The establishment of oak (*Quercus* spp.) plantations has greatly increased in practice to reduce fragmentation and to promote wildlife habitat and valuable timber production across the midwestern United States. However, influences such as competing vegetation, previous land cover, plantation size, and site preparation techniques may result in varying outcomes on restorative successes. We established 237 plots (0.05 ac) in 32 oak plantations located within Crab Orchard National Wildlife Refuge (Williamson County, Illinois) 15-18 years after mechanical planting. Sampling data for all trees included species, diameter, and lianas existence on the main bole of the tree. Additionally, an evaluation of free-to-grow status was documented for all oaks present, and an estimation of cover of autumn olive (*Elaeagnus umbellata*) and Japanese honeysuckle (*Lonicera japonica*) was completed within the plots. In general, sites with the previous crops of soybeans and clover tended to have greater oak survival, more trees per acre, larger oak diameters, more free-to-grow oaks, and fewer lianas than sites that had been fallow at least 2 years prior to planting. These results will be used to develop management recommendations in similar afforestation efforts throughout the Midwest.

The content of this paper reflects the views of the authors(s), who are responsible for the facts and accuracy of the information presented herein.

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POTENTIAL GAINS FROM BREEDING AND SELECTION OF EASTERN COTTONWOOD AND HYBRID POPLARS ON LOWER MISSISSIPPI ALLUVIAL FLOODPLAIN AND UPLAND SITES

Oludare S. Ogunlolu, Randall J. Rousseau, B. Landis Herrin, and Jason C. Mack¹

Abstract.—*Populus* is a genus of exceedingly fast-growing trees, with tremendous potential for sawtimber, pulpwood, and as a source for renewable biomass energy. The aim of this study was to determine growth rates, adaptability to different environments, and disease resistance of selected eastern cottonwood and hybrid poplar clones on alluvial (i.e., Lower Mississippi Alluvial Valley) and uplands sites. This study was established over a 4-year period (2010 and 2013), using plant materials from four different sources and taxa. Results from the 2010 and 2011 upland test sites showed that nine hybrid poplar clones exhibited growth rates great enough to place them in the top 15 percent of their respective test population. Fourteen eastern cottonwood (*Populus deltoides*) clones were also among the top 15 percent of the test population. On the alluvial test site, eastern cottonwood clones performed better than hybrid poplars in term of growth and disease resistance. The top performing clone at age 3 years of the 2010 trial was 8019 (0.56 ft³), and age 3 for the 2011 trial was 26-2 (0.44 ft³). *Septoria* canker was generally higher in hybrid poplars on the alluvial sites compared with the upland sites.

INTRODUCTION

Eastern cottonwood (*Populus deltoides* Bartr.) is the fastest growing hardwood species in the southern United States (Garnett et al. 2008, Kelliher and Tauer 1980) and attains its best growth on newly developed alluvial soils that are high in natural fertility and moisture availability (Garnett et al. 2008, Keith and Coleman 2010). When planted on less fertile upland soils, eastern cottonwood has not demonstrated rapid growth (Kline and Coleman 2010).

Hybrid poplars are known to exhibit hybrid vigor, thus making them superior to native *Populus* species and commercially significant for a number of geographic areas of the United States (Stettler et al. 1996). However, hybrid poplars have not demonstrated that same ability in the lower Mississippi Alluvial Valley (LMAV) of the southern United States due to their susceptibility to *Septoria musiva*, a common leaf spot disease that can manifest into stem cankers and result in mortality. Eastern cottonwood is susceptible to *Septoria* leaf spot, but is resistant to this disease occurring as a canker.

Hybrid poplar testing in the LMAV has been limited and had disappointing results. However, new selections of hybrid poplars may prove worthwhile even in the LMAV if they show resistance to *Septoria* and superior growth to the best cottonwood clones. This study is focused on these new selections of both eastern cottonwood as well as different hybrid poplar taxa to determine adaptability, growth, and disease resistance on sites in and outside of the LMAV.

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Objectives

The objectives of these studies were to determine performance ability of hybrid poplars on LMAV sites in the presence of diseases, to determine if cottonwood clones will demonstrate rapid growth on upland sites, and to estimate the type and intensity of diseases that may inhibit the performance of hybrid poplars on sites in Mississippi.

METHODS

In 2010, a clone test identified as the 2010 Populus Consolidated Trial was established on two test sites: an alluvial site near New Madrid, MO, and an upland site near Pontotoc, MS. Clones used in this test originated from four cooperators and were selected for their performance in the geographical region of the cooperators. The four cooperators were ArborGen, GreenWood Resources, Mississippi State University, and University of Minnesota. Each group donated 20 clones; GreenWood Resources and the University of Minnesota provided all hybrid taxa, while ArborGen and Mississippi State University provided primarily eastern cottonwood clones. The 80 clones included in the test consisted of a combination of taxa, which included a pure eastern cottonwood and five hybrid poplar taxa. Taxa identification is as follows: eastern cottonwood—*P. deltoides* (DD); *P. deltoides* x *P. nigra* (DN); *P. deltoides* x *P. maximowiczii* (DM); *P. deltoides* x *P. trichocarpa* (DT); and *P. nigra* x *P. maximowiczii* (NM).

In 2011, a second round of testing was established with the 2011 Populus Consolidated Trial, consisting of 60 unique clones (i.e., 15 from each cooperator) and 20 common clones (i.e., 5 clones from each cooperator) that were planted in the 2010 trial. Under this strategy, the 2010 and 2011 trials are connected by 20 common clones. Due to flooding, no alluvial field site was planted in 2011.

The planting stock was canker free, dormant, unrooted cuttings, with all hybrid poplars being 9 inches long, while the eastern cottonwood was 18 inches long, with all cuttings having a top stem diameter not greater than 0.5 inches, and a bottom diameter of less than 1 inch. After the cutting of the plant materials into specified sizes, they were hydrated for 24 hours in water that contained the labeled rate for Admire® Pro (Bayer CropScience, Research Triangle Park, NC).

Following the hydration treatment, the cuttings were placed in 4 mil plastic bags and placed in a cooler at 35 °F until planting. Prior to planting, the site was disked, then subsoiled at 9 foot spacing and a depth of 14 inches. Both the 2010 and 2011 tests employed a nested design consisting of three blocks, four sources, and 20 clones/source. Each clone was arranged in two-tree row plots at a spacing of 6 feet by 9 feet. Goal 2XL® selective herbicide was applied at 64 oz per acre immediately after planting. Competing vegetation (weed and vines) control was done mechanically and by hand-pulling throughout the study period.

Annual measurements taken for all tests included total height at age 1 year, diameter at breast height (d.b.h), and total height at ages 2, 3, and 4 years. In addition to growth measurements, all of the tests were graded for overall crown leaf health and retention (CLHR) and the presence of stem cankers. All measurements taken involved height in feet to the nearest tenth of a foot, and d.b.h in inches to the nearest tenth of an inch for all tests.

Table 1.—Top performing clones, age 3 years, 2010 Consolidated Trial at Pontotoc, MS, (upland)

	Clones	Taxa	Origin	Survival (%)	DBH (in)	Height (ft)	Volume ¹ (ft ³)
1	8019	DM	GreenWood Resources	100	2.7	28.1	0.6360
2	AG443	DD	ArborGen	83	2.7	22.5	0.4998
3	147-1	DD	Mississippi State Univ.	100	2.6	23.0	0.4864
4	80-5	DD	Mississippi State Univ.	67	2.3	19.8	0.4842
5	7388	DM	GreenWood Resources	83	2.4	22.3	0.4429
6	24-128	TD	ArborGen	100	2.4	22.0	0.4340
7	110412	DD	Mississippi State Univ.	100	2.5	22.7	0.4267
8	105-1	DD	Mississippi State Univ.	100	2.3	19.3	0.4161
9	6323	DM	GreenWood Resources	100	2.4	22.4	0.4144
10	3-1	DD	Mississippi State Univ.	83	2.2	22.4	0.4104
11	AG412	DD	ArborGen	83	2.4	20.9	0.3804
12	7416	DM	GreenWood Resources	67	2.2	22.7	0.3715

¹ Using Krinards equation= $0.09 + .002216(D^2H)$ (Krinard 2008).

We used SAS statistical software (SAS Institute Inc., Cary, NC) to compare sources and clones within sources for each site as well as among sites.

RESULTS

Results from the 2010 and 2011 upland test sites showed that nine hybrid poplar clones exhibited growth rates great enough to place them in the top 15 percent of their respective test population. Whereas, 14 eastern cottonwood clones were also among the top 15 percent of the test population (Tables 1 and 3).

On the alluvial test site, eastern cottonwood clones performed better than hybrid poplars in term of growth and disease resistance (Table 2).

Of the clones tested in the 2010 and 2011 Populus Consolidated Trials on the upland site, 12 clones exhibited mean height growth greater than 18 feet (at 3 years). Among these clones were 15 eastern cottonwood clones and nine hybrid poplar clones representing two taxa (i.e., DM and TD), as shown in Table 1 and 3.

The top performing clone at age 3 years of the 2010 trial was 8019 (0.56 ft³); and the top performing clone at age 3 years for the 2011 trial was 26-2 (0.44 ft³) (Tables 1 and 3).

Septoria canker rates were generally higher in hybrid poplars on the alluvial sites compared with the upland sites (Fig. 1).

Table 2.—Top performing clones, age 3 years, 2010 Consolidated Trial at New Madrid, MO (alluvial)

	Clones	Taxa	Origin	Survival (%)	DBH (in)	Height (ft)	Volume ¹ (ft ³)
1	13788	DN	GreenWood Resources	16	5.6	27.8	2.0219
2	AG414	DD	ArborGen	50	4.3	29.3	1.3888
3	27-5	DD	Mississippi State Univ.	33	4.4	29.9	1.3759
4	011-32S	DD	ArborGen	67	4.4	29.3	1.3357
5	NM6	NM	Univ. of Minnesota	67	3.9	27.0	1.1253
6	4491	DT	GreenWood Resources	67	3.9	27.8	1.1034
7	9732-24	DN	Univ. of Minnesota	83	3.8	28.7	1.0606
8	3-1	DD	Mississippi State Univ.	100	3.8	26.3	1.0577
9	30-4	DD	Mississippi State Univ.	83	3.9	27.3	1.0418
10	9732-31	DN	Univ. of Minnesota	50	3.9	27.1	1.0220
11	S7C1	DD	Mississippi State Univ.	100	3.9	25.5	1.0218
12	AG413	DD	ArborGen	16	3.7	29.7	0.9910

¹ Using Krinards equation= $0.09+.002216(D^2H)$ (Krinard 2008).

Table 3.—Top performing clones at age 3 years of the 2011 Consolidated Trial at Pontotoc, MS, (upland)

	Clones	Taxa	Origin	Survival (%)	DBH (in)	Height (ft)	Volume ¹ (ft ³)
1	26-2	DD	Mississippi State Univ.	83	2.8	20.0	0.4375
2	110412	DD	Mississippi State Univ.	50	2.3	18.5	0.3069
3	S7C8	DD	Mississippi State Univ.	83	2.4	18.9	0.3312
4	AG185	DD	ArborGen	100	2.6	20.2	0.3926
5	AG229	TD	ArborGen	100	1.9	18.1	0.2348
6	AG414	DD	ArborGen	100	2.2	19.5	0.2991
7	AG434	DD	ArborGen	83	2.3	21.0	0.3362
8	AG435	DD	ArborGen	67	2.0	18.2	0.2513
9	AG439	DD	ArborGen	83	2.3	19.8	0.3221
10	6329	DM	GreenWood Resources	100	2.2	18.3	0.2863
11	6612	TD	GreenWood Resources	100	2.4	18.2	0.3223
12	7300	TD		100	2.4	18.2	0.3223

¹ Using Krinards equation= $0.09+.002216(D^2H)$ (Krinard 2008).

CONCLUSION

Hybrid poplars continue to show problems from the susceptibility of *Septoria*, even at early ages of 1 to 3 years. We will follow those hybrid poplar clones that possess *Septoria* resistance in the LMAV and upland sites to determine long-term resistance.

Eastern cottonwood growth on the uplands is as expected dramatically reduced when compared to alluvial sites.

Adaptability of selected eastern cottonwood clones to upland sites may allow new clonal mating to define a group of clones that will exhibit better performance on upland sites.

Hybrid poplars also seem to be less adapted to annual flooding when established on alluvial sites, thus resulting in additional mortality.

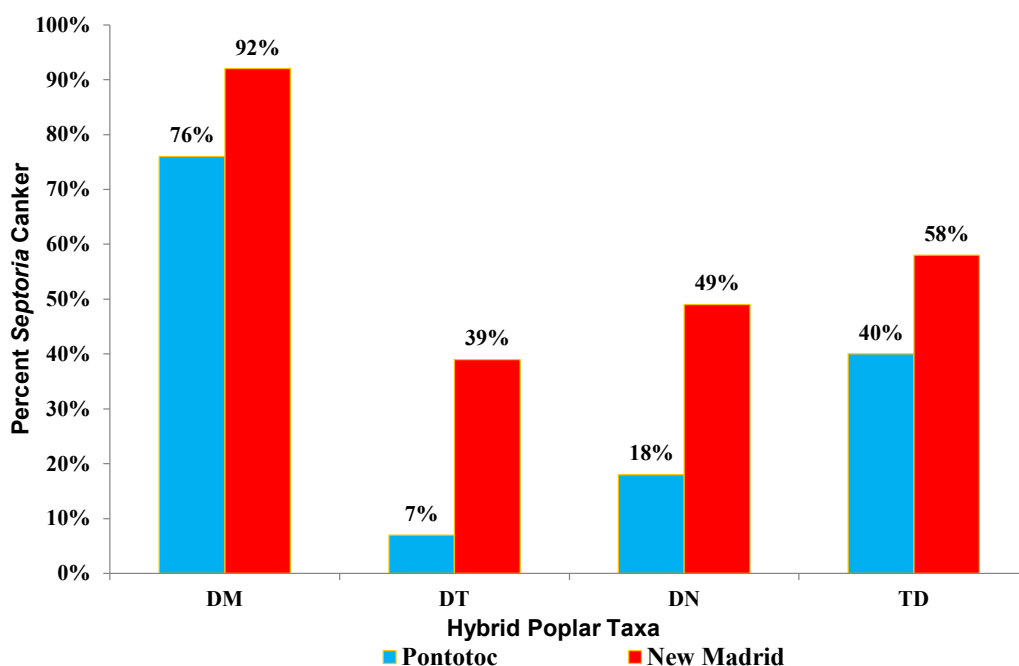


Figure 1.—Percentage of hybrid poplars with *Septoria* canker presence in 2010 Consolidated Trials by taxa for upland (Pontotoc) and alluvial (New Madrid) sites at age 4 years.

ACKNOWLEDGMENTS

This research was funded by SunGrant Initiative and College of Forests Resources, Mississippi State University. We also acknowledge cooperators ArborGen, GreenWood Resources, Mississippi State University, and University of Minnesota, donors of the plant materials. Appreciation also goes to all student workers who contributed to the success of the study.

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THE APPLICABILITY OF TILMAN'S RESOURCE-RATIO HYPOTHESIS TO FOUR AMARANTHACEAE SPECIES

Lauren M. Schwartz, David J. Gibson, and Bryan G. Young

The resource-ratio hypothesis of succession states that plant species are specialized on different proportions of limiting resources (Tilman 1982). Thus, if resource levels are sufficient, the plant will have positive growth, and will draw down resource levels leading to a reduction in population growth rate for intra- and inter-species competition. Since different plant species use the same major resources, then the resource-ratio hypothesis predicts that the species that can maintain a positive growth rate at the lowest resource level will be the best competitor for that resource. Early seedling growth of four herbaceous species in the Amaranthaceae family were studied to test the applicability of the resource-ratio hypothesis for predicting competitiveness among southern Illinois forest herbaceous plants and field crop weeds. *Achyranthes japonica* and *Iresine rhizomatosa* are two perennial species that occur in similar habitats but differ in invasiveness. *Achyranthes japonica* is a nonnative invasive species that is threatening natural forested areas and also has been observed on the margins of agricultural fields. *Iresine rhizomatosa* also occurs in forest habitats but is an endangered species in Illinois. *Amaranthus palmeri* and *A. tuberculatus* are summer annuals typically found as undesirable agricultural weeds. The objective of this study was to determine the relative competitiveness of the closely related species in comparison to soybean (*Glycine max*). A greenhouse study was conducted in which each species was transplanted at the seedling stage, so that all species were similar in growth stage, in a closed system to assess resource use of an aboveground (light) and belowground (nitrogen) resource. Resource manipulation treatments were implemented by adding nitrogen as ammonium nitrate and by shading using a 60 percent shade cloth. Total nitrogen drawdown was significantly higher in the shaded treatments when ammonium nitrate was added ($P=0.0003$), but there was no species interaction ($P>0.05$). There was, however, a significant three-way interaction between species, shading treatment, and day ($P=0.0002$). In comparison to controls, the four species each used light, but not nitrogen, when treatments were compared. These results allowed a relative R^* ranking (Miller et al. 2005) to be proposed based upon light use: *A. palmeri* > *A. tuberculatus* > *A. japonica* > *I. rhizomatosa*. Final biomass was reduced when plants were shaded ($P=0.006$) and there was a trend toward an increase in biomass with additional soil nitrogen ($P=0.08$). *Achyranthes japonica* produced the most belowground biomass of the four Amaranthaceae species in all treatment groups. *Amaranthus palmeri* and *A. japonica* had an increased amount of aboveground biomass when nitrogen was added. In the shading treatment, however, all species had a decreased amount of aboveground biomass in comparison to the controls. These results suggest that *A. japonica* could be a similar competitor with *G. max* to the *Amaranthus* species based on biomass, nitrogen use, and shade tolerance, with the assumption that all species would begin at the same seedling stage simultaneously. Applicability of the resource-ratio theory could lead to more effective management tactics by allowing prediction of susceptible areas of infestation or competitive outcomes based on resource levels.

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SALVAGING ASH FROM URBAN WOODLANDS IN SOUTHERN ONTARIO

Peter A. Williams¹

Ash species (*Fraxinus* spp.) are important upland and frequently dominant lowland trees in southern Ontario that are being affected by emerald ash borer (EAB; *Agrilus planipennis*). Ash can dominate in urban forests because of their agricultural history and site characteristics. Since it is cost-prohibitive to chemically protect woodland ash from EAB, most will be killed and might become hazards.

A harvest-salvage strategy can remove ash from urban woodlands, reducing potentially hazardous situations and debris and at the same time, recover some removal costs. Oakville and Toronto have successfully implemented harvest strategies to remove dead and dying ash from parklands. This includes developing forest management plans and regeneration prescriptions approved by a registered professional forester, designating trees for removal, and planning the work using an integrated logging/arboricultural approach where conventional and small-scale harvesting methods are used. Arboricultural methods are important to help remove trees that may damage property, workers, or other trees. Cable skidders, tracked mini-skidders, and tractor skidders and forwarders have been used. The equipment used depends on site and weather conditions, equipment availability, access, and tree/forest conditions. A communication strategy aimed at the public is critical and should include public meetings, advertising, and direct contact with adjoining landowners and the neighborhood. Contractor diligence, landing management, and prompt trucking with appropriate use of the arboricultural methods and equipment are important in minimizing site and stand disturbance. Seventy-thousand board feet of logs and 150 full cords of fuelwood were salvaged and marketed from seven Oakville parks in 2013, recovering about 15 percent of the planning and removal costs.

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EFFECTS OF EVEN- AND UNEVEN-AGE FOREST MANAGEMENT ON SOFT MAST AVAILABILITY AND CORRESPONDING CHANGES TO SMALL MAMMAL POPULATIONS IN THE MISSOURI OZARKS

Elizabeth K. Olson and Alexander J. Wolf¹

The food and shelter resources that small mammals rely upon can be drastically changed after timber harvest. Significant increases in soft mast production are common after timber harvest, and berries make up a substantial proportion of small mammal diet in summer months. Increases in small mammal population densities may mirror increases of soft mast production due to enhanced survival and increased reproduction. Additionally, changes in vegetative cover, stand basal area, and canopy cover can affect small mammal habitat choices. We investigated small mammal responses to forest management practices in the Missouri Ozark Forest Ecosystem Project (MOFEP). MOFEP is a long-term, landscape scale study designed to assess the impacts of no harvest, uneven-age, and even-age forest management on wildlife and other ecosystem components in upland oak-hickory forests. Using data collected prior to and after a timber harvest in 1996, we examined *Peromyscus* mouse abundance in nine forested sites (average 400 ha, range 312-514 ha) in southeast Missouri. We constructed regression models to address the question: how is mouse abundance affected by soft mast abundance (i.e., berry counts), soft mast vegetative coverage, basal area, and canopy cover? Based on a multiple-regression model with mouse abundance, berry counts, soft mast vegetative coverage, and harvest type, preliminary results suggest that berry counts in the previous year and the number of mice in the previous year positively affect mouse abundance (overall regression model: $df = 44$, $F = 7.01$, $P < 0.0001$, $R^2 = 0.47$). Although there were not enough data to include basal area and canopy cover in the multiple regression model, single regression models of each variable showed some evidence of a negative effect on mouse abundance (basal area: $df = 53$, $F = 6.60$, $P = 0.0132$, $R^2 = 0.1125$; canopy cover: $df = 44$, $F = 15.08$, $P = 0.0003$, $R^2 = 0.26$). This is perhaps due to reduced production of soft mast brought about by shading soft mast species in the understory. These results show support for the hypothesis that timber harvests can increase mouse abundance by providing more soft mast berries. Future research directions include additions to the regression model of acorn production and climate data (precipitation and temperature) as environmental filters on *Peromyscus* abundance. As forest land managers widen their focus to encompass landscape scale and ecosystem wide impacts, results from the MOFEP experiment will provide important information about long- and short-term small mammal population dynamics in response to different forest management practices.

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Proceedings from the 2014 Central Hardwood Forest Conference in Carbondale, IL. The published proceedings include 27 papers and 47 abstracts pertaining to research conducted on biofuels and bioenergy, forest biometrics, forest ecology and physiology, forest economics, forest health including invasive species, forest soils and hydrology, geographic information systems, harvesting and utilization, silviculture, and wildlife management.

KEYWORDS: forest management, central hardwood forest, forest ecology, silviculture, invasive species, bioenergy, wildlife management

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