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17TH

CENTRAL HARDWOOD FOREST CONFERENCE

Proceedings of a Conference held at the University of Kentucky, Lexington, Kentucky

April 5-7, 2010

Edited by:

Songlin Fei, University of Kentucky
John M. Lhotka, University of Kentucky
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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.treesearch.fs.fed.us/pubs/38058>. For the correction, see page 238 of this paper.

FOREWORD

The Central Hardwood Forest Conference is a series of biennial meetings dedicated to the sustainability and improvement of the 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. This, the 17th Central Hardwood Forest Conference, is hosted by the Department of Forestry, University of Kentucky. The conference includes presentations pertaining to forest regeneration, invasive species, socioeconomics, wildlife, forest ecology, biometrics and modeling, silviculture, and forest hydrology. The conference consisted of 67 oral presentations and 39 poster presentations; most of these papers and some extended abstracts are published herein.

The 18th Central Hardwood Forest Conference (2012) will be hosted by West Virginia University and the Northern Research Station, U.S. Forest Service; the 19th Central Hardwood Forest Conference (2014) will be hosted by Southern Illinois University and the Northern Research Station; and the 20th Central Hardwood Forest Conference (2016) has been proposed to be hosted by Pennsylvania State University and the Northern Research Station.

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REVIEW PROCEDURES

A double-blind review process was used in reviewing manuscripts for oral presentations. Each manuscript was assigned to one of the editors and 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 rejected 10 percent of the proposed oral presentations and six percent of the manuscripts.

ACKNOWLEDGMENTS

The Steering Committee wishes to thank all the professionals who assisted in reviewing the manuscripts for the proceedings. The Steering Committee is grateful to Susan Wright and other members of the U.S. Forest Service Northern Research Station communications unit for technical editing and publishing of these proceedings. The Steering Committee also thanks the following people for their assistance: Chris Reeves, Billy Thomas, and Darryl Cremeans, Department of Forestry at University of Kentucky.

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BIOMETRICS AND FOREST MODELING

THINNING STRATEGIES TO INCREASE THE REGIONAL AVAILABILITY OF OAK TIMBER IN THE MID-APPALACHIAN REGION

John R. Brooks, Jingxin Wang, and Chris LeDoux¹

Abstract.—Plot data from 6,727 fully stocked oak-hickory stands were selected from Forest Inventory and Analysis data from five ecoregions common to West Virginia, Pennsylvania, Maryland, Virginia, Kentucky, and Ohio. Each plot was thinned from below using an existing thinning algorithm, where 30, 50, and 70 percent of the existing basal area was removed. These thinning strategies were selected to capture most thinning intensities applied in this region. Utilizing three (low, average, and high) delivered pulpwood and sawtimber values and estimated harvesting and transportation costs from the ECOST simulator, we determined net thinning value for two cut-to-length (CTL) systems and two feller buncher systems. Net thinning value increased with increasing thinning intensity, higher pricing levels, and use of feller-bunchers. Under the low pricing scenario, neither CTL system was profitable based on average net value per acre. The feller buncher systems possessed, on average, a slightly positive net value with little difference exhibited by thinning intensity. An attempt was made to isolate those stand variables that would correctly identify profitable thinning scenarios. The main problem with quadratic mean diameter (QMD) and the other stand variables explored, is that negative thinning values are found across the range of the stand variable examined. Under higher price regimes and heavier thinning intensities, however, value tends to increase directly with the stand QMD.

INTRODUCTION

Many fully stocked oak-dominated stands exist in the central Appalachian region. Thinning is one operation that could be employed to provide landowners a possible source of income while improving stand condition and residual density for the remaining trees without conducting a complete harvest. Whether thinning can be done economically, given current estimates of harvest and delivery costs, is unknown. This study is designed to examine this question and investigate the feasibility of economical thinning operations in fully stocked oak-hickory stands in the central Appalachian region.

METHODS

The current study utilizes U.S. Forest Service Forest Inventory and Analysis data from all available plot data through 2006. In this analysis, each location (made up of four 24-foot radius subplots) was considered a stand. Per-acre estimates of number of stems, basal area, and volume are based on the reported unadjusted plot size. Volumes used were the reported net board foot volume, net merchantable cubic volume for pulpwood-sized trees, net cubic foot volume for sawtimber-sized trees, and merchantable dry weight of pulpwood-sized trees. For this analysis, pulpwood-sized trees are considered as trees from 5.0 to 11.5 inches

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diameter at breast height (d.b.h.) and sawtimber trees are those having a d.b.h. of at least 11.6 inches and having sufficient quality to be considered a sawtimber tree. Plot data were selected for West Virginia and the neighboring counties of the same ecological regions in Pennsylvania, Maryland, Virginia, Kentucky, and Ohio (Fig. 1). Plot selection in the states bordering West Virginia was based at the county level. All plot data were queried to select only oak-hickory stands, living trees, those having a d.b.h. greater than 4.9 inches, and those trees that are considered growing-stock quality. From this subset, only those stands that were at least “near” full stocking were included. A method for evaluating stocking based on stand density index (SDI) as published by Williams (2003) was employed. Only those stands having an SDI value of at least 125 were retained. This dataset contained 6,727 plots meeting these criteria (Fig. 2).

Each plot (stand) was thinned from below using a thinning algorithm developed by Brooks and Wang (unpublished). Stands were thinned using three thinning intensities:

- Intensity 1: Remove 70 percent of the existing stand basal area
- Intensity 2: Remove 50 percent of the existing stand basal area
- Intensity 3: Remove 30 percent of the existing stand basal area

These intensities were selected because they represent the range of most current thinning intensities utilized in Appalachian hardwoods. Tree lists were sorted from smallest to largest by d.b.h. and removed sequentially until the target basal area was approximated.

Once the thinned tree list was created, the delivered price for the thinned material was determined by stand, based on low, average, and high delivered price scenarios. For pulpwood products, the thinned merchantable dry weight was converted to green weight by multiplying by 1.85, an average value obtained as the average difference between green and dry weight estimates based on equations by Wiant and others (1979). Delivered

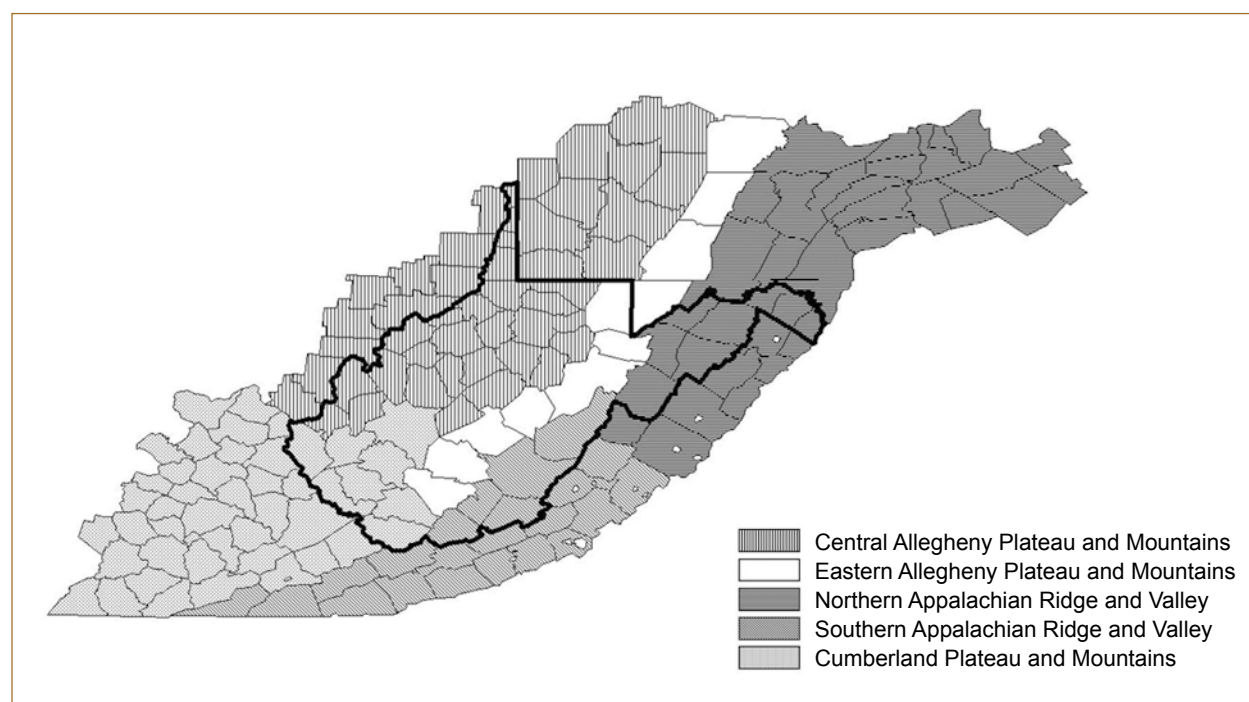


Figure 1.—Ecological regions of the central Appalachian area.

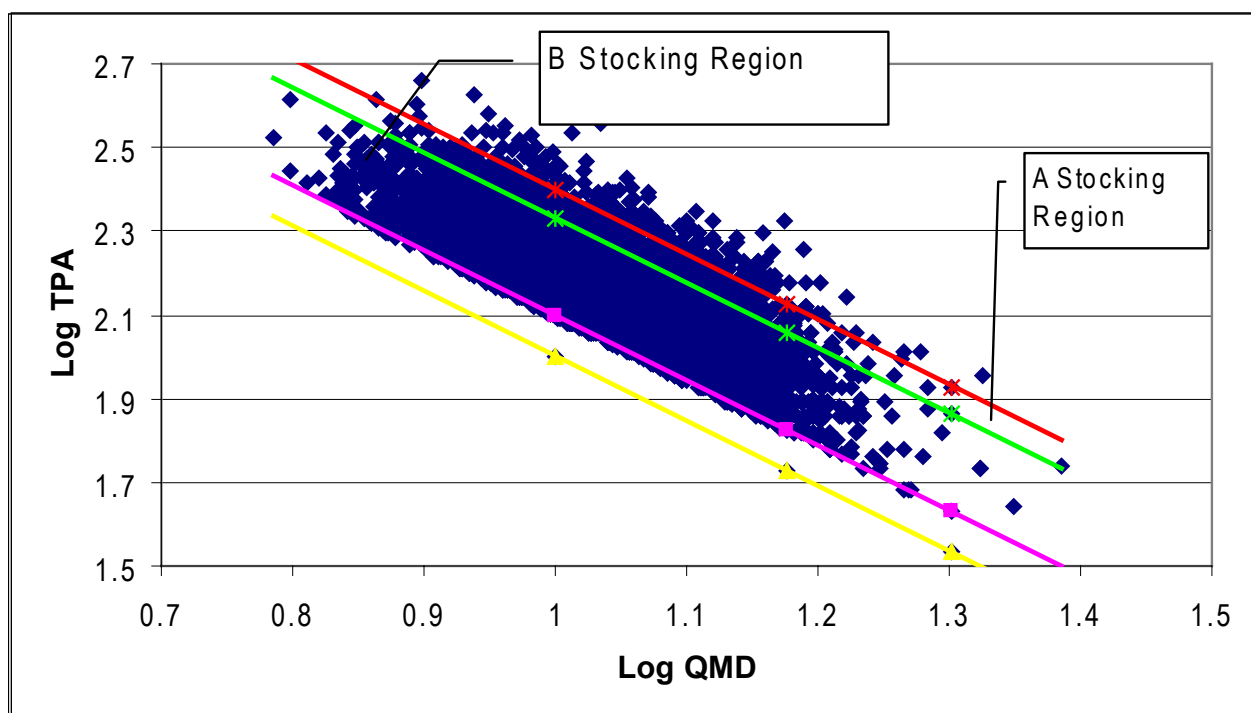


Figure 2.—Distribution of plot stand density index (SDI) for fully-stocked oak-hickory stands in central Appalachia (QMD is quadratic mean diameter and TPA is trees per acre).

price per ton for 2007 was obtained from NewPage Corporation in Luke, MD (personal communication) and the three scenarios were defined as:

Scenario 1:	Low pulpwood value:	\$15.50/ton
Scenario 2:	Average pulpwood value	\$21.50/ton
Scenario 3:	High pulpwood value	\$27.50/ton

Sawtimber value was based on delivered log value prices by species and grade. A low, average, and high value were assigned by species based on existing sale records in the Pennsylvania Woodlands Market Report (Pennsylvania State University 2007). The simulation utilized the higher product value for each tree. Thus, if pulpwood value was higher than sawtimber value for any tree, the higher value was employed. Total value per acre for the thinned stems was calculated, along with thinned volume, basal area, trees per acre, and the average diameter of the removed trees. These data were exported to ECOST simulator (LeDoux 1985, 1987) to determine harvesting and hauling costs for truck class 1 and 2 using road class 2 and 3. These classes are defined as:

Truck Class 1:	Truck tractor, 6x4, tandem axle with tandem 30- to 35-foot trailer w/additional 15- to 20-foot trailer.
Truck Class 2:	Truck tractor, 6x4, tandem axle with tandem trailer
Road Class 2:	Designated Speed 35 mph
Road Class 3:	Designated Speed 25 mph.

Harvesting scenarios are based on four types of equipment, small and large cut-to-length (CTL) systems and two feller buncher (FB) units. In this study we used harvesting performance data for two of the Timbco 400 series feller bunchers to simulate the felling, sorting, and bunching of the trees. To simulate the cut-to-length

operations, we used a Timbco T425 and a John Deere 988 to represent a small and large machine scenario, respectively. A Valmet 524 forwarder was used to simulate the transportation of the processed trees to the landing. Mechanized feller bunchers such as the Timbco 425 and the 445 models can fell, sort, and bunch the trees as they are being processed. Mechanized CTL machines such as the Timbco T425 and the John Deere 988 can fell, delimb, debark, sort, and bunch the trees. All of these machines are track-mounted and work very well on flat, moderate, or moderately steep terrain. The Timbco 445 is a larger machine and has a wider track than the Timbco 425 model. The narrower track gives the smaller machine an advantage over the 445 model in that it is easier to maneuver within the stand. The Timbco machines have Caterpillar undercarriages and can be equipped with a range of cutting heads and cut-to-length heads. In this study, all machines were equipped with single grip, fixed saw tooth felling heads. The Valmet forwarder has a load capacity of about 600 cubic feet.

This simulation is based on an average slope yarding distance of 850 feet and a one-way trucking distance of 25 miles. Restrictions on the use of a particular harvesting system included eliminating the small CTL system if average thinned diameter exceeded 14.0 inches and using the feller bunchers only if average thinned diameter exceeded 20 inches. Harvesting and hauling costs were deducted from the thinning value, discussed earlier, to determine the net thinning value.

The purpose of this study was to determine under what combinations of thinning intensity, pricing scenarios, and four commonly employed yarders, could profitable thinning operations be conducted for fully stocked oak-hickory stands in this region.

RESULTS AND DISCUSSION

Net thinning value increased with increasing thinning intensity, higher pricing levels, and the employment of feller-bunchers (Fig. 3). The results for thinning intensity and product pricing scenarios were not unexpected, but what is more important is under what equipment, pricing, and thinning intensities can these operations be expected to produce a positive net value. Average net values for Truck Class 1 and Road Class 1 were based on the mean net value for the 6,727 thinned plots. Figure 3a indicates that under low-pricing scenarios, average net value for CTL systems does not appear to be profitable under any thinning intensity. However, the feller buncher systems do show a positive net value that roughly doubles as thinning intensity is increased from 30 to 50 percent of current basal area. For the average pricing scenario, all systems appear to provide an average positive net value; the feller buncher units exhibited a higher average net value (Fig. 3b). The increase in value between the lower thinning intensity (30 percent) and the middle thinning intensity (50 percent) for the first feller buncher is approximately 1.9 times that of the lower thinning intensity. Under the high-pricing scenario, all simulations provided positive average net values. The feller buncher systems show a positive net value that roughly doubles as thinning intensity is increased from 30 to 50 percent of current basal area (Fig. 3c). The difference between the average net value under the low- and high-pricing scenarios was 3.2 and 3.6 times greater for the 30-percent and 50-percent thinning intensity, respectively.

Average net values based on utilizing Truck Class 2 and Road Class 2 are shown in Figure 4. Under the low-pricing scenario, neither CTL system is profitable based on average net value per acre. The feller buncher systems possessed, on average, a slightly positive net value with little difference exhibited by thinning intensity. Differences for the average and high pricing were similar to those shown for the net values for Truck Class 1 and Road Class 1 values, except that they were 29 percent and 17 percent lower, respectively.

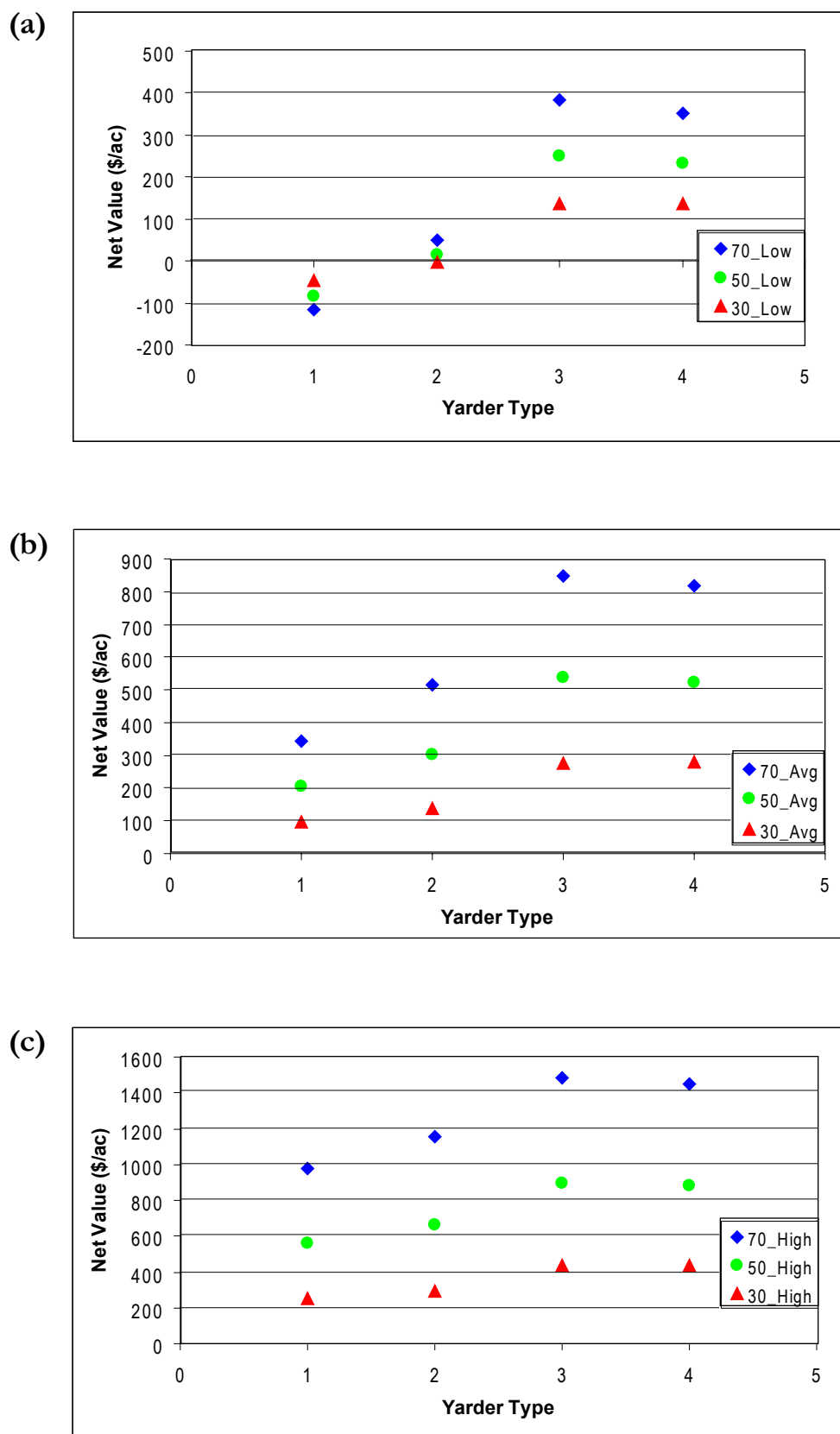


Figure 3.—Net thinning value per acre by yarder type (small CTL [1], large CTL [2], small FB [3], large FB [4]) for three thinning intensities (30-70 percent) using three pricing scenarios (low [a], average [b], high [c]) for Truck Class 1, Road Class 1.

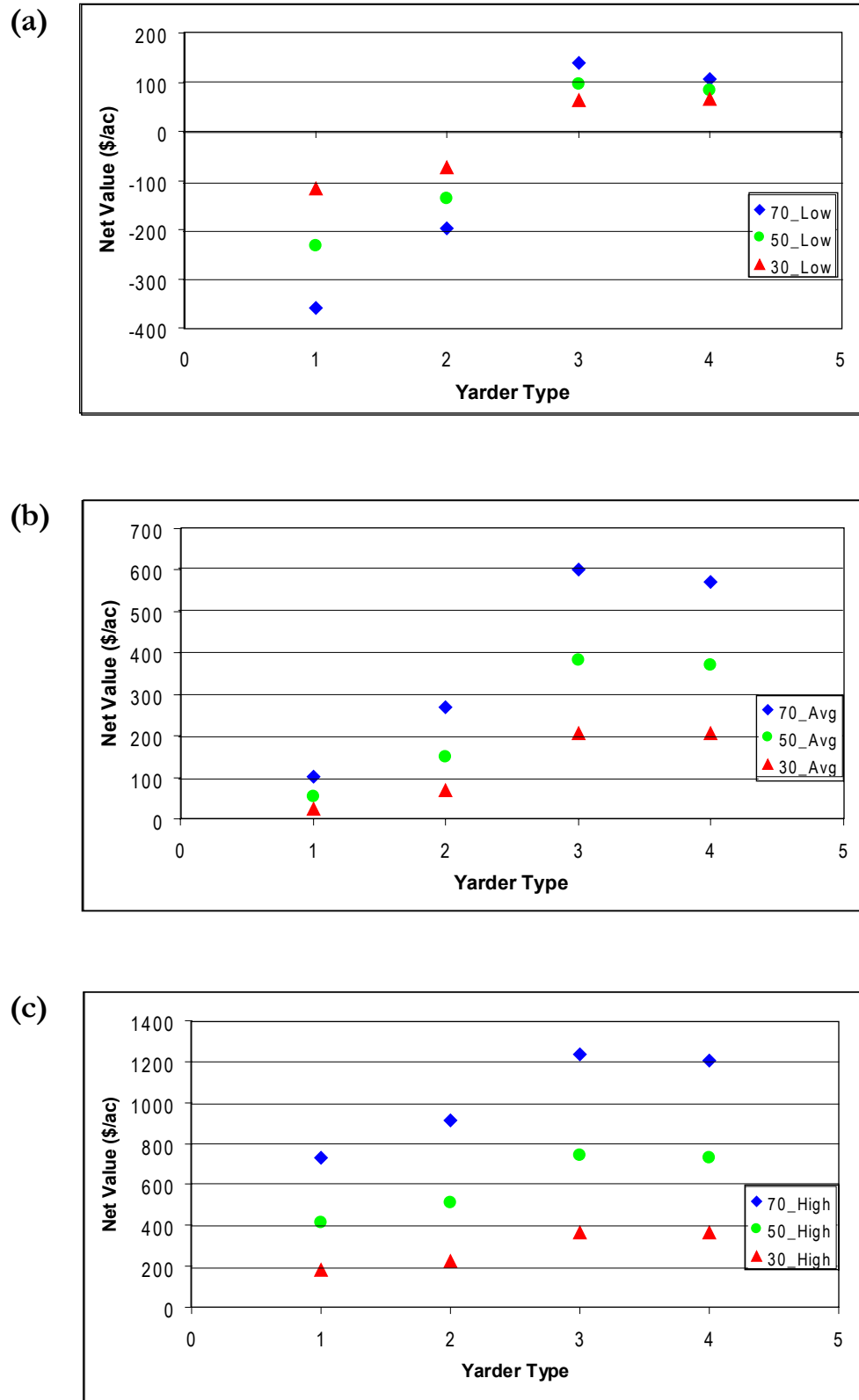


Figure 4.—Net thinning value per acre by yarder type (small CTL [1], large CTL [2], small FB [3], large FB [4]) for three thinning intensities (30-70 percent) using three pricing scenarios (low [a], average [b], high [c]) for Truck Class 2, Road Class 2.

Under the simulations tested, the smaller feller buncher provided the highest average net value under all regimes, although the values for both feller buncher systems did not vary greatly. The average net values for Truck Class 1 and Road Class 1 for the three thinning intensities (30-, 50-, and 70-percent basal area removal) and the three pricing scenarios are displayed in Figure 5.

CONCLUSIONS

An attempt was made to isolate those stand variables that would correctly identify profitable thinning scenarios. Figure 6 displays the net value per acre for each plot for the small feller buncher using the average pricing schedule at both the 30-percent thinning intensity (Fig. 6a) and the 70-percent thinning intensity (Fig. 6b) by quadratic mean diameter. The main problem with QMD and the other stand variables explored, is that negative thinning values are found across the range of the stand variable examined. It is apparent that under higher price regimes and heavier thinning intensities, value does tend to increase directly with the stand parameter being examined. Additional work on this topic is justified.

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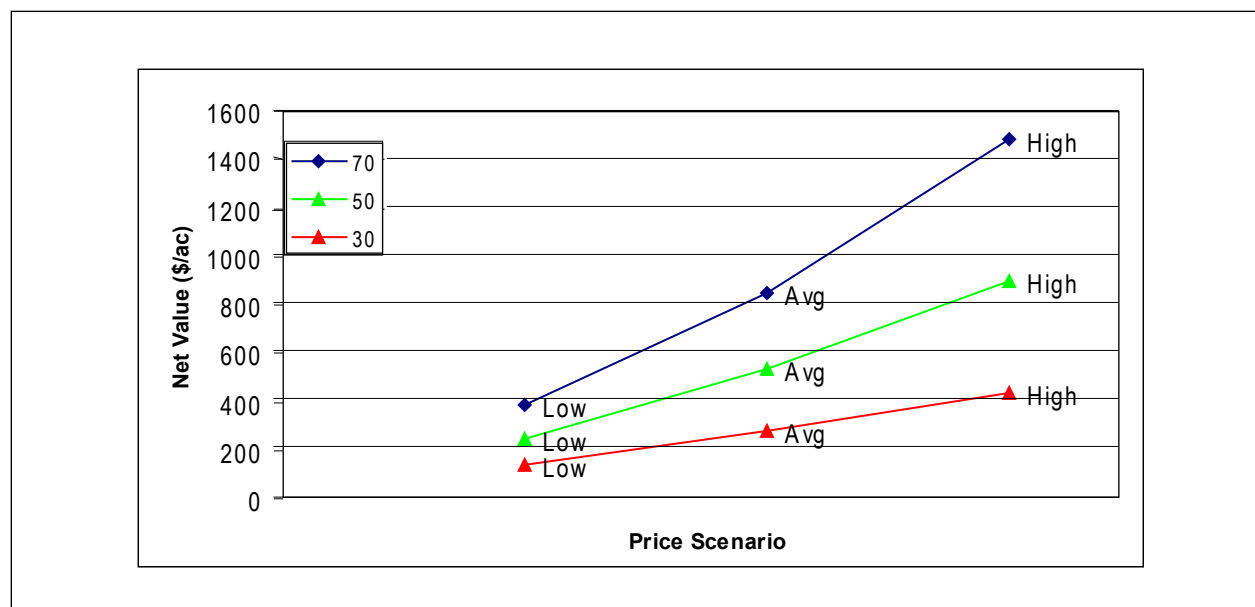


Figure 5.—Net thinning value per acre for the smaller feller buncher by thinning intensity and pricing scenario (Truck Class 1, Road Class 1).

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

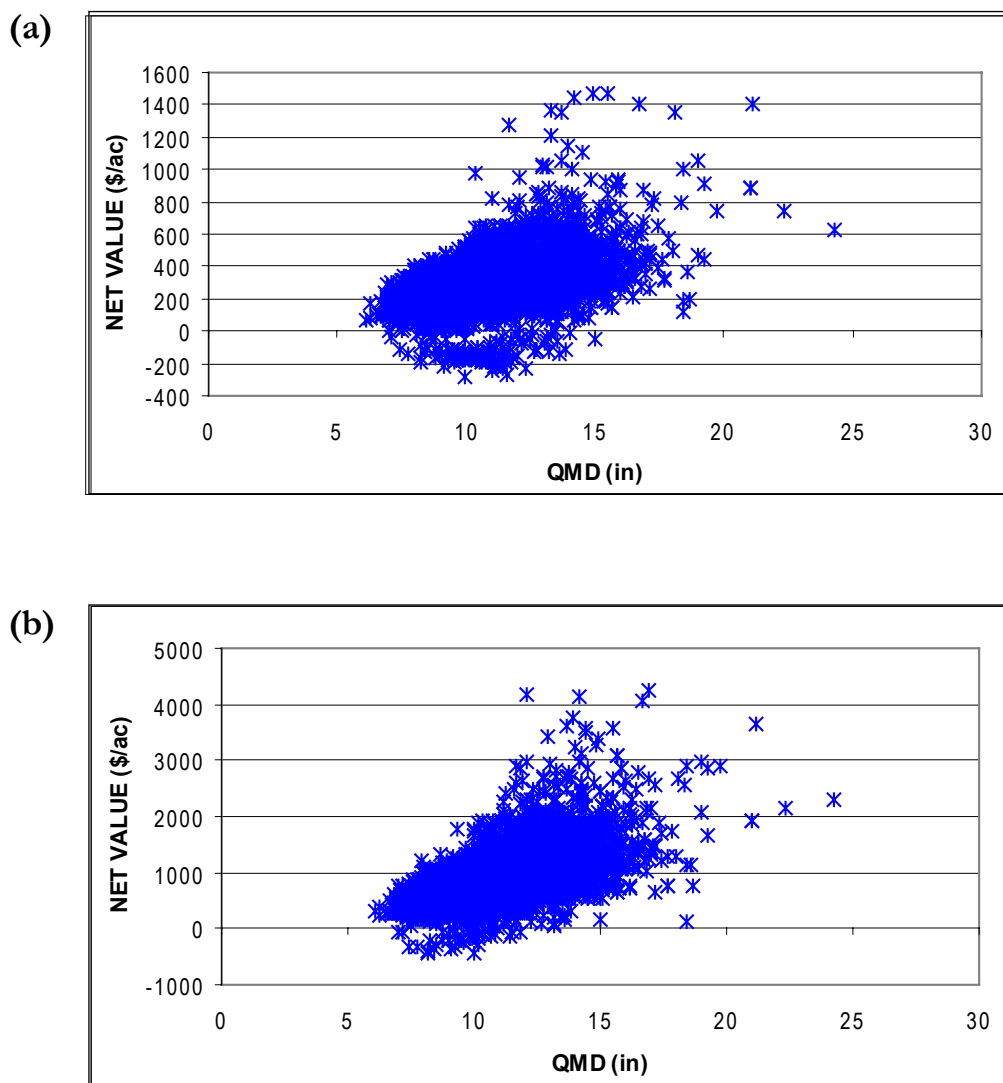


Figure 6.—Net thinning value per acre by stand quadratic mean diameter using the average pricing schedule for the smaller feller buncher by thinning intensity (30 percent[a] and 70 percent[b]) for Truck Class 1 and Road Class 1.

STUMP SPROUT DOMINANCE PROBABILITIES OF FIVE OAK SPECIES IN SOUTHERN INDIANA 20 YEARS AFTER CLEARCUT HARVESTING

Dale R. Weigel, Daniel C. Dey, and Chao-Ying Joanne Peng¹

Abstract.—Oak (*Quercus* spp.) stump sprouts are vital to sustaining oak's presence and long-term dominance when regenerating oak or mixed-hardwood forests in southern Indiana. A study was initiated on the Hoosier National Forest in southern Indiana in 1987 to predict the sprouting potential and dominance probability of oaks. Before clearcut harvesting, we sampled 2,188 trees of five oak species and measured tree age and diameter at breast height along with site index to develop sprouting and dominance probability models for subsequent follow-ups at years 1, 5, 10, 15, and 20. For the current study a dominant oak was one that had one or more competitively successful sprouts per stump 20 years after clearcutting. Two species were in the white oak group—white oak (*Q. alba* L.) and chestnut oak (*Q. prinus* L.)—and three in the red oak group—black oak (*Q. velutina* Lam.), scarlet oak (*Q. coccinea* Muenchh.), and northern red oak (*Q. rubra* L.). We used logistic regression to develop preharvest predictive models for sprouting potential and dominance probabilities as well as postharvest models for dominance probabilities of the five species. The 20-year results will be compared to results obtained 1, 5, 10, and 15 years after harvest (Weigel and Peng 2002, Weigel and others 2006). Managers can use our models to anticipate regeneration failures before harvesting or sprout success after harvest.

INTRODUCTION

Oaks are important for timber, wildlife food, and stand biodiversity. Oak regeneration continues to be a problem (Lorimer 1983, 1989). Oak advanced reproduction has been considered the main source of stems for the future forest (Sander and others 1976, 1984). One component of the future stand that is often overlooked is oak stump sprouts. The contribution of stump sprouts is overlooked in part because of the limited information about the percent of oak stumps that sprout and produce competitively successful sprouts. Thus, predicting the success or dominance of stump sprouts following overstory removal is important for understanding the role of stump sprouts in regenerating oaks.

Early research showed that parent tree age, diameter, and site quality were significant predictors of stump sprouting (Roth and Hepting 1943, McGee 1978). In Missouri, parent tree age, stump diameter, and site index were important predictors of oak stump sprouting (Johnson 1977, Dey and others 1996). In northern lower Michigan, parent tree age and stump diameter were important predictors of stump sprouting for white oak (*Quercus alba* L.) and black oak (*Q. velutina* Lam.) (Bruggink 1988). Diameter at breast height (d.b.h.) was a significant predictor for white, black, northern red (*Q. rubra* L.), and chestnut oak (*Q. prinus* L.) in

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Tennessee (Mann 1984). In Pennsylvania, species and d.b.h. were significant predictors at year 1 for white, black, northern red, and chestnut (Gould and others 2007). These prior studies were primarily concerned with short-term success of oak species in certain regions, not necessarily with the sustained dominance of sprouts in other locations (such as southern Indiana) at longer time intervals, such as 20 years.

Thus, our objectives were to determine significant predictors of oak stump sprouting in southern Indiana and to develop dominance probability models for oaks at year 20 that permit the forest manager to make an informed prediction of the approximate number of dominant or codominant oak stump sprouts in future stands. Two different types of models were developed to provide the forest manager with the ability to predict dominance probability when either preharvest or postharvest data are available.

STUDY AREAS

The study was conducted on the Hoosier National Forest in south-central Indiana. Nine stands scheduled to be clearcut were selected for measurement. There were three stands in each of three age classes: 71-90, 91-110, and 110+ years. Harvesting was done between October 1987 and May 1989. In any given year, it was not possible to determine what season (growing or dormant) individual stems were harvested, because harvesting occurred over two seasons. For a complete discussion of the study sites, measurements, model building, and data analysis, see Weigel and Peng (2002).

METHODS

MEASUREMENTS

Prior to harvest, 0.04-ha plots were established along transects in the nine stands. We inventoried and tagged 1,371 white oak, 180 chestnut oak, 399 black oak, 130 scarlet oak (*Q. coccinea* Muenchh.), and 108 northern red oak > 4.0 cm d.b.h. on the plots. Measurements included d.b.h. on all trees and heights and ages of selected trees used for site index determination. Postharvest measurements were completed 1, 5, 10, 15, and 20 years after clearcutting. First-year measurements included determining parent-tree age by counting rings on the stump surface and noting whether any sprouts were present. At years 5, 10, 15, and 20, we remeasured surviving oak stump sprouts and recorded the number of sprouts and the height of the tallest sprout. Tenth-, fifteenth-, and twentieth-year measurements included recording the crown class. Fifteenth- and twentieth-year measurements included d.b.h. of the tallest sprout.

Each stand was subdivided into smaller units that were uniform in aspect (north, 315°-135°; south, 136°-314°) and slope (ridge, upper slope, lower slope, and bottom). The mean height of the competition was computed for each of these units by averaging the heights of measured dominant or codominant competition within 1 m of selected stumps. The mean height of competition was used to determine whether a stump sprout at years 5 and 10 was competitively successful; successful competition was achieved if the sprout's height equaled or exceeded 80 percent of the mean height of the competition for the individual unit. This approach was taken because stand crown closure does not happen in the first 10 years, so the traditional concept of crown class (Smith and others 1997) is not useful in determining the social position, or competitiveness, of tree reproduction. At ages 15 and 20, oak sprout success was determined by its crown class position. This measure of sprout potential, success, or competitiveness is embodied in the concept of dominance probability (Spetich and others 2002). By year 15, crown closure had occurred; therefore, the use of crown class would provide meaningful success rates.

DATA ANALYSIS

We used logistic regression to model the dominance probability of oak stump sprouts based on the above definition of a successful, competitive, or dominant sprout, i.e., that the main sprout was at least 80 percent of the mean competition height at stand ages 5 and 10, or that the oak sprout was in the dominant or codominant crown class at ages 15 and 20. We used logistic regression because the outcome variable, success, is dichotomous and not continuous, as with the usual linear regression model.

The five-step model building approach suggested by Hosmer and Lemeshow (2000) was used. We used the maximum likelihood method implemented in PROC LOGISTIC of SAS version 9.1.3 (SAS Institute Inc., Cary, NC, 2004) to perform the logistic modeling.

Two different types of models were developed. The first type of model used preharvest measurements and therefore was not dependent on sprouting success at years 1, 5, 10, 15, or 20. The second type of model did not use preharvest data and consequently was dependent on sprouting success at years 1, 5, 10, 15, or 20.

Because the first type of model does not depend on stump sprouting status at years 1, 5, 10, 15, or 20, the model is useful when preharvest measurements can be made. Thus, probability estimates for year 20 can be obtained for the stand before harvest. The same dependent variable and independent variables for year 20 were used as in the 1-, 5-, 10-, or 15-year models (Weigel and Peng 2002, Weigel, Dey, and Peng 2006). The dependent variable was presence of a dominant or codominant stump sprout 20 years after the parent stem was harvested. The independent variables were species, parent tree age, d.b.h., natural log of d.b.h., site index, natural log of site index, and interactions between two or more of these independent variables.

Previous research emphasized developing preharvest models, which are not useful for evaluating regeneration after harvesting. To accommodate the need for predicting oak stump sprout performance after harvest, the postharvest models were developed in the present study. Postharvest models estimated dominance probabilities of stems at age 20 from stumps that had at least one live sprout at age 1, 5, 10, or 15. These models used the same dependent variable as in the preharvest models, but the number of independent variables was reduced so that only stump diameter, species, and site index were considered. These models were developed with the understanding that foresters would be examining the harvested stands at 1, 5, 10, or 15 years after harvest, and thus they would not have preharvest tree age or d.b.h. information.

The species were grouped into the white oak group and the red oak group for both types of models. The white oak group consisted of white and chestnut oaks, and the red oak group consisted of northern red, black, and scarlet oaks.

RESULTS

PREHARVEST TO AGE 20-YEAR MODELS

White Oak Group

Chestnut oak had higher dominance probabilities than white oak for a given tree age, d.b.h., and site quality (Fig. 1). For example, when age, d.b.h., and site index were held constant at 50 years, 10 cm, and 18 m, respectively, 92 percent of chestnut oak stumps are expected to produce a dominant sprout compared to only 54 percent of white oak stumps at stand age 20 years. Also, lower-quality sites (site index 18 m) had higher

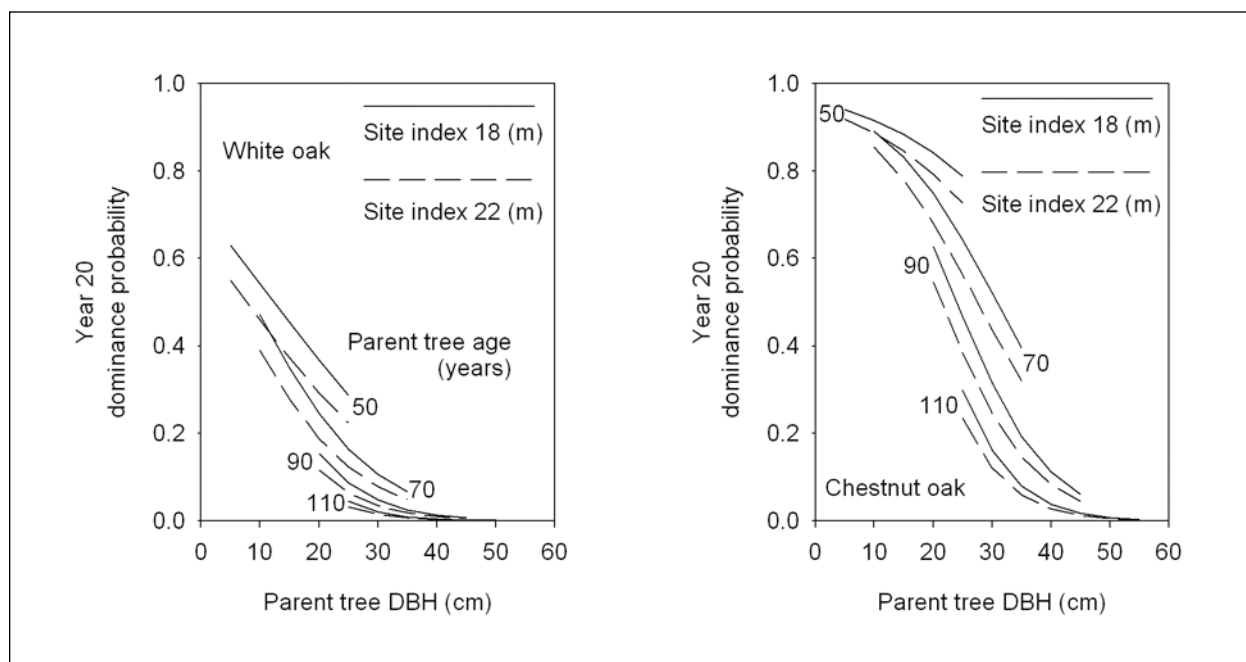


Figure 1.—Estimated dominance probability that a white oak or chestnut oak stump will produce a sprout that is either dominant or codominant 20 years after the parent tree is cut in a clearcut regeneration harvest based on parent age, d.b.h., and black oak site index.

dominance probabilities than higher-quality sites (site index 22 m). For instance, the dominance probability for white oak was 54 percent at site index 18 m, compared to 46 percent at site index 22 m. This influence of site quality on sprout dominance has been reported for 10-year- and 15-year-old stands (Weigel and Peng 2002, Weigel and others 2006). As in previous years, dominance probabilities decreased as oak trees became older and larger in diameter.

The best preharvest model (model 1 in Table 1) included four predictors: species, the interaction of parent tree age with d.b.h., site index, and the natural log of site index. The rationales for identifying the best model were explained in Weigel and Peng (2002); hence, they are not repeated here.

Table 1.—Preharvest Models: logistic regression models for estimating the probability that an oak stump sprout will be dominant or codominant at year 20.

Model No.	Species	Parameter estimates ^{a,b}				Model evaluation statistics	
		b_0	b_1	b_2	b_3	χ^2	H-L ^c
1	White oak	-72.0747	-0.00144	-1.9628	37.4688	404.7816	12.1489
	Chestnut oak	-69.8599	-0.00144	-1.9628	37.4688	($p < 0.0001$)	($p=0.1447$)
2	Red & black oaks	0.2260	-0.00060			124.9196	14.2244
	Scarlet oak	1.7408	-0.00060			($p < 0.0001$)	($p=0.2441$)

^aRegression models are of the form $P = [1 + e^{-(b_0 + b_1X_1 + b_2X_2 + b_3X_3)}]^{-1}$, where P is the estimated probability that a cut tree will produce a successful (dominant or codominant) stump sprout at age 20; X_1 = (dbh in centimeters x tree age); X_2 is black oak site index in meters (where site index is derived from Carmean and others 1989); X_3 is the natural log of site index.

^bAll parameter estimates differ significantly from zero at $p < 0.05$.

^cHosmer-Lemeshow goodness-of-fit test (Hosmer and Lemeshow 2000).

The overall significance of model 1 reached a likelihood ratio chi-square value of 404.7816, which was significant at $p < 0.0001$ with 4 degrees of freedom. The four predictors were each significant at $p < 0.05$. The goodness-of-fit of model 1 was confirmed by the insignificant Hosmer-Lemeshow (H-L) test (chi-square = 12.1489, $p = 0.1447$) (Table 1) (Hosmer and Lemeshow 2000).

The overall correct classification rate based on model 1 was 84.3 percent, which was a significant and meaningful improvement over the chance level. Model 1 was more successful in classifying stumps that did not produce a dominant or codominant stump sprout (specificity) than those that did (sensitivity). This observation was supported by the magnitude of specificity (91.9 percent), compared with that of sensitivity (45.3 percent). False positive and false negative rates were 47.7 percent and 10.5 percent, respectively.

Red Oak Group

In general, scarlet oak trees had higher stump sprout dominance probabilities than northern red oak and black oak combined (Fig. 2). Overall dominance probabilities continued to decline from year 1 (Weigel and Peng 2002) to year 20. The continued decline in dominance probability indicated that the three species were not able to compete with the surrounding vegetation. Scarlet oak had higher dominance probabilities at smaller d.b.h. and younger ages than northern red and black oak (81 percent at age 50 and 10 cm d.b.h. versus 48 percent at age 50 and 10 cm d.b.h., respectively).

The dominance probabilities for northern red oak and black oak did not differ ($p > 0.05$) at year 20. They did, however, differ from scarlet oak ($p < 0.05$). Therefore, the two species were combined in subsequent analysis. Similar to the white oak model, the best red oak year-20 model (model 2 in Table 1) included the same variables as those in the year-15 dominance probability model presented by Weigel and others (2006). Species and the interaction of parent tree age with d.b.h. were significantly related to future dominance probability in red oak group stump sprouts.

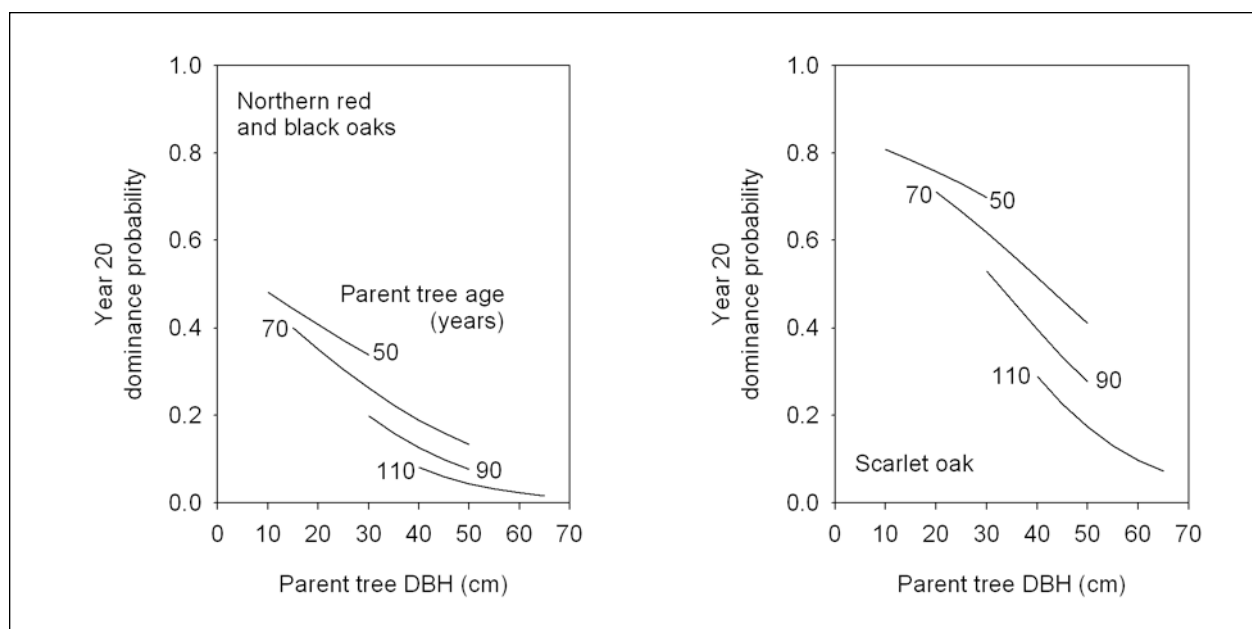


Figure 2.—Estimated dominance probability that a black oak or northern red oak, or scarlet oak, stump will produce a sprout that is either dominant or codominant 20 years after the parent tree is cut in a clearcut regeneration harvest based on parent age and d.b.h.

The overall significance of model 2 reached a likelihood ratio chi-square value of 124.9196, which is significant at $p < 0.0001$ with 2 degrees of freedom. The two predictors were each significant at $p < 0.05$. The goodness-of-fit of model 2 was confirmed by the insignificant H-L test (chi-square = 14.2244, $p = 0.0761$) (Table 1).

The overall correct classification rate based on model 2 was 65.0 percent which was a significant and meaningful improvement over the chance level. Model 2 correctly classified stumps that produced a dominant or codominant stump sprout more frequently than those that did not. This observation was supported by the magnitude of sensitivity (83.7 percent), compared with that of specificity (57.0 percent). False positive and false negative rates were 54.7 percent and 10.8 percent, respectively.

POSTHARVEST TO AGE 20 MODELS

The after-harvest models allow foresters to enter a harvested stand 1, 5, 10, or 15 years after harvest to determine the dominance probability at year 20 for those stumps that have sprouts at 1, 5, 10, or 15 years.

White Oak Group

Year 1—Oaks were more likely to produce dominant or codominant stems at year 20 on lower-quality sites than on higher-quality sites (Fig. 3). Dominance and codominance probabilities also were greater for smaller-diameter stumps. The best results were for 10-cm diameter stumps with site index of 18 m (79 percent), while the lowest dominance probability was for sprouts on 60-cm diameter stumps with site index of 22 m (6 percent). The higher-quality sites most likely had more and faster growing competition. Consequently, the oaks were unable to compete as well on higher-quality sites as on lower-quality sites.

At year 1, the success probabilities for white and chestnut oak were not statistically different, so they were combined ($p > 0.05$). The significant predictors were diameter at stump height and the interaction of site index with the natural log of site index (model 3 in Table 2).

The overall significance of model 3 reached a likelihood ratio chi-square value of 85.8272, which is significant at $p < 0.0001$ with 2 degrees of freedom. Both predictors were significant at $p < 0.05$. The insignificant H-L test (chi-square = 7.4618, $p = 0.3824$) confirmed the goodness-of-fit of model 3 (Table 2).

The overall correct classification rate based on model 3 was 66.0 percent, which was a significant and meaningful improvement over the chance level. Model 3 more correctly classified stump sprouts whose

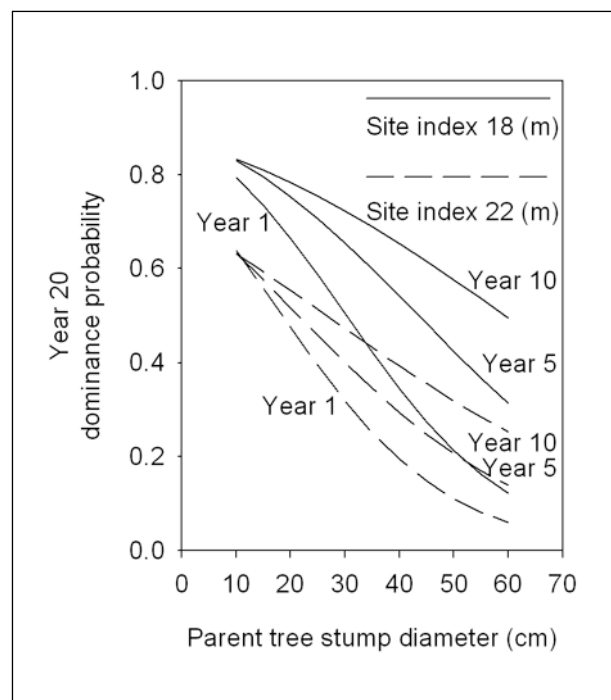


Figure 3.—Estimated dominance probability that a white or chestnut oak stump sprout that is present at year 1, 5, or 10 will produce a sprout that is either dominant or codominant 20 years after the parent tree is cut in a clearcut regeneration harvest based on parent tree stump diameter and site index.

sprouts were dominant or codominant at year 20 (sensitivity) than those that were no longer dominant or codominant at year 20 (specificity). This observation was supported by the magnitude of sensitivity (74.4 percent), compared with that of specificity (58.1 percent). False positive and false negative rates were 37.3 percent and 29.4 percent, respectively.

Year 5—As in the year-1 model, the smallest stumps on lower quality sites had the highest dominance rates (Fig. 3). The dominance probabilities decreased with increasing parent tree stump diameter and increasing site index.

Table 2.—Postharvest Models: logistic regression models for estimating the dominance probability that an oak stump sprout will be in either the dominant or codominant crown class at year 20 when sprouts were present at year 1, 5, 10, or 15 after clearcutting.

Model No.	Species and year	Parameter estimates ^{a,b}			Model evaluation statistics	
		b ₀	b ₁	b ₂	X ²	H-L ^c
3	White & chestnut oaks, 1	4.5566	-0.0664	-0.0489	85.8272 (p < 0.0001)	7.4618 (p=0.3824)
4	White & chestnut oaks, 5	5.4713	-0.0474	-0.0655	57.6083 (p < 0.0001)	9.7525 (p=0.2828)
5	White & chestnut oaks, 10	5.4129	-0.0325	-0.0669	44.9576 (p < 0.0001)	6.1552 (p=0.6299)
6	White & chestnut oaks, 15	5.7753		-0.0779	47.5109 (p < 0.0001)	10.5270 (p=0.1606)
7	Black & red oaks, 1	3.5918	-0.0321	-0.0422	63.8500 (p < 0.0001)	6.9883 (p=0.5379)
	Scarlet oak, 1	5.5106	-0.0321	-0.0422		
8	Black & red oaks, 5	4.3213		-0.0632	51.2045 (p < 0.0001)	2.4795 (p=0.9286)
	Scarlet oak, 5	5.7665		-0.0632		
9	Black & red oaks, 10	4.1130		-0.0568	41.9118 (p < 0.0001)	0.9743 (p=0.9952)
	Scarlet oak, 10	5.6756		-0.0568		
10	Black & red oaks, 15	4.2917		-0.0531	28.7678 (p < 0.0001)	1.1007 (p=0.9930)
	Scarlet oak, 15	5.9057		-0.0531		

^aRegression models are of the form $P = [1 + e^{-(b_0 + b_1X_1 + b_2X_2 + b_3X_3)}]^{-1}$, where P is the estimated dominance probability that a cut tree will produce a stump sprout that is successful (dominant or codominant crown class) at age 20: X_1 = stump diameter in centimeters 15 centimeters above ground level; X_2 is black oak site index in meters x natural log of site index (where site index is derived from Carmean and others 1989); X_3 is the natural log of site index.

^bAll parameter estimates differ significantly from zero at $p < 0.05$.

^cHosmer-Lemeshow goodness-of-fit test (Hosmer and Lemeshow 2000).

Two predictors for model 4 were significant ($p < 0.05$): diameter at stump height and the interaction of site index with the natural log of site index (model 4 in Table 2).

The overall significance of model 4 reached a likelihood ratio chi-square value of 57.6083, which is significant at $p < 0.0001$ with 2 degrees of freedom. The goodness-of-fit of model 4 was confirmed by the insignificant H-L test (chi-square = 9.7525, $p = 0.2828$) (Table 2).

The overall correct classification rate based on model 4 was 68.6 percent, which was a significant and meaningful improvement over the chance level. Model 4 more correctly classified stump sprouts that were dominant or codominant at year 20 than those that were no longer dominant or codominant at year 20. This observation was supported by the magnitude of sensitivity (78.0 percent), compared with that of specificity (54.4 percent). False positive and false negative rates were 28.0 percent and 37.8 percent, respectively.

Year 10—As with year 1 and 5 models, the smallest stumps on lower-quality sites were more likely to produce dominant or codominant stems at year 20 than stumps on higher-quality sites. The dominance or codominance probabilities at year 20 were best predicted by the presence of a stump sprout at year 10, followed by that at year 5, then that at year 1 (Fig. 3).

As with years 1 and 5, the same two predictors once again were significant in year 10 ($p < 0.05$): diameter at stump height and the interaction of site index with the natural log of site index (model 5 in Table 2).

The overall significance of model 5 reached a likelihood ratio chi-square value of 44.9576, which is significant at $p < 0.0001$ with 2 degrees of freedom. The goodness-of-fit of model 5 was confirmed by the insignificant H-L test (chi-square = 6.1552, $p = 0.6299$) (Table 2).

The overall correct classification rate based on model 5 was 68.3 percent, which was a significant and meaningful improvement over the chance level. Model 5 more correctly classified stump sprouts that were dominant or codominant at year 20 than those that were no longer dominant or codominant at year 20. This observation was supported by the magnitude of sensitivity (90.2 percent), compared with that of specificity (26.9 percent). False positive and false negative rates were 30.0 percent and 41.0 percent, respectively.

Year 15—The year-15 model was very simple with only one predictor. Similar to the other white oak group models, the year-15 model predicted higher probabilities for lower-quality sites and lower probabilities for higher-quality sites (Table 3).

Table 3.—The estimated dominance probability that a white or chestnut oak stump sprout present at year 15 will produce a sprout that is either dominant or codominant 20 years after the parent tree is cut.

Initial year and species	Site index	
	18	22
Year 15 white & chestnut oaks	0.848	0.556

Only one predictor was significant in year 15 ($p < 0.05$): the interaction of site index with the natural log of site index (model 6 in Table 2).

The overall significance of model 6 reached a likelihood ratio chi-square value of 47.5109, which is significant at $p < 0.0001$ with 1 degree of freedom. The goodness-of-fit of model 6 was confirmed by the insignificant H-L test (chi-square = 10.5270, $p = 0.1606$) (Table 2).

The overall correct classification rate based on model 6 was 75.8 percent, which was a significant and meaningful improvement over the chance level. Model 6 more correctly classified stump sprouts that were dominant or codominant at year 20 than those that were no longer dominant or codominant at year 20. This observation was supported by the magnitude of sensitivity (88.2 percent), compared with that of specificity (41.9 percent). False positive and false negative rates were 19.4 percent and 43.5 percent, respectively.

Red Oak Group

Year 1—Scarlet oak stump sprouts present at year 1 had much higher probabilities of being dominant or codominant at year 20 than did northern red or black oak (Fig. 4). Scarlet oak's probabilities ranged from 95 percent (10-cm stump diameter, 18-m site index) to 71 percent (70-cm stump diameter, 22-m site index), compared to 75 percent and 27 percent for the combined northern red and black oak over the same range. Stump sprouts on lower-quality sites had higher probabilities for dominance or codominance than those on higher-quality sites. Similar to the white oak group (Fig. 3), red oak group stump sprouts (Fig. 4) were able to compete better on the lower-quality sites than on the higher-quality sites. However, the difference between lower-quality and higher-quality sites was not as great for the red oak group as it was for the white oak group (Fig. 4 vs. Fig. 3).

Scarlet oak was significantly different ($p < 0.05$) from black and northern red oak and thus was modeled separately. The significant predictors ($p < 0.05$) besides species were: diameter at stump height and the natural log of site index (model 7 in Table 2).

The overall significance of model 7 reached a likelihood ratio chi-square value of 63.6104, which is significant at $p < 0.0001$ with 3 degrees of freedom. The goodness-of-fit of model 7 was confirmed by the insignificant H-L test (chi-square = 10.5321, $p = 0.2296$) (Table 2).

The overall correct classification rate based on model 7 was 66.0 percent, which was a significant and meaningful improvement over the chance level. Model 7 more correctly classified stump sprouts that were

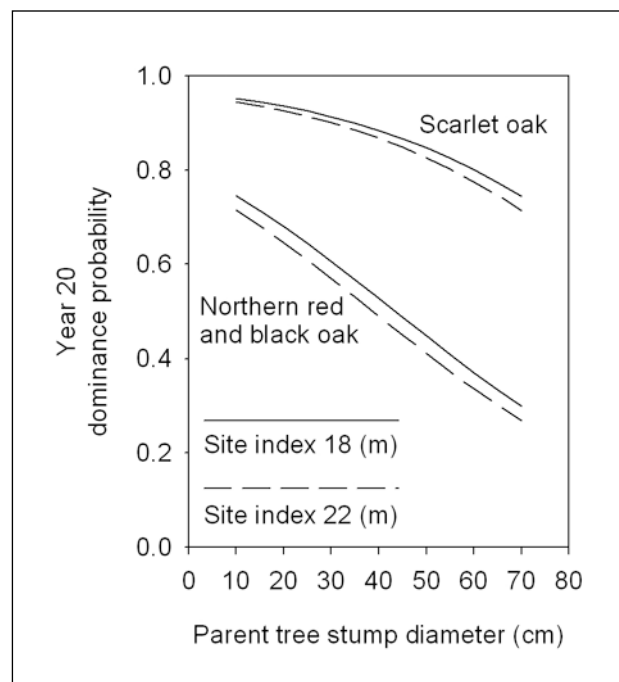


Figure 4.—Estimated dominance probability that a black oak or northern red oak, or scarlet oak, stump sprout that is present at year 1 will produce a sprout that is either dominant or codominant 20 years after the parent tree is cut in a clearcut regeneration harvest based on parent tree stump diameter and site index.

dominant or codominant at year 20 (sensitivity) than those that were no longer dominant or codominant at year 20 (specificity). This observation was supported by the magnitude of sensitivity (85.2 percent), compared with that of specificity (42.3 percent). False positive and false negative rates were 35.5 percent and 30.0 percent, respectively.

Year 5—While diameter at stump height was no longer a significant predictor, the simplified model 8 still predicted higher probabilities for scarlet oak than for northern red or black oaks combined (Table 4). And again, lower-quality sites resulted in higher probabilities for sprout's dominance or codominance than higher-quality sites.

Scarlet oak differed significantly ($p < 0.05$) from black and northern red oaks. The other significant predictor ($p < 0.05$) was the interaction of site index with the natural log of site index (model 8 in Table 2).

The overall significance of model 8 reached a likelihood ratio chi-square value of 51.2045, which is significant at $p < 0.0001$ with 2 degrees of freedom. The goodness-of-fit of model 8 was confirmed by the insignificant H-L test (chi-square = 2.4795, $p = 0.9286$) (Table 2).

The overall correct classification rate based on model 8 was 72.7 percent, which was a significant and meaningful improvement over the chance level. Model 8 more correctly classified stump sprouts that were dominant or codominant at year 20 than those that were no longer dominant or codominant at year 20. This observation was supported by the magnitude of sensitivity (87.4 percent), compared with that of specificity (44.4 percent). False positive and false negative rates were 24.9 percent and 35.3 percent, respectively.

Year 10—Similar to model 8, model 9 was very simple, predicting higher probabilities for scarlet oak than for northern red and black oaks combined (Table 4). Stump sprouts on lower-quality sites performed better than those on higher-quality sites. The probabilities for dominance or codominance increased from model 8 to model 9 for any given species and site index. This is a reasonable and predictable finding because model 9 was based on a shorter time span than model 8 (until year 20). Consequently, the predicted probability that stump sprouts would survive was higher and more accurate.

Table 4.—The estimated dominance probability that a black, red, or scarlet oak stump sprout present at year 5, 10, or 15 will produce a sprout that is either dominant or codominant 20 years after the parent tree is cut.

Initial year and species	Site index	
	18	22
Year 5		
scarlet oak	0.923	0.813
northern red & black oaks	0.738	0.506
Year 10		
scarlet oak	0.938	0.860
northern red & black oaks	0.761	0.562
Year 15		
scarlet oak	0.959	0.908
northern red & black oaks	0.822	0.664

Similar to year 1 and year 5, scarlet oak differed ($p < 0.05$) from black and northern red oak. The other significant predictor ($p < 0.05$) was the interaction of site index with the natural log of site index (model 9 in Table 2).

The overall significance of model 9 reached a likelihood ratio chi-square value of 41.9118, which is significant at $p < 0.0001$ with 2 degrees of freedom. The goodness-of-fit of model 9 was confirmed by the insignificant H-L test (chi-square = 0.9743, $p = 0.9952$) (Table 2).

The overall correct classification rate based on model 9 was 74.3 percent, which was a significant and meaningful improvement over the chance level. Model 9 more correctly classified stump sprouts that were dominant or codominant at year 20 than those that were no longer dominant or codominant at year 20. This observation was supported by the magnitude of sensitivity (87.4 percent), compared with that of specificity (43.0 percent). False positive and false negative rates were 21.3 percent and 41.4 percent, respectively.

Year 15—Like models 8 and 9, model 10 was again very simple, predicting higher probabilities for scarlet oak than northern red and black oaks combined (Table 4). Stump sprouts on lower-quality sites had higher probabilities than stump sprouts on higher-quality sites. Dominance or codominance probabilities continued to increase from year-5 and year-10 models.

As in years 1, 5, and 10, scarlet oak differed ($p < 0.05$) from black and northern red oak. The other significant predictor ($p < 0.05$) was the interaction of site index with the natural log of site index (model 10 in Table 2).

The overall significance of model 10 reached a likelihood ratio chi-square value of 28.8108, which is significant at $p < 0.0001$ with 2 degrees of freedom. The goodness-of-fit of model 10 was confirmed by the insignificant H-L test (chi-square = 1.0406, $p = 0.9941$) (Table 2).

The overall correct classification rate based on model 10 was 76.8 percent, which was a significant and meaningful improvement over the chance level. Model 10 more correctly classified stump sprouts that were dominant or codominant at year 20 than those that were no longer dominant or codominant at year 20. This observation was supported by the magnitude of sensitivity (87.4 percent), compared with that of specificity (37.3 percent). False positive and false negative rates were 16.2 percent and 55.8 percent, respectively.

DISCUSSION

Across all 10 models, stump sprouts were more successful on lower-quality sites than on higher-quality sites. Additionally, stump sprouts from younger parent trees had higher success probabilities than stump sprouts of older parent trees. In the white oak group, chestnut oak did better than white oak in the preharvest model, but in the postharvest models there was no difference. In the red oak group, however, scarlet oak always performed better in both preharvest and postharvest models than did northern red and black oaks. Site index was the most important predictor. It was present in all models except for the red oak group preharvest model. In the two preharvest models the interaction of parent tree age and d.b.h. was also an important predictor. In the postharvest models stump diameter was an important predictor for the white oak group while species was an important predictor for the red oak group.

The 10 models presented in this paper are valuable for predicting the contribution of stump sprouts to forest regeneration. The models allow forest managers to predict the percent of competitive oak stump sprouts 20 years after an even-aged timber harvest. Models 1 and 2 can be used to predict the likelihood of dominant and codominant stump sprouts 20 years after clearcut harvest based on preharvest information. These models also permit forest managers to assess the contribution of stump sprouts to the desired stocking of oak advanced reproduction and to adjust stand prescriptions to promote oak advance reproduction by reducing the vigor and abundance of major woody competitors.

The remaining eight models, models 3 to 10, predict the likelihood of dominant and codominant stump sprouts at year 20 based on their presence at year 1, 5, 10, or 15. Forest managers are then able to assess the need for crop tree release or another type of precommercial thinning to maintain desired stocking of oak. Forest modelers can use these models to better predict and describe the influence of oak stump sprouts on future stands and stand stocking.

Our study differs from many other stump sprout studies by using logistic regression to predict the contribution of stump sprouts to the future stand and hence the sustainability of oak in that stand. Our model incorporates data regarding whether a stump produces sprouts, whether those sprouts survive and grow, and how competitive these sprouts are relative to competing vegetation. Another unique quality of this study is that it provides a long-term understanding of stump sprouts. We examined the fate of oak stump sprouts at age 20, when crown closure and differentiation are occurring, providing a better indication of the reproduction assuming dominance.

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AN EVALUATION OF THREE GROWTH AND YIELD SIMULATORS FOR EVEN-AGED HARDWOOD FORESTS OF THE MID-APPALACHIAN REGION

John R. Brooks and Gary W. Miller¹

Abstract.—Data from even-aged hardwood stands in four ecoregions across the mid-Appalachian region were used to test projection accuracy for three available growth and yield software systems: SILVAH, the Forest Vegetation Simulator, and the Stand Damage Model. Average root mean squared error (RMSE) ranged from 20 to 140 percent of actual trees per acre while RMSE ranged from 2 to 65 percent of actual basal area per acre (BAAC), depending on thinning category and dataset. Overall, SILVAH produced the smallest average error across all datasets and thinning categories, although significant improvement in projection accuracy is deemed feasible.

INTRODUCTION

In the late 1960s Schlaegel (unpublished) developed cubic foot and basal area growth and yield models for even-aged yellow-poplar (*Liriodendron tulipifera* L.) stands in West Virginia. These models were based on the concept of the mathematical compatibility of growth and yield as developed by Clutter (1963). Perkey (1985) published an adaptation of Martin E. Dale's FORTRAN GROAK program in MBASIC. Perkey's (1985) whole stand model predicted future basal area per acre (BAAC), trees per acre (TPA), quadratic mean diameter, percent stocking, cubic foot yield, and board foot yields at some future age. The equations were based on a thinning study in even-aged white oak and black oak stands in Iowa, Missouri, southern Ohio, and southeastern Kentucky (Dale 1972). Although whole stand models can provide quantitative information on future estimates of TPA, BAAC, and volume per acre at the stand level, the need for tree-level information within those stands is obvious when such characteristics as product quality and value are being considered.

During this period, development of growth and yield systems in the southern United States took the diameter distribution approach to disaggregating stand-level data through the use of the Weibull distribution function, as developed by Bailey and Dell (1973). The approach first employed in the Central Appalachian region followed a distance-independent individual tree growth and yield system. Hilt (1985) developed such a system (OAKSIM) for even-aged upland oak stands in southern Ohio and southeastern Kentucky. This growth and yield system was based on a thinning study in white, black, and scarlet oak stands typical of this region and the computer program OAKSIM was developed in FORTRAN for use on mainframe computers. Although this system was incorporated in the NED SIPS (Stand Inventory Processor and Simulator) program, the DOS platform was not consistently stable and so no further analysis of this system was attempted. Yandle and others (1987) developed whole stand growth models for even-aged hardwood stands based on remeasurement data collected in east central Ohio and north central West Virginia. The system provides basal area and volume

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projection models that are based on current stand parameters of percent basal area of shade-intolerant species, percent basal area of shade-tolerant species, merchantable basal area, site index, number of merchantable trees, stand age, quadratic mean diameter of tolerant species, and the square of current mean diameter. Although these models were based on a limited dataset, fairly accurate estimates were reported.

During this same period, The Woodsman's Ideal Growth Projection System (TWIGS) was developed starting with the Lake States version (LS TWIGS) in 1982, followed by the release of the Central States version (CS TWIGS) in 1987 and the Northeast version (NE TWIGS) in 1989 (Hilt and Teck 1989). The NE TWIGS computer software was a distance-independent individual tree growth and yield system written in FORTRAN for the DOS operating system having the same functional structure as the LS and CS TWIGS versions. Individual tree models were fitted to data collected on U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis plots from the 14 northeastern states, including West Virginia. Numerous organizations modified this system to represent more localized conditions. This system later became the basis for the northeast variant of the U.S. Forest Service Forest Vegetation Simulator (FVS), which was initially developed in 1991 and released in 1993 (Bush 1995).

In 1992, the SILVAH software system was released (Marquis and Ernst 1992). This software system is more of an expert system designed to manage hardwood forests of the Allegheny Plateau region. As part of this expert system, SILVAH includes a stand growth simulator. Unlike the individual tree models discussed thus far, this system employs a modified stand table projection algorithm. Although this system includes a diameter growth algorithm for oak species from OAKSIM, the overall growth projection and mortality systems are significantly different from the original OAKSIM program. The last system to enter the hardwood growth and yield platform was the Stand Damage Model (SDM), which was released in 1995. This system is a distance-independent tree-growth simulator based on gap modeling theory (Colbert and Racine 1995, Colbert and Sheehan 1995, Racine and Colbert 2004).

Within the past 5 years, Brooks and Wiant (2004, 2006) have developed stand-level cubic and board foot volume prediction models for northern West Virginia based on stand basal area and average dominant height. In addition, a basal area projection model for this same area was published by Brooks and others (2008). This algebraic difference equation model predicts future basal area based on initial basal area and changes in trees per acre and stand age. This model was originally applied to a slash pine growth and yield system in the Southeast (Pienaar and Shiver 1986).

The purpose of this report is to review those growth and yield systems specific to hardwood stands in the mid-Appalachian region, to identify and review those systems available in computer program format, and to test existing systems on actual remeasured plot data from this region.

METHODS

The Central Appalachian mountain region is made up of several ecoregions with distinctly different soils, length of growing season, annual precipitation, elevation, and species distributions and thus growth rates. Height-diameter and volume relationships have been shown to be different among these regions (Brooks and Wiant 2007, 2008).

The datasets selected for study represented long-term remeasurement data from the unglaciated Allegheny Plateau (Allegheny National Forest Data), the Western Allegheny Plateau and Mountains (OAKSIM Data), the Southern Appalachian Ridge and Valley (MeadWestvaco Data), and Eastern Allegheny Plateau and Mountains (WVU Hatch Data) (Fig. 1). These even-aged datasets were selected because they were available, had a thinning component, and provided an opportunity to evaluate growth and yield among the different ecoregions. Only plot data that spanned three measurement periods were included in this analysis. For thinned treatments, only growth following initial thinning but prior to a second thinning was included. Since these and other datasets evaluated for this study did not include height measurements at each of the remeasurement periods, differences in volume were not evaluated. Analysis was based on TPA and BAAC. These datasets included trees greater than or equal to the 1-inch diameter class.

The four long-term datasets provided 125 permanent plots and 1,401 actual remeasurement periods. Since at least some of the data were based on a 5-year remeasurement cycle, this analysis is based on 5-year projection intervals from 5 to 30 years. A maximum of 100 growth projection comparisons resulted. Plot-level summary statistics for each dataset are described in Table 1.

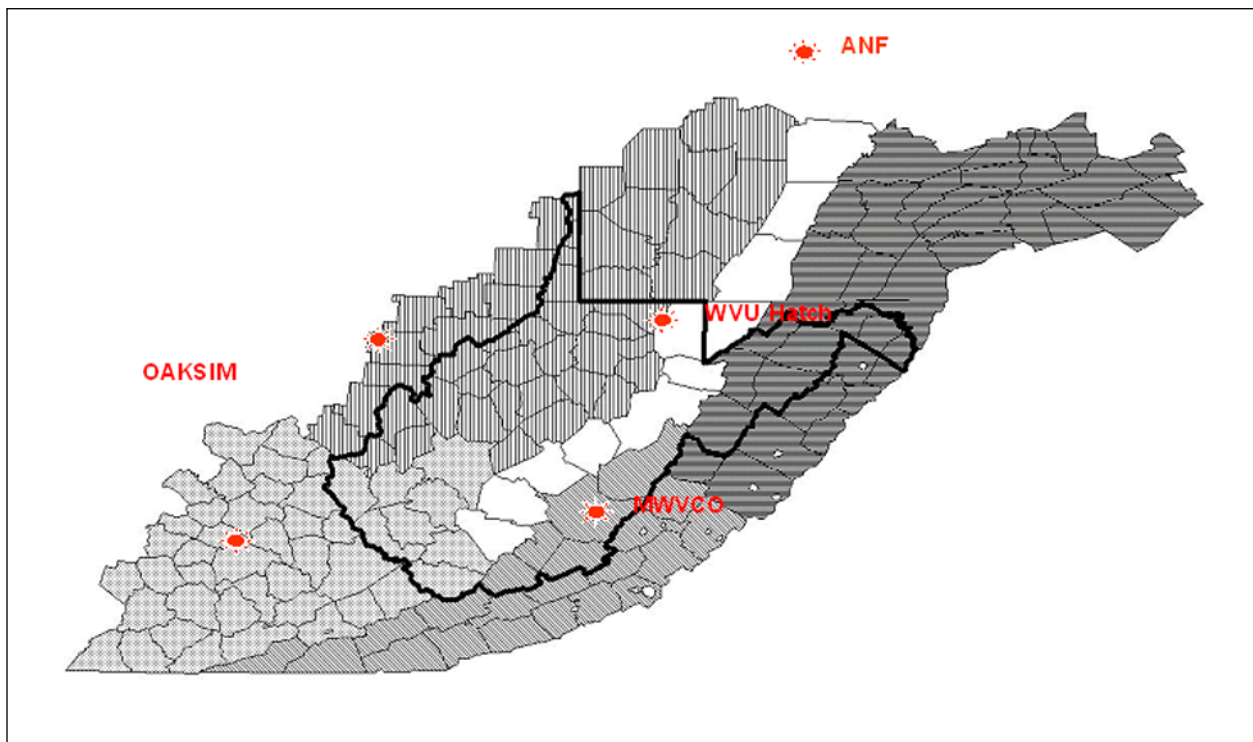


Figure 1.—Location of sample datasets by ecoregion.

Table 1.—Plot-level summary statistics for trees per acre and basal area per acre by dataset.

	Min TPA	Max TPA	Min BAAC (ft ² /ac)	Max BAAC (ft ² /ac)	Initial Stand Age (yrs)	No. Sample Plots
ANF	92	997	45.2	181	53 - 54	35
OAKSIM	50	798	15.1	116.5	20 - 93	53
MWVCO	200	2,218	10.3	75.6	10 - 16	20
WVU Hatch	374	1,026	60.2	115.3	20 - 24	17

Tree data files for each plot were summarized for BAAC and TPA in each remeasurement period. For thinned stands, the plot-level summary was based on post-thinning conditions and continued until the remeasurement period prior to the second thinning. Initial tree lists for each plot were entered into each growth simulator and projected for 30 years or for the longest remeasurement period available for that particular plot, in 5-year intervals. In some instances, actual remeasurement periods did not fall exactly in 5-year intervals and in these situations a linear interpolation was used between adjacent measurement intervals. For each growth projection interval the average error (bias) and root mean squared error (RMSE) for TPA and BAAC were calculated based on observed measured data and the model prediction. These simple sample statistics were used for their ease of interpretation. Average error (bias) was used in situations where predictions either continuously over- or underestimated the stand parameters, as well as in cases where both situations occurred during the growth projection period. In these situations, the algebraic sign of the bias may change during the projection period. The RMSE provides an estimate of average error unaffected by changes in the algebraic sign of the estimation. As mentioned previously, no attempt to compare volume estimates was made due to the general lack of height data to obtain actual volume estimates. Summary statistics were generated by the growth and yield systems separately for each dataset and thus ecoregion. In all cases, only the original tree list was used in computer simulations with no simulated ingrowth.

The three available growth and yield computer systems that were selected for testing were: SILVAH (version 4.04, DOS), FVS (Northeast variant, Windows version 6.21, release 07.05.06) and the SDM (version 2.0, DOS). All simulators were tested based on an “out-of-the-box” condition using all default settings, with the exception of the Northeast Variant of FVS, where the most appropriate national forest location was employed. To evaluate model projection error, RMSE was plotted as the percent of actual TPA and actual BAAC. Model projection length was limited to the range of the actual measurement periods in each dataset. Averages for thinned (T) and unthinned (NT) plots (stands) were plotted separately and presented separately by dataset and thus ecoregion.

RESULTS

PROJECTION ACCURACY

Evaluation of the performance is solely the subjective evaluation by the authors, but it is based on preliminary whole stand model development in these same regions.

Allegheny National Forest (ANF) Data

The ANF data were projected with each simulator for three 5-year growth intervals. SILVAH and FVS overestimated TPA, while the SDM underestimated TPA for these stands. In general, average TPA prediction error increased with increasing projection length, a commonly found scenario for growth and yield systems. Thinned plots exhibited smaller average prediction error than their unthinned counterparts.

Average TPA error (RMSE) at the end of the 15-year projection period ranged from 20 to 30 percent of actual stand density. The smallest error was associated with thinned stands when projected using the SILVAH simulator.

SILVAH and FVS underestimated BAAC for the thinned stands and overestimated BAAC for unthinned stands. The SDM underestimated BAAC for both thinning categories; the largest errors were associated with the unthinned stands. For BAAC, the SDM exhibited average prediction errors above 25 percent of actual

BAAC within a 5-year projection period and exceeded 35 percent by the end of the 15-year projection. Both SILVAH and FVS performed well with prediction errors less than 15 percent of the actual BAAC over the range of projection years; their performance was slightly better performance for unthinned stands.

OAKSIM Data

The OAKSIM data were projected using each simulator for six 5-year growth intervals. SILVAH, FVS, and SDM all overestimated TPA. In general, average bias for thinned stands exceeded that of unthinned stands, although these differences were small for the SDM. Average TPA error (RMSE) at the end of the 30-year projection period ranged from 60 to more than 140 percent of actual stand density; the smallest errors were associated with the SILVAH and SDM (unthinned stand only) simulators. Initial differences between thinning category were not apparent, but the unthinned stands appear to have smaller prediction errors than their thinned counterparts.

SILVAH and SDM underestimated BAAC for thinned and unthinned stands with the unthinned stands having a slightly larger average error, although the magnitude of these differences was small, especially for SDM. For the FVS simulator, BAAC was overestimated for both thinned and unthinned stands, but these differences were small. Both the SDM and FVS simulators exhibited average prediction errors above 25 percent of actual BAAC by the end of the 30-year projection period. SILVAH performed well with prediction errors less than 15 percent of actual BAAC across the entire 30-year projection period for both thinned and unthinned stands; the thinned stands performed slightly better than the unthinned stands.

MeadWestvaco Data (MWVCO)

The MeadWestvaco data were projected with each simulator for two 5-year growth intervals. All three simulators overestimated TPA. RMSE projection errors for TPA were generally lower for the thinned stands than for their unthinned counterparts. Maximum error was associated with the FVS simulator, which overestimated TPA in excess of 140 percent of actual TPA in the 10-year projection period.

All simulators underestimated BAAC with the unthinned stands having smaller average errors than their thinned counterparts. The smallest errors were associated with the SILVAH simulator for unthinned stands and the largest errors were associated with unthinned stands projected with the FVS simulator.

WVU Hatch Data

The WVU Hatch data were projected with each simulator for six 5-year growth intervals. SILVAH and FVS overestimated TPA while SDM underestimated TPA for these stands. One exception to this pattern was for thinned stands using the SILVAH simulator, where initial error (average bias) was positive and then turned negative from projection age 20 through 30. Generally, thinned plots exhibited smaller average prediction errors than their unthinned counterparts. SILVAH underestimated BAAC for unthinned stands and initially overestimated then underestimated BAAC for thinned stands. The magnitude of these differences was small. For the FVS simulator, BAAC was underestimated for both thinned and unthinned stands; the unthinned stands exhibited a slightly larger average error. The SDM underestimated BAAC for both thinning categories, with the largest errors associated with the unthinned stands. Average TPA error (RMSE) at the end of the 30-year projection period ranged from 20 to 140 percent of actual stand density; the smallest error was associated with SILVAH and the SDM simulators. Although some initial differences were apparent, little practical difference exists between thinning categories at the end of the projection period. SDM (unthinned

stands only) and FVS simulators exhibited average prediction errors above 12 percent of actual BAAC within the 5-year projection period and exceeded 25 percent by the end of the 30-year projection. Both SILVAH and the SDM (thinned stands only) performed well, with prediction errors less than 15 percent across the entire 30-year projection period.

PROJECTION ACCURACY BY ECOREGION

To evaluate model projection accuracy by ecoregion, RMSE as a percent of actual TPA and BAAC were plotted for both thinning categories for all datasets, separately by simulator. Evaluation of the performance is solely the subjective evaluation by the authors, but it is based on preliminary whole stand model development in these same regions.

SILVAH

In terms of RMSE for TPA, SILVAH performed better for the ANF and WVU Hatch datasets (Fig. 2). For the OAKSIM dataset, average projection error was nearly twice that for the WVU Hatch dataset when projected for 30 years. The largest error was associated with the unthinned stands for the younger MWVCO dataset, but this result may be more a function of stand age than of ecoregion. For BAAC, all projections were considered good with few visible differences by ecoregion, except for the thinned MWVCO stands (Fig. 3).

FVS

In terms of RMSE for TPA, FVS performed poorly for all datasets except the thinned stands from the MWVCO and ANF datasets (Fig. 4). In terms of average RMSE for BAAC, FVS performed poorly for all datasets except those from the ANF dataset (Fig. 5). Again, the larger error associated with the MWVCO dataset may be more a function of stand age.

SDM

In terms of RMSE for TPA, the SDM performed best for both thinned and unthinned stands for the WVU Hatch dataset (Fig. 6). The poorest long-term projections were associated with the stands from the OAKSIM dataset. In terms of average RMSE for BAAC, the SDM performed poorly for the OAKSIM, ANF, and MWVCO datasets (Fig. 7). The best predictions were associated with the WVU Hatch data, perhaps because baseline models were fit to data from the same area.

CONCLUSIONS

Datasets from four ecoregions across the mid-Appalachian region were used to test projection accuracy for three available growth and yield software systems, SILVAH, FVS, and the SDM. Average prediction error (RMSE) generally ranged from 20 to 140 percent of actual TPA, depending on thinning category and dataset. Average prediction error (RMSE) ranged from 2 to 65 percent of actual BAAC, depending on thinning category and dataset. Preliminary investigation by Brooks and Wiant (2004, 2006), using very simple model forms, indicate that whole stand estimates of volume within ± 10 to 15 percent of actual stand volume seem feasible in these Appalachian hardwood stands. It is recognized that these volume prediction errors were based on the same dataset as model fitting, thus reducing prediction variability. However, whole stand estimates in excess of 20 percent of actual stand values seem unacceptably high and a reduction in prediction error appears attainable for this region. Using a more complex basal area projection model, Brooks and others (2008) were able to provide basal area predictions within 10 percent of actual stand basal area for undisturbed

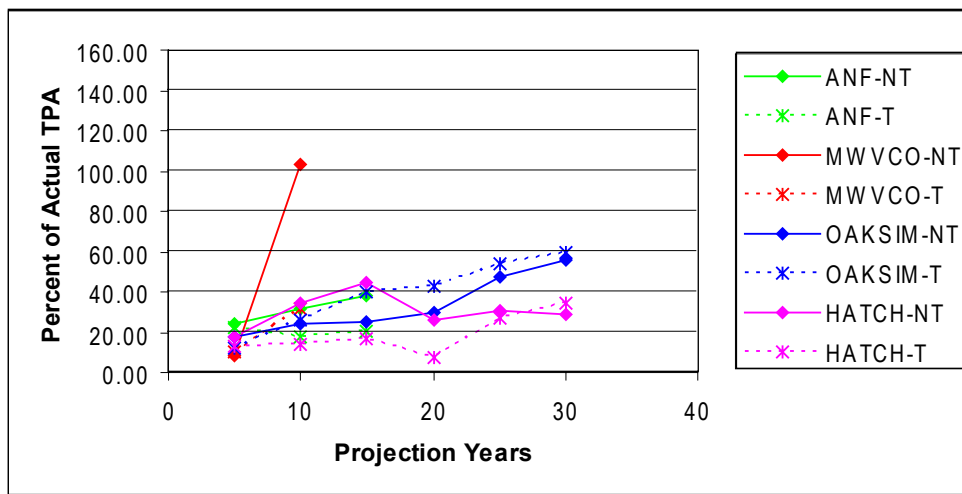


Figure 2.—SILVAH RMSE as a percent of actual TPA by projection year for all datasets.

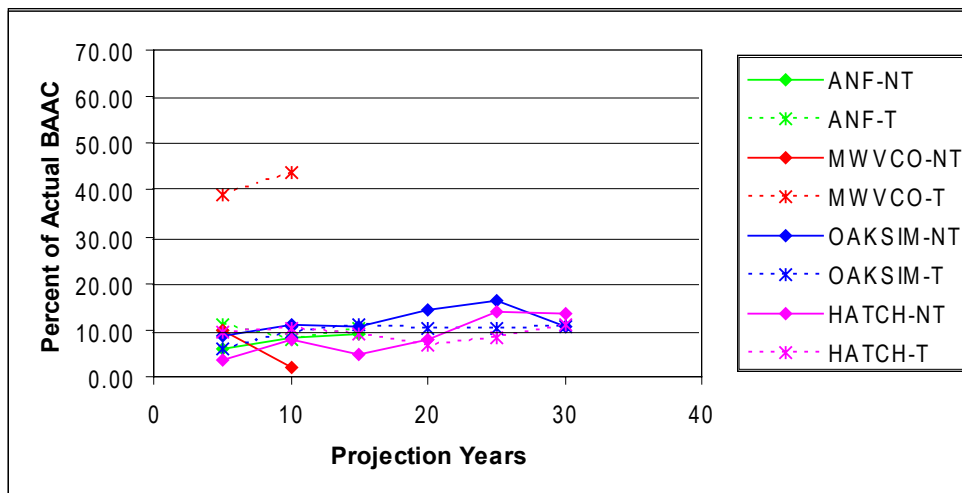


Figure 3.—SILVAH RMSE as a percent of actual BAAC by projection year for all datasets.

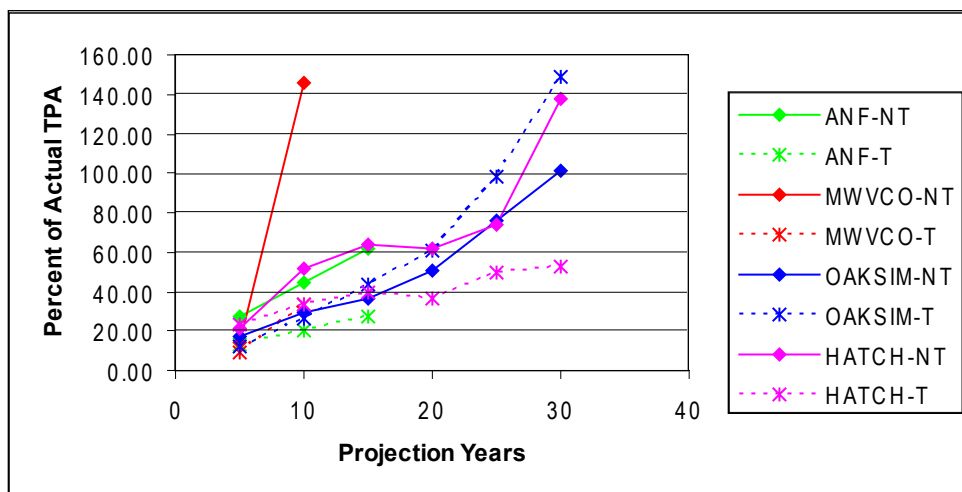


Figure 4.—FVS RMSE as a percent of actual TPA by projection year for all datasets.

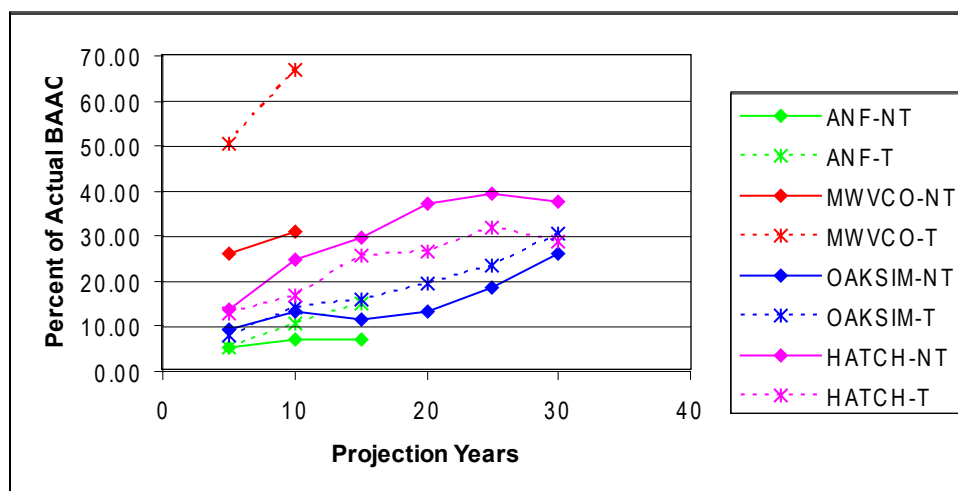


Figure 5.—FVS RMSE as a percent of actual BAAC by projection year for all datasets.

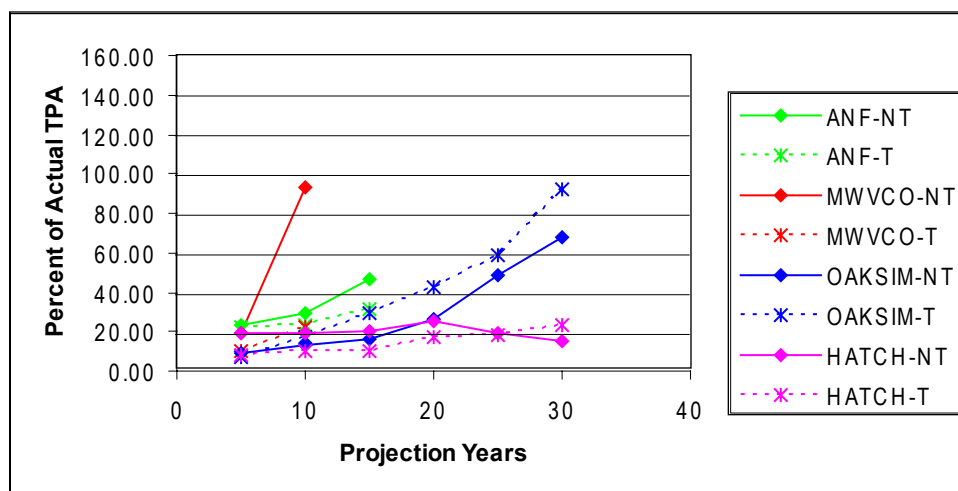


Figure 6.—SDM RMSE as a percent of actual TPA by projection year for all datasets.

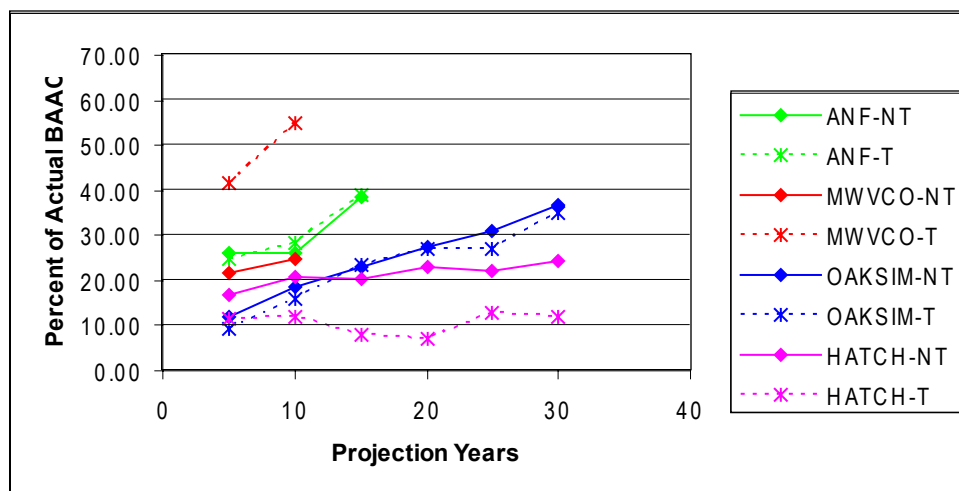


Figure 7.—SDM RMSE as a percent of actual BAAC by projection year for all datasets.

even-aged stands from this region. In addition, the authors believe that the increasing prediction error associated with increasing projection length can be controlled, though not eliminated. Overall, SILVAH produced the smallest average error across all datasets and thinning categories, although significant improvement in projection accuracy is deemed feasible. All results are based on remeasurement data from even-aged stands and should not be extrapolated to selectively harvested conditions. In terms of total basal area prediction, Schuler and others (1993) recommended SILVAH and NE-TWIGS for Allegheny hardwoods and oak-hickory stands in their earlier evaluation of hardwood growth and yield systems. The poorer performance of FVS in this project may indicate that changes to model structure or parameters of the original NE-TWIGS system negatively influenced projection accuracy in these datasets.

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A TECHNIQUE FOR PREDICTING CLEAR-WOOD PRODUCTION IN HARDWOOD STEMS: A MODEL FOR EVALUATING HARDWOOD PLANTATION DEVELOPMENT AND MANAGEMENT

Christopher M. Oswalt, Wayne K. Clatterbuck, and Dave R. Larsen¹

Abstract.—The management of artificial hardwood stands suffers from a paucity of information. As a result, many managers and scientists turn to conventional pine plantation management as a source for informing silvicultural decisions. Such an approach when managing hardwoods ignores the development occurring in natural hardwood stands, which produce stems prized for their growth and form. Contrary to a volumetric focus, stem quality is extremely important in the valuation of hardwood stands. The monetary value of a stand is directly related to the quality of individual stems within that stand. Growing a hardwood bole that is clear of branches, knots, and defects, or growing “clear wood,” can significantly increase the return on a hardwood plantation investment. We investigated an approach to forecast the impact of hardwood plantation establishment and management decisions on the production of “clear wood.” A model was developed that predicts, through a series of simultaneous equations, the growth and development of first-order branches and subsequent branch occlusion. Our investigation focused on cherrybark oak (*Quercus pagoda*), an economically valuable southern bottomland species. The resultant model provides the user with output regarding stem quality. Users then can evaluate the impact of decisions such as initial spacing, species composition, and plantation arrangement on hardwood log quality. This approach may be expanded to other economically important species. Therefore, we briefly discuss the data requirements for expanding this approach to other species or species groups. In addition, we explore some of the possible uses for this tool in future research and management planning.

INTRODUCTION

Multiple challenges continue to plague silviculturists’ attempts to naturally regenerate and promote the development of high-quality hardwoods such as oak (*Quercus* spp.) (Loftis and McGee 1993, Oswalt and others 2006). Many landowners and foresters turn to artificial regeneration techniques to satisfy oak establishment objectives and to meet demand for hardwood forest products; accordingly, hardwood plantings apparently have increased. In addition, governmental cost-share programs such as the Wetland Reserve Program and the Conservation Reserve Program are contributing to the increase in hardwood planting and hardwood plantation establishment (King and Keeland 1999, Devall and others 2001, Gardiner and others 2004). Consequently, increasing attention and efforts are being directed toward the development of prescriptions for artificially regenerated hardwood stands (Lockhart and others 2006).

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Artificial hardwood stand management suffers from a lack of credible scientific information. The myriad of different species with different site requirements and growth habits makes establishing and managing hardwood plantations more complex than with single-species pine (*Pinus* spp.) plantations. The importance of stem quality introduces an additional complicating variable. Numerous forest growth and yield models have been developed for modeling pure and mixed stands (Peng 2000, Porté and Bartelink 2002). Generally, the focus of these models is volumetric in nature; that is, they forecast tree or stand level metrics such as volume/unit-area, diameter distributions, stem density/unit-area, or basal area/unit-area, metrics from which individual stem quality cannot be quantified. Few models have focused on forecasting measures of stem quality. Simple linear (Ernst and Marquis 1979, Myers and others 1986) and nonlinear regression (Dale and Brisbin 1985) and discriminant function (Lyon and Reed 1987, Belli and others 1993) equations have been used to predict tree or log grades and/or to predict tree/log grade distributions. None of the solutions has been ideal due to the focus on subjective discrete classification variables. In addition, these models often assume no change in stem defect over time; they assume only changes in log or tree diameter. As a result, these models are limited in usefulness.

To develop a model without the constraints of traditional log/tree grading systems, we investigated an approach to forecast the impact of hardwood plantation establishment and management decisions on the production of “clear wood.” Through a series of simultaneous equations, our model predicts the growth and development of first-order branches and subsequent branch occlusion. Our investigation focused on cherrybark oak (*Quercus pagoda* Raf.), an economically valuable southern bottomland hardwood. In contrast to most stand and individual tree scale models, the resultant model provides the user with output regarding stem quality. The development of a model to predict specific tree or log grades was not the objective of this work. Rather, the purpose was to produce a model that can be used as a decision support tool. Model-users have the capability to evaluate the impact of decisions such as initial spacing, species composition, and plantation arrangement on hardwood log quality.

MODELING APPROACH

The primary objective of this work is to develop a model that can be used to evaluate the impacts of various silvicultural decisions on the development of stem quality. The overall problem has been dissected into three issues: 1) predicting the amount of overwood that is necessary for dead branch occlusion through subsequent diameter growth; 2) predicting branch population dynamics within the live crown; and 3) incorporating 1 and 2 into a model for evaluating the effects of various silvicultural decisions on the development of clear wood. Initially, the effort is to predict either a positive or negative impact on clear-wood production through altering stand level parameters impacted by different management decisions.

The intention of the modeling activity is to develop a model of the crown system dynamics of bottomland oaks that, when incorporated within an individual tree growth model, can be used to predict branch and knot characteristics with enough accuracy to evaluate potential tree or log quality. Additionally, we attempt to develop the model in such a way that will allow relatively easy interaction between land managers and the model while allowing a certain degree of flexibility that allows land managers to adapt the model to local conditions.

Several key points in the objectives statement warrant elaboration. First, *crown system dynamics* refers to the birth, growth, development, and death of first-order branches that make up the living crown of the modeled tree. The model, therefore, tracks the conception of first-order branches within the crown, subsequent branch

growth and branch death as crown rise occurs due to competition. Crown characteristics, such as crown surface area, crown volume, crown diameter, and crown shape are calculated and used as metrics to compare model results with other modeling efforts and empirical data.

Second, bottomland oaks refers to species from the genus *Quercus*, in particular cherrybark oak. This modeling activity is in pursuit of a mixed-species model that can accommodate natural stands. Yet, it is important to note that the immediate objective is to develop a model that will accurately reflect the stem-quality development of monospecific plantations and planted two-species mixtures. Upon satisfactory development of such a model, the additional information needed to model the spatial and competitive variation found in natural stands will be incorporated.

Easy interaction is imperative to allow natural resource managers with very little or no modeling experience access to the predictive powers of the developed model. It is extremely important that resource managers be able to use the model as a tool for an additional level of sophistication in their planning process. Finally, the ability for the model to be *adapted* to reflect local conditions, through the use of local knowledge of site indices, is essential.

BRANCH OCCLUSION MODEL

As gross crown dimensions are proportional to and determinants of tree growth (Assman 1970, Rennolls 1994), the number and size of branches within the crown are major determinants of stem structure and, therefore, wood quality. Wood quality is heavily affected by the development of first-order branches within the crown, particularly the self-pruning and subsequent occlusion of branches as crown recession occurs (Makinen 1998, Makinen and Colin 1998, Makinen and Makela 2003).

A logical first step is to evaluate the effects of variable branch sizes on the stem diameter needed for branch occlusion through diameter growth following crown recession, but little is known regarding this relationship. The knowledge gap is particularly large for hardwoods, including highly valuable species like cherrybark oak. Models combining growth and development of stem structure, including internal characteristics, are in development (Maguire and others 1994, Makela and Makinen 2003). However, researchers have focused primarily on conifer species (e.g., Norway spruce [*Picea abies* (L.) Karst], Scots pine [*Pinus sylvestris* L.], and loblolly pine [*Pinus taeda* L.]).

Data were obtained by mapping branch-knots from 21 boards strategically sawn from three logs of one cherrybark oak. Each log end was divided into quarters and marked for reassembly following sawing. All boards were flat sawn in the field using a Wood-Mizer (Wood-Mizer Products Inc., Indianapolis, IN) portable band saw with a 2-mm kerf. The first cut for each log followed the log pith. Boards were carried to the laboratory and the logs were reassembled. Distance from the pith to each board face was recorded. Mean sawn board thickness was 2.82 cm with a range of 2.3 to 4.6 cm.

Branch-knots were numbered and mapped along three axes according to board-face location, height from base of tree, and distance from the centerline of each board (board centerline corresponded to initial quarter lines drawn for reassembly). In addition, branch-knot diameter was recorded at each location. Branch-knots retained a unique identifier among sequential boards to chart the development of each branch. For each branch, maximum diameter and stem radius at the point where a branch no longer appeared were used for development of a simple predictive equation.

CROWN ALLOMETRY MODEL

An understanding of the growth dynamics of individual first-order branches within the developing crown is integral to forecasting stem quality. Similar to the capacity to forecast branch occlusion, it is essential to have the capability of predicting not only the number of branches, but also the spatial distribution and size of branches within the live crown. It is particularly important to be able to describe the population of branches at the base of the live crown. Branches at the base of the live crown are commonly the largest branches within the crown and therefore have the greatest potential for impacting future stem quality (Oswalt and others 2006).

The goal for the crown allometry model was to produce a series of equations in which the dynamics of first-order branches within the active crown, with special focus on the branch population at the base of the live crown, could be predicted. Predictor variables of interest were gross crown metrics that could be derived from either individual tree models or common inventory data. Predictor variables were crown width (CW), crown length (CL) (referred to as “crown depth” in the case of branches not at crown base), and the base of the live crown (CB).

Data for developing equations to describe the allometry of cherrybark oak crowns were obtained from 50 trees located in four separate stands across Tennessee. Three stands were located on the Natchez Trace State Forest (NTSF) in western Tennessee (latitude: 35.781° N; longitude: 88.360° W) and one stand was located on privately owned land in Dixon Springs, TN (latitude: 36.358° N; longitude: 86.050° W). Diameter at breast height (d.b.h.) varied from 8 to 51 cm. Height varied from 6 to 30 m and stand age varied from 14 to 50 years. Before felling, total height, d.b.h., and crown metrics were collected for each tree. Crown metrics consisted of crown length (height to base of live crown subtracted from total tree height), multiple crown radii ($n = 5$ to 18), and height to the widest part of the live crown. All first-order branches on each felled tree were mapped according to insertion height, departure angle, and departure quadrant. Departure quadrant, which corresponded to a four-quadrant polar coordinate grid, was derived from the quadrant in which the majority of the branch was located. In addition, branch basal diameter and branch length were recorded for each branch. Mean crown width was calculated from crown radii observations. A system of equations was developed that allowed prediction of individual branch metrics.

SYLVAN FRAMEWORK

In the late 1980s an approach to modeling forest growth, based on the three-dimensional crown characteristics of trees and stand dynamics principals to predict tree growth and development, was developed by Larsen (1991a, 1991b, 1994). This approach produced the Sylvan Stand Structure Model. Offering great flexibility, this model is dependent on individual tree distance and was designed to derive model parameters from the stand in question. The Sylvan model has successfully been applied to a wide variety of species and used to predict the development of Douglas-fir/western hemlock (*Pseudotsuga menziesii*/ *Tsuga heterophylla* [Raf.] Sarg.) in the Pacific Northwest, cherrybark oak/sycamore (*Quercus* spp./ *Platanus occidentalis* L.) in Arkansas, European beech (*Fagus sylvatica* L.) and Austrian pine (*Pinus nigra* Arnold) in Italy, Scots pine in Finland and upland hardwoods in Missouri (D.R. Larsen, personal communication). The Sylvan model was chosen because it offers the needed flexibility to develop site-specific and species-specific coefficients, forecasts growth of real stands, produces realistic interspecific and intraspecific crown interaction, is largely influenced by crown size, is based on accepted stand dynamics theory, and tracks crown recession over time.

COMPLETE MODEL

The design of the branch occlusion and crown allometry sub-models is such that the branch occlusion sub-model requires inputs from both a forest growth model and the crown allometry sub-model and the crown allometry sub-model requires inputs from a forest growth model. In combination, the forest growth model and crown allometry and branch occlusion sub-models constitute the complete wood quality model. The completed wood quality model is a set of simultaneous equations that draw inputs from the model user and the Sylvan individual tree model (Fig. 1). User inputs include a local dataset for developing customized growth model parameters for the Sylvan tree model. Second, model users define the initial plantation design by specifying species, spacing, and arrangement. In addition, users can export treelists from Sylvan at any point during the management timeline to utilize programs such as the U.S. Forest Service's Stand Visualization System (SVS) platform, which allows the user to define intermediate treatments such as thinnings. The SVS-altered treelist can be imported back into Sylvan and simulation continued.

Preliminary output from the wood quality model consists of both standard stand-level and standard tree-level metrics along with measures of knotty-wood and clear-wood production. Knotty-wood and clear-wood production can be expressed both at the stand and individual tree scale.

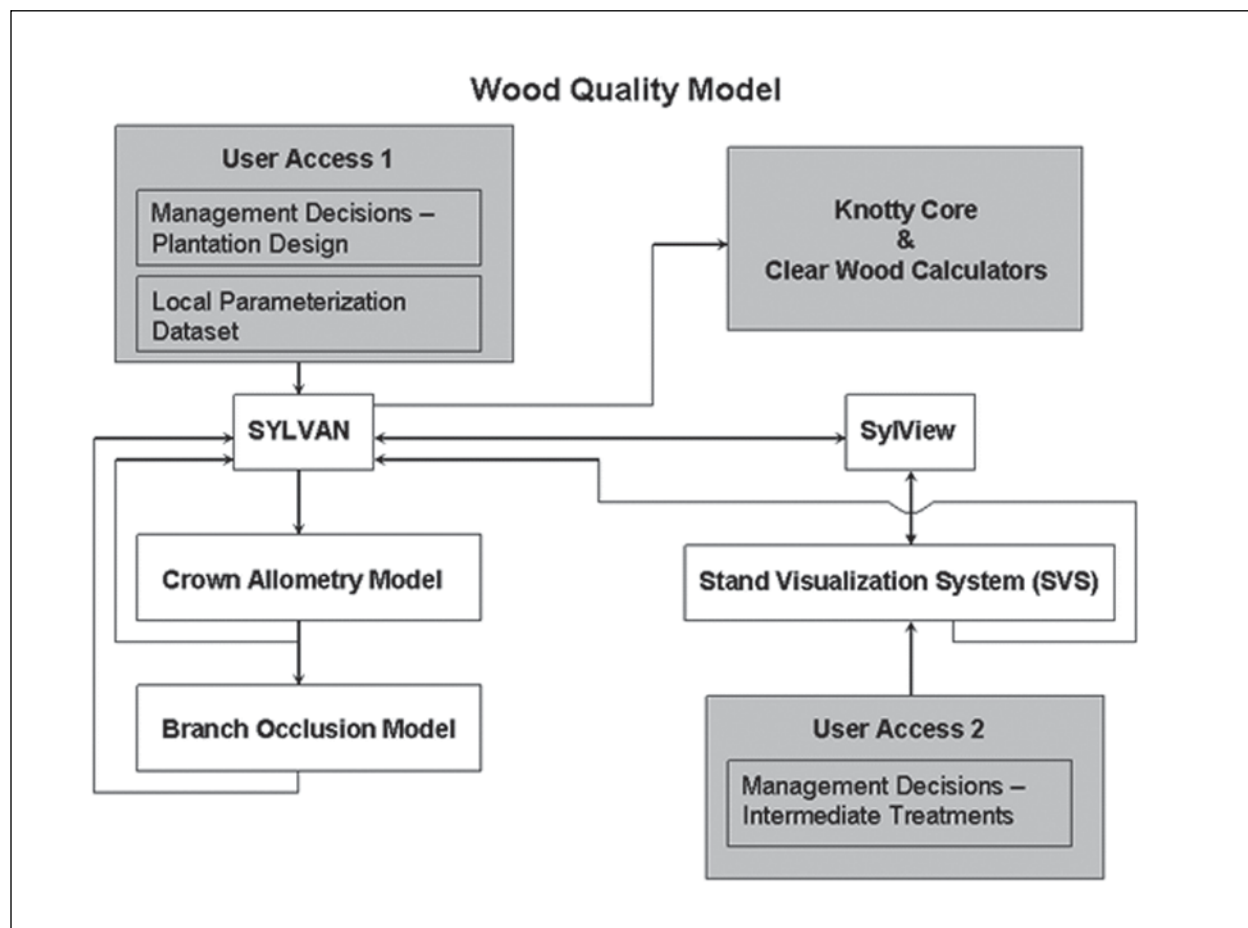


Figure 1.—Conceptual diagram of the crown allometry, branch occlusion and Sylvan models along with the knotty core and clear wood calculators, the SylView and SVS GUI's and user access points, which constitute the overall wood quality model.

The complete model is an empirical approach. Though the model may be more limited in its applicability than a mechanistic approach, empirical models generally produce more accurate predictions (Porte and Bartelink 2002). Additionally, the ability to parameterize the Sylvan model with localized tree datasets allows for increased applicability across sites. However, the branch occlusion and crown allometry equations are also empirically derived, so they would need to be re-parameterized for new species and species mixtures. Unlike for the Sylvan model, simple routines have not yet been developed for the re-parameterization of the branch occlusion and crown allometry equations with localized datasets.

OUTCOMES AND APPLICATIONS

RESEARCH

We are currently interested in aiding the design of plantation trials, such as identifying the most promising type of spacing and species combinations to investigate. In addition, comparisons can be made among management scenarios to fine-tune hardwood plantation management investigations. Furthermore, the wood quality model can be used to explore the relative importance of various processes, such as canopy stratification, in producing clear-wood; eventually researchers may be able to outline some general theories of how hardwood plantations develop.

HARDWOOD PLANTATION MANAGEMENT

Following completed evaluation of the wood quality model and packaging within a user-friendly graphical user interface (GUI), forest land owners and land managers will have a decision support tool that can help guide evaluations of potential hardwood plantation management approaches. Landowners and land managers will be able to effectively compare management scenarios with regards to impacts on clear-wood production (analogous to tree grade). Information about tree quality has been missing from hardwood simulation models; the wood quality model will equip decisionmakers with a key variable for assessing potential stand value.

CONCLUSION

The use of forested ecosystems as sinks for global carbon has gained considerable attention. Concomitantly, the reestablishment of bottomland hardwoods across the South is a priority for many organizations. As a result, reforestation and afforestation projects have increased considerably. Additionally, many landowners are interested in investing in hardwood plantations. Missing from current decision support tools is the ability to predict the quality of wood produced through the management of hardwood plantations and the impact of management decisions on individual tree quality. The hardwood quality model described in this study incorporates the much needed level of sophistication that will enable landowners and land managers to make sound decisions regarding hardwood plantations. The wood quality model can also be used by research scientists to develop sound hardwood plantation management strategies that include spacing and species assemblage recommendations.

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A 3-POINT DERIVATION OF DOMINANT TREE HEIGHT EQUATIONS

Don C. Bragg¹

Abstract.—This paper describes a new approach for deriving height-diameter (H-D) equations from limited information and a few assumptions about tree height. Only three data points are required to fit this model, which can be based on virtually any nonlinear function. These points are the height of a tree at diameter at breast height (d.b.h.), the predicted height of a 10-inch d.b.h. tree from an existing H-D model, and the height at species maximum d.b.h., estimated from a linear regression of big trees. Dominant sweetgum (*Liquidambar styraciflua* L.) from the Arkansas region and yellow-poplar (*Liriodendron tulipifera* L.) from across the southeastern United States were used to estimate height at species maximum d.b.h. A composite of these field-measured heights and site index trees from the U.S. Forest Service's Forest Inventory and Analysis (FIA) database were used to compare the 3-point equations (fit to the Chapman-Richards model) with the Forest Vegetation Simulator (FVS) default H-D models. Because of the limited range of diameters in the FIA site trees, the Chapman-Richards equations developed from site trees under-predicted large tree heights for both species. For the sweetgum, the 3-point equation was virtually identical to the FVS default model. However, the 3-point equation noticeably improved dominant height predictions for yellow-poplar.

INTRODUCTION

Simulation models are increasingly used to project forest dynamics over large spatial and temporal scales. The Forest Vegetation Simulator (FVS), for example, now encompasses virtually the entire continental United States, as well as parts of Alaska and Canada (Dixon 2009). Most of these models incorporate several mathematical relationships to define the nature of simulated trees and stands, usually in the form of regression equations derived from field-sampled data. One of the best examples of these models can be found in height-diameter (H-D) models, which use any of a number of functions to describe the trend between total tree height and (typically) diameter at breast height (d.b.h.).

Numerous H-D models have been developed for North American tree species (for example, Huang and others 1992, 2000; Lootens and others 2007; Zhang 1997). However, it is unusual for H-D models to be derived across the size range of a given taxon, primarily because few data sets include very large trees. Centuries of lumbering, land clearing, and catastrophic disturbance have greatly reduced the number of big trees, making their presence rare in most inventories. Some have sought to improve upon existing allometric relationships by expanding the range of data via the incorporation of champion-sized trees (for instance, Bragg 2008b, Shifley and Brand 1984). This approach has some limitations, including sensitivity to the individual chosen as the champion tree and the possibility that the specimen selected may not adequately represent the maximal expression of height and diameter, especially for open-grown champions (Bragg 2008b).

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Regional H-D models are often used in forest simulators because of their relatively simple and consistent implementation. However, these regional H-D models can suffer from inadequacies in their derivation. One possible option is to use some form of local adjustment for stand conditions (such as mixed-effects modeling), although this approach would require additional processing of the original data, and without more field sampling is not capable of extending the range of data. In this paper, I present a new approach to creating H-D equations for sweetgum (*Liquidambar styraciflua* L.) and yellow-poplar (*Liriodendron tulipifera* L.) from limited field data and existing models.

STUDY AREAS

Field data on tree heights for this study were gathered from two distinct sources. Big tree heights were taken with the sine height measurement technique (Bragg 2008a) by the author and other members of the Eastern Native Tree Society (ENTS) on several visits to different forests. For sweetgum, this sampling occurred in the Arkansas area, and included trees from Overton Park (Memphis, TN), the Levi Wilcoxon Demonstration Forest (Ashley County, AR), Hyatt's Woods (Drew County, AR), Noxubee National Wildlife Refuge (near Starkville, MS), and Big Oak Tree State Park (Mississippi County, MO). The more geographically diverse yellow-poplar sample came from stands in the Bankhead National Forest, AL; Cohutta Wilderness, GA; Ocmulgee Flats, GA; Panther Creek, GA; Sosbee Cove, GA; Sugar Cove, GA; Blanton Forest, KY; Bullen Creek, MS; Western North Carolina Nature Center, NC; Shelton Laurel, NC; the Great Smoky Mountains National Park, NC and TN; Tamassee Knob, SC; Lee Branch, SC; Meeman-Shelby State Forest Park and Overton Park near Memphis, TN; Savage Gulf, TN; and Fall Creek Falls State Resort Park, TN.

Site index trees from the U.S. Forest Service's Forest Inventory and Analysis (FIA) program were combined with the ENTS height data to help validate the models' predictions. For sweetgum, this sample consisted of 298 trees from the FIA plots across the state of Arkansas. For yellow-poplar, 2,860 FIA site index trees from Alabama, Georgia, Mississippi, North Carolina, South Carolina, and Tennessee were used. These publically available data were downloaded from the FIA website (<http://fiatools.fs.fed.us/fiadb-downloads/datamart.html>).

METHODS

The use of limited numbers of data points to fit tree attributes is not new—for example, Zeide (1999) developed a system to predict height growth in southern pines using only two points. His design assumed simple two-parameter equations, while many H-D models use at least three. Hence, for this mathematical reason, as well as the biological behavior of most trees, I will pursue a 3-point design.

NONLINEAR MODEL FITTING APPROACH

The 3-point H-D derivation approach makes several assumptions about tree height and the nonlinear nature of its relationship to d.b.h. Under ideal circumstances, only three data pairs are needed to fit simple nonlinear equations—one to establish the beginning point, one to express the most common portion of the data range, and a final one to delimit the maximum value of an asymptotic H-D function (Fig. 1).

Fortunately, the first point to establish the beginning of the range can be considered a universal constant—by definition, a tree that has just obtained a d.b.h. has a height of 4.5 ft, regardless of species, site quality, stand density, and so on. Thus, this data pair can be defined as (>0.0, 4.5).

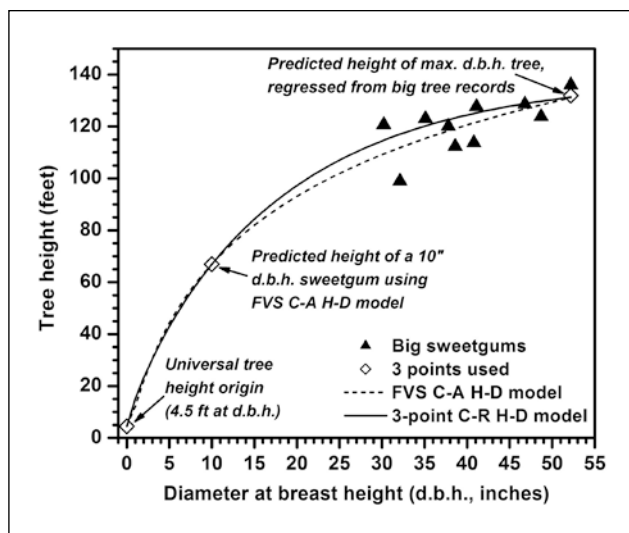


Figure 1.—A diagram of how the 3-point dominant H-D modeling approach worked for sweetgum from the Arkansas area. The three points (open diamonds) are chosen as the universal tree height origin, the predicted height of a tree at 10 inches d.b.h. (using a reasonable model), and the height of a tree of the “biggest” d.b.h., predicted from a simple linear regression of big tree records.

The second data point should be taken from where a preponderance of tree diameters can be found. Across most of the Central Hardwoods Region, centuries of logging and other disturbances have greatly reduced the number of very large stems, so the overwhelming majority of trees have relatively small diameters. Since this work focuses on species with considerable potential to grow large (in other words, no large shrubs), a tree of modest diameter (say, 10-inch d.b.h.) should provide a reasonable approximation of this central tendency. But what height should be used for this 10-in. d.b.h. stem? It would be possible to take samples either from the field or a publically available database (such as FIA), but it is easier to use an existing H-D model to predict its height. For trees of this size, most H-D models were derived from an abundance of trees. It is unlikely that there will be substantial differences between most regression equations at modest diameters—this conclusion is apparent in other research comparing different H-D functions (for example, Huang and others 1992, 2000).

The final information needed for the 3-point approach is a reliable estimate of the maximum height. Because virtually every H-D equation is designed to be monotonically increasing, by definition tree height will be maximized at the species’ maximum diameter. Though there is no biologically maximal tree diameter as there appears to be for height (for example, Ryan and Yoder 1997, Koch and others 2004, Domec and others 2008), there is a general frontier that can be considered to be an upper diameter limit. For this paper, I assumed that the maximum diameters were 52.2 inches for sweetgum (from a sample of 10 very large individuals) and 87.2 inches for yellow-poplar (from a sample of 72 very large individuals). These data were then used to fit linear ordinary least squares regression equations to produce a maximum tree height estimate for the actual (measured) maximum diameters.

I followed the same approach for developing the 3-point regression equations for both species. The minimum pair tree data followed the universal height starting point, while the moderate pair (10-inch d.b.h.) height was predicted using the Curtis-Arney equation of the Forest Vegetation Simulator (FVS) (Forest Vegetation Simulator Staff 2009) and the maximum pair was generated as described in the previous paragraph. Three sets of data pairs (Table 1) were then fit with nonlinear regression to the Chapman-Richards function:

$$HT = 4.5 + b_1(1 - e^{-b_2 \cdot d.b.h.})^{b_3} \quad [1]$$

where HT = total tree height and b_1 - b_3 are species-specific coefficients. The Chapman-Richards function is popular among H-D modelers because of its biologically interpretable parameters and simple yet flexible

Table 1.—Data pairs (d.b.h. = x, height = y) used to fit 3-point regression models to Chapman-Richards function. See text for definitions of these data pairs.

Species	At minimum		At moderate		At maximum	
	d.b.h.	height	d.b.h.	height	d.b.h.	height
	in.	ft.	in.	ft.	in.	ft.
Sweetgum	0.0001	4.5	10.0	66.9	52.2	131.9
Yellow-poplar	0.0001	4.5	10.0	69.6	87.2	171.2

form (Kershaw and others 2008). The FVS default (Curtis-Arney) equation (Forest Vegetation Simulator Staff 2009) is similar in some aspects:

$$HT = 4.5 + b_1 e^{(-b_2 \cdot \text{d.b.h.} \cdot b_3)} \quad [2]$$

but the parameters are not as interpretable and this particular equation does not generate an asymptote for most species within biologically reasonable heights. Note that the Chapman-Richards and 3-point models were designed to predict dominant tree heights, not the average height of the entire population. However, it is not unusual for researchers to exclude suppressed trees or those with broken tops (for example, Lootens and others 2007). Presumably, site index trees were chosen from dominant and co-dominant specimens in the stands where they were found—this approach was definitely true of the data in the ENTS big tree collection.

COMPARISON OF MODELS

The effectiveness of the 3-point approach was evaluated using a comparison between the three different regression models: the 3-point Chapman-Richards derived model, the FVS default height equation (Curtis-Arney), and a Chapman-Richards model fit only with the FIA site trees (CRSI). Goodness-of-fit comparisons are made using the following measures—fit index (R^2), root mean square error (RMSE), bias, and corrected Akaike Information Criterion (AIC_c):

$$R^2 = 1 - \frac{\sum_{i=1}^n (HT_i - \widehat{HT}_i)^2}{\sum_{i=1}^n (HT_i - \overline{HT})^2} \quad [3]$$

$$RMSE = \sqrt{\sum_{i=1}^n (HT_i - \widehat{HT}_i)^2 / (n-p)} \quad [4]$$

$$\text{bias} = \sum_{i=1}^n (\widehat{HT}_i - HT_i)^2 / n \quad [5]$$

$$AIC_c = 2p + n[1n(\hat{\sigma}^2)] + \frac{2p(p+1)}{n-p-1} \quad [6]$$

In these equations, height (HT) and predicted height ($\widehat{HT}$) are for the i^{th} tree, n is the total number of observations, p is the number of function parameters, and $\hat{\sigma}^2 = \sum_{i=1}^n \hat{\epsilon}_i^2 / n$, where $\hat{\epsilon}_i$ are the estimated residuals from the fitted model. In addition to these test statistics, visual comparisons in model performance have been made.

RESULTS

Nonlinear regression coefficients for the three different models of sweetgum and yellow-poplar can be found in Table 2. Of the models tested, the Chapman-Richards equation fit specifically to the FIA data (CRSI) fit the best for both sweetgum and yellow-poplar (Table 3). All sweetgum models did a reasonable job of projecting the data, with fit indices exceeding 0.52 in each case and only limited amounts of error and modest negative bias appearing in all of the models. However, using the ΔAIC_c criterion of 2 as the threshold for separating the models with the substantial evidence as the best approximating model from those with less support (Burnham and Anderson 2002), it appears that the 3-point model is just as good as the CRSI for sweetgum (Table 3).

Figure 2 shows the sweetgum data (a) from the Arkansas FIA database and a comparison (b) of the FVS default H-D equation with the FIA site index-derived Chapman-Richards equation. While there is little apparent difference between these models up to about 25 inches d.b.h., there are growing differences as tree size increases. The 3-point and FVS H-D equations differ even less across the entire range of tree diameters considered, especially in the context of the large tree data (Fig. 2c).

Table 2.—Nonlinear regression coefficients for the three models.

Model by species	b_1	b_2	b_3	n
Sweetgum				
FVS default (Curtis-Arney) ^a	290.90548	3.62395	-0.37201	?
Chapman-Richards, SI only	107.017	0.10970	1.29264	298
3-point Chapman-Richards	134.359	0.05198	0.84997	3
Yellow-poplar				
FVS default (Curtis-Arney) ^a	625.76966	3.87321	-0.23349	?
Chapman-Richards, SI only	130.620	0.07882	1.05417	2,860
3-point Chapman-Richards	159.821	-0.10301	2.03246	3

^a Coefficients from Forest Vegetation Simulator Staff (2009) and rounded off to 5 decimal places.

Table 3.—Goodness-of-fit statistics by species for comparisons among the FVS default height models, Chapman-Richards models fit to the FIA site index (SI) trees only, and the 3-point Chapman-Richards models, all using the SI and Eastern Native Tree Society height data for validation. All models have three parameters.

Model by species	R ²	RMSE	bias	AIC _c	ΔAIC_c *
Sweetgum (304 trees, average height = 76.6 ft)					
FVS default (Curtis-Arney)	0.527	10.936	-1.900	1457.5	13.0
Chapman-Richards, SI only	0.547	10.705	-0.122	1444.5	0.0
3-point Chapman-Richards	0.544	10.734	-0.798	1446.1	1.7
Yellow-poplar (2,932 trees, average height = 88.2 ft)					
FVS default (Curtis-Arney)	0.446	14.993	-7.620	15880.3	972.8
Chapman-Richards, SI only	0.602	12.701	-0.771	14907.5	0.0
3-point Chapman-Richards	0.446	14.986	4.730	15877.6	970.1

* ΔAIC_c = current AIC_c – minimum AIC_c

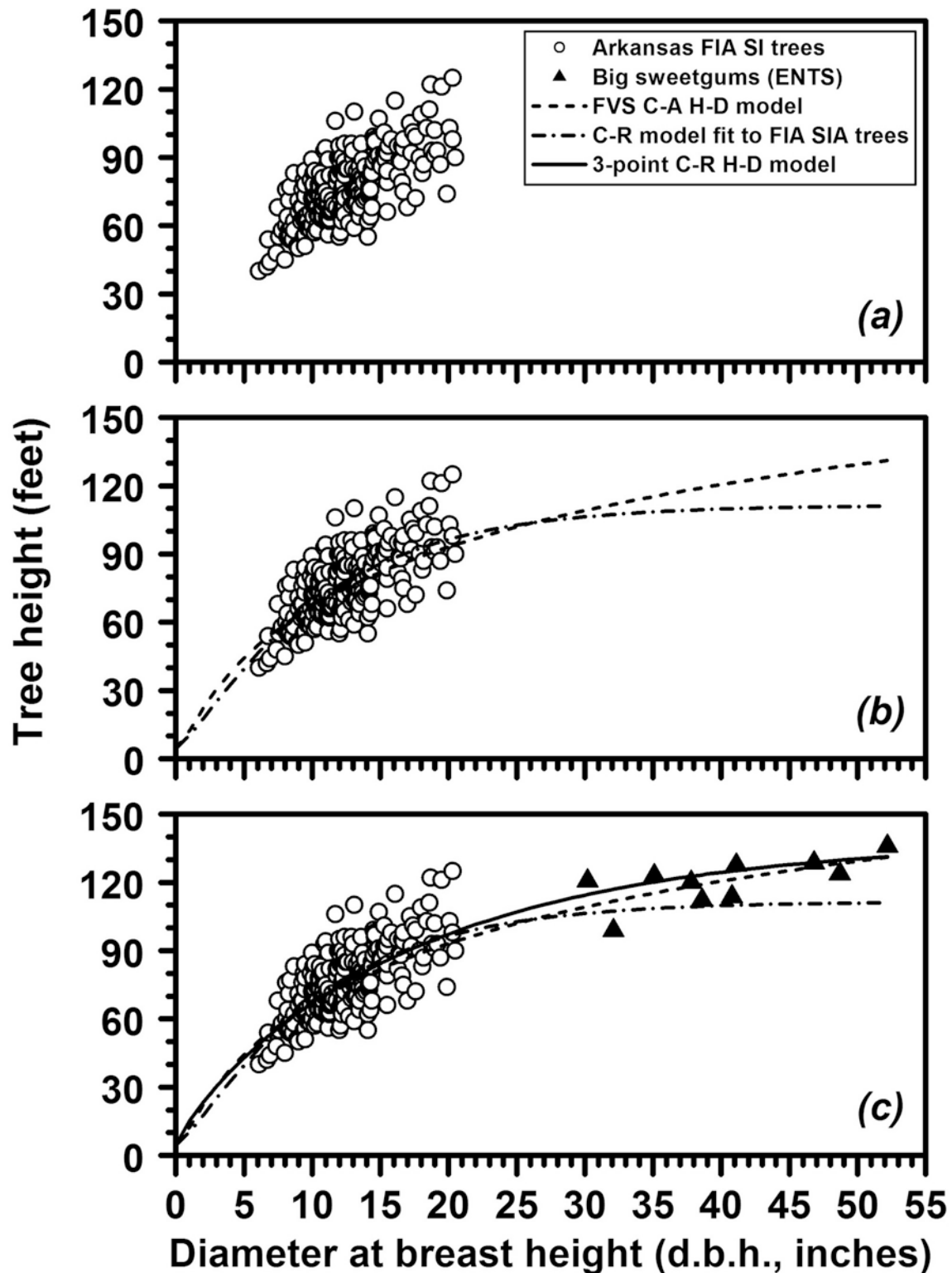


Figure 2.—Validation of the 3-point method for Arkansas-area sweetgum height prediction, using the independent FIA site index tree data for the state (a). The FVS default H-D model and a H-D equation developed from the Chapman-Richards function fit to the FIA site index data differed little (b) until sweetgums exceeded 25 inches d.b.h. The 3-point and FVS default H-D models were virtually indistinguishable across much of the size range (c).

Yellow-poplar had substantially better goodness-of-fit measures with the CRSI than with the 3-point or FVS default (Curtis-Arney) H-D equations (Table 3). Both of these models had fit indices of 0.446 (compared to 0.602 for the CRSI model) and significantly greater bias and AICc scores (there are only very minor differences between the 3-point and FVS default equations). Figure 3 illustrates the differences among these models—unlike sweetgum, noticeable departures from the yellow-poplar 3-point model appeared for small-, medium-, and large-diameter trees with both of the other competing models. The test statistics (Table 3) are highly influenced by the large number of small- to moderate-sized yellow-poplars, so the better fit of the 3-point model across the large-diameter trees was not enough to substantially improve the goodness-of-fit outcomes.

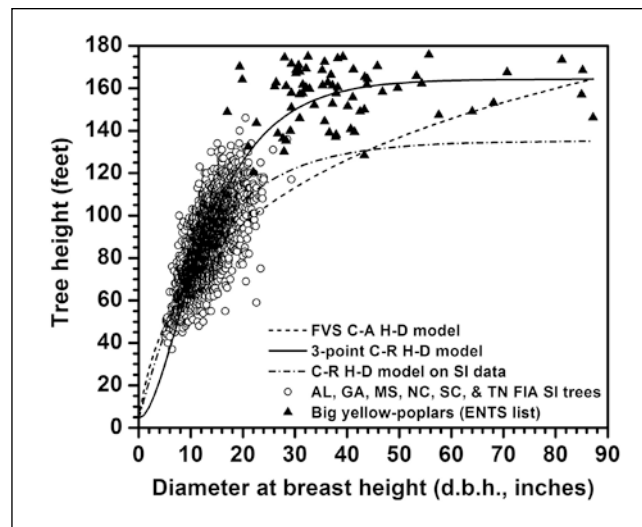


Figure 3.—H-D models developed for yellow-poplar across the southeastern U.S. differed considerably more than those for the Arkansas sweetgum. The 3-point model did a noticeably better job of fitting large-diameter trees.

Figure 4 highlights the differences in outcomes based on model type compared to the validation data. Unlike the sweetgum models from the Arkansas region, which showed very little difference, the yellow-poplar models varied considerably in their predictions. In general, the 3-point model predicted noticeably shorter trees at small diameters (up to 10-in. d.b.h.) and substantially greater heights for moderate- to large-sized yellow-poplars. The CRSI and FVS default H-D models for yellow-poplar differed less up to about 50-in. d.b.h., after which considerable differences in height were predicted between these models. Interestingly, the 3-point and FVS default models predicted virtually the same height for the tree of maximum d.b.h. with both species.

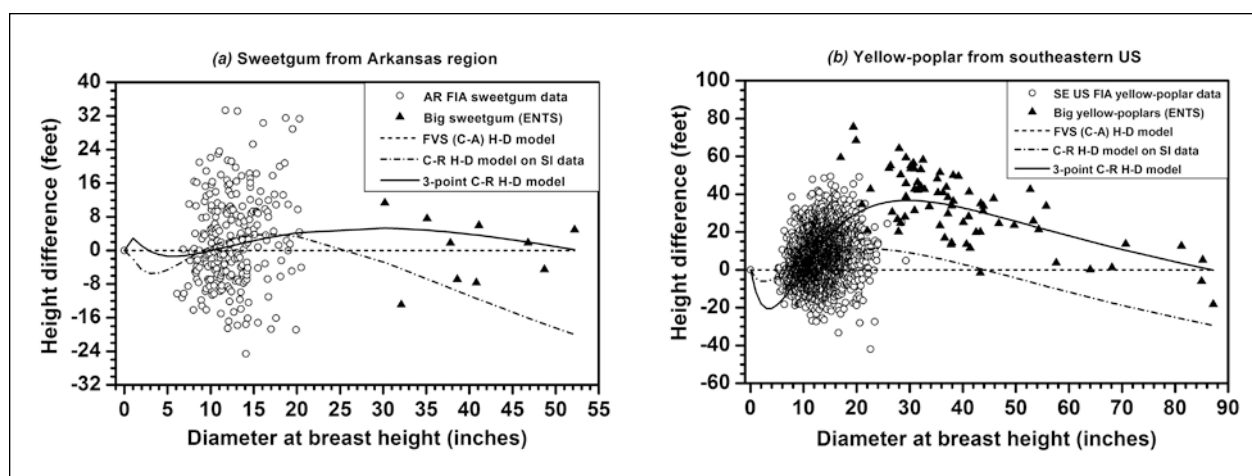


Figure 4.—Residual analysis for Arkansas-area sweetgum (a) and southeastern United States yellow-poplar (b). All comparisons are made relative to the published FVS default model (dashed line). Note the remarkable (and likely coincidental) agreement between the 3-point and FVS default models at the maximum diameter simulated for both species.

DISCUSSION

MODEL PERFORMANCE

It is not surprising that the CRSI models had the best fit to the validation data of the three compared—it was the only one using all of this information. This result applies especially to the yellow-poplar model because of the very large size ($n = 2,932$ trees) of the data set. However, conventional goodness-of-fit statistics are not necessarily the best measure of predictive success when models are fit to this type of data.

Most H-D data sets have a large quantity of noise—there simply are too many factors influencing actual tree height to account for them all in a single, simple equation. For example, the 95-percent confidence intervals for the equations Lootens and others (2007) fit to their data are quite wide, suggesting that any of several model forms could have been used and would have been capable of predicting height from diameter just about as reliably. Others have used additional factors (for example, dominant canopy height [Kershaw and others 2008] or local spatial patterns [Zhang and others 2004]) or more complicated model-fitting approaches (for example, mixed-effects [Trincado and others 2007, Budhathoki and others 2008] or geographically weighted regression [Zhang and others 2004]) to improve the fit of their equations.

However, relatively small gains in predictive ability, even if significant, are probably less meaningful than an accurate portrayal of the relationship between height and diameter across the full range of tree sizes (see also Lei and Parresol 2001). For instance, many researchers are concerned about the behavior of their models at the extremes of the diameter range (for example, Lei and Parresol 2001, Lootens and others 2007, Bragg 2008b), knowing that many such equations are often applied beyond the range of the data for which they were fit. In this study, the CRSI equation for yellow-poplar matched the validation data much better than either the 3-point or the FVS default equations, but because of the limited data range of the CRSI model, it had a maximum tree height asymptote of about 131 feet. Many yellow-poplar exceed this height across the southeastern United States.

This prediction highlights one of the biggest challenges to using a data set where most of the samples are in the smaller end of the possible tree size range. These data unduly influence model fit and may fail to reflect the true relationship of height and diameter as the trees reach larger size classes. Bragg (2008a) proposed a different model extension technique using champion tree data and a range of data points from the original H-D model to address this issue. Because of how Bragg's (2008a) approach is applied, however, it is more sensitive to a single big tree height than is this 3-point method.

FUTURE APPLICATIONS

The 3-point approach for dominant tree H-D equations is still in the earliest phases of development. These preliminary results nevertheless suggest that this approach offers a biologically and statistically reasonable alternative to other forms of model design. In an era of increasingly limited resources and a growing dependence on simulation models, the creation of accurate predictive systems with limited development costs will be important.

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EXAMINING SPRING PHENOLOGY OF FOREST UNDERSTORY USING DIGITAL PHOTOGRAPHY

Liang Liang, Mark D. Schwartz, and Songlin Fei¹

Abstract.—Phenology is an important indicator of forest health in relation to energy/nutrient cycles and species interactions. Accurate characterization of forest understory phenology is a crucial part of forest phenology observation. In this study, ground plots set up in a temperate mixed forest in Wisconsin were observed with a visible-light digital camera during spring 2007. High-frequency microclimatic data were collected concurrently. A green pixel extraction technique was used to derive phenological patterns from digital photos. Correlations were examined between derived phenology signals and climatic variables. Results indicate that though understory phenology was generally correlated with spring warmth accumulation, its daily change rate was significantly correlated with air moisture variability rather than with temperature-based weather fluctuations. We inferred that understory flush regime during springtime in a mesic temperate forest could depend more on precipitation than on temperature, due to the presumably shallower root systems. Furthermore, the study suggests that visible-light digital photography is an effective, easy-to-implement method for observing understory phenology and for other forest monitoring tasks.

INTRODUCTION

Phenology tracks periodic life-cycle events driven primarily by changes in meteorological conditions (Schwartz 2003). A warming climate has led to shifts in the timing of plant phenology over the past half-century as detected from both observations and retrospective models (Menzel and others 2006, Schwartz and others 2006). Systematic monitoring of plant phenology is important for improved understanding of vegetation dynamics within the complex ecological context (Liang and Schwartz 2009). Within forest ecosystems, understory plant communities form an important component that influences the general composition and health of forests (Fei and Steiner 2008, Kaeser and others 2008). Observing forest understory phenology is necessary for gaining insight into how forest ecosystems would respond to the changing environment.

Given the multiplicity of species present on the forest floor and their respective physiologies in relation to growth stages, synthesizing understory phenology observed at the species level is difficult. We believe that reading the aggregated “greening” effect is an option for practical forest understory phenology observation and that digital photography serves as a useful tool for objective assessment. Applications of digital photography to phenological quantification have been recently investigated using networked or robotic imaging systems for forest canopies and selected plant species (Richardson and others 2007, Graham and others 2009). Applying such techniques to observations conducted under field conditions holds potential for effectively capturing phenological signals from the forest understory within areas of interest. In addition, the improved accuracy

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of measurement allows the detection of delicate interrelationships between understory plant phenology and microclimatic variables. Such inter-relationships have rarely been addressed by previous studies.

STUDY AREA

The study area is located in the Park Falls Range District of the Chequamegon National Forest in northern Wisconsin. The field sampling plots were set up near an AmeriFlux tower (WLEF, 45.946°N, 90.272°W). The gently sloping topography underlying the forest is covered with soils developed from glacial-fluvial deposits. The area has a humid continental climate with annual average temperature of 4.8° C, and annual precipitation around 810 mm, as recorded by the closest weather station at Park Falls. The forest is composed of 70 percent deciduous (such as *Populus tremuloides*, *Acer* spp., and *Alnus regosa*) and 30 percent conifer (such as *Abies balsamea*, *Pinus resinosa*, and *Thuja occidentalis*) tree species. More than 100 understory plant species are commonly found on the forest floor (Broszofske and others 2001).

METHODS

FIELD MEASUREMENTS

Phenology of understory vegetation cover was investigated with digital photography during spring 2007. Twenty-one 1-m² ground plots were outlined in the study area. Each plot was set using a PVC pipe “square” and marked with stakes at four corners. These understory plots were observed every other day from April 27 to May 25, 2007. A consumer-grade Kodak DX4530 visible-light digital camera was used to take nadir-pointing photos over each plot on each observation day. The observation was mostly conducted during the morning to early afternoon on each day. The species composition of each plot was recorded by digital photos. The understory plants were dominated by grasses and herbs; saplings of woody species were present in a few plots. Concurrently with phenology observation, microclimatic data were collected with automatic HOBO sensors (Onset Computer Corp. Bourne, MA) deployed across the study area. These sensors took ambient air temperature and humidity readings (at shelter height) every 10 minutes.

EXTRACTING PHENOLOGICAL SIGNALS

Digital photos taken with a visible-light camera contain separable red-green-blue (RGB) color bands. Different algebraic combinations of the brightness values of these color bands were utilized to quantify greenness of plants (Graham and others 2006, Richardson and others 2007). A recent study suggests that hue-saturation-luminance (HSL), which is translatable from RGB color bands, provides more accurate estimation of leaf pigment alterations by separating luminance/brightness (affected by illumination conditions) from the hue (Graham and others 2009). We therefore adopted this color-band transformation method to construct metrics of understory phenology for this study.

Each digital photo was cropped to include only the 1-m² plot area marked by four corner sticks. The RGB color bands were translated into HSL color space, and then a range of hue values (192-288) was applied to extract green pixels. The percentage of these green pixels was computed for each photo and used as the phenometric value of the understory at the time of observation. Preprocessing of digital photos was conducted with the Paint.NET program. Band algebra and green pixel extraction were performed using IMAGINE 9.2 software (ERDAS, Atlanta, GA).

COMPARISON WITH CLIMATIC VARIABLES

Temperature-based climatic indices were calculated for daily maximum, mean, and minimum temperatures and temperature range. In addition, accumulated growing degree hours (AGDHs) were calculated from the hourly mean temperature based on 0 °C threshold over selected time windows with an arbitrary start date of April 5, 2007. Likewise, we calculated precipitation-related climatic indices: daily max/mean/min/range of absolute humidity (AH), relative humidity (RH), and air water potential (WP). Air water potential in relation to plant transpiration was calculated using the Kelvin equation for temperature and relative humidity (Lambers and others 1998, Kirkham 2004). Daily advance rate of understory phenology averaged across plots was then compared with the averaged values of climatic variables using correlation analysis.

RESULTS

PHOTO-BASED PHENO-METRICS

An example from one of the ground plots is provided to demonstrate the binary image series with green pixels (shown in white) distinguished from the background (shown in black) using an HSL-based thresholding method for counting greenness percentages (Fig. 1). The incremental changes of green pixels with time could be clearly identified in the photos.

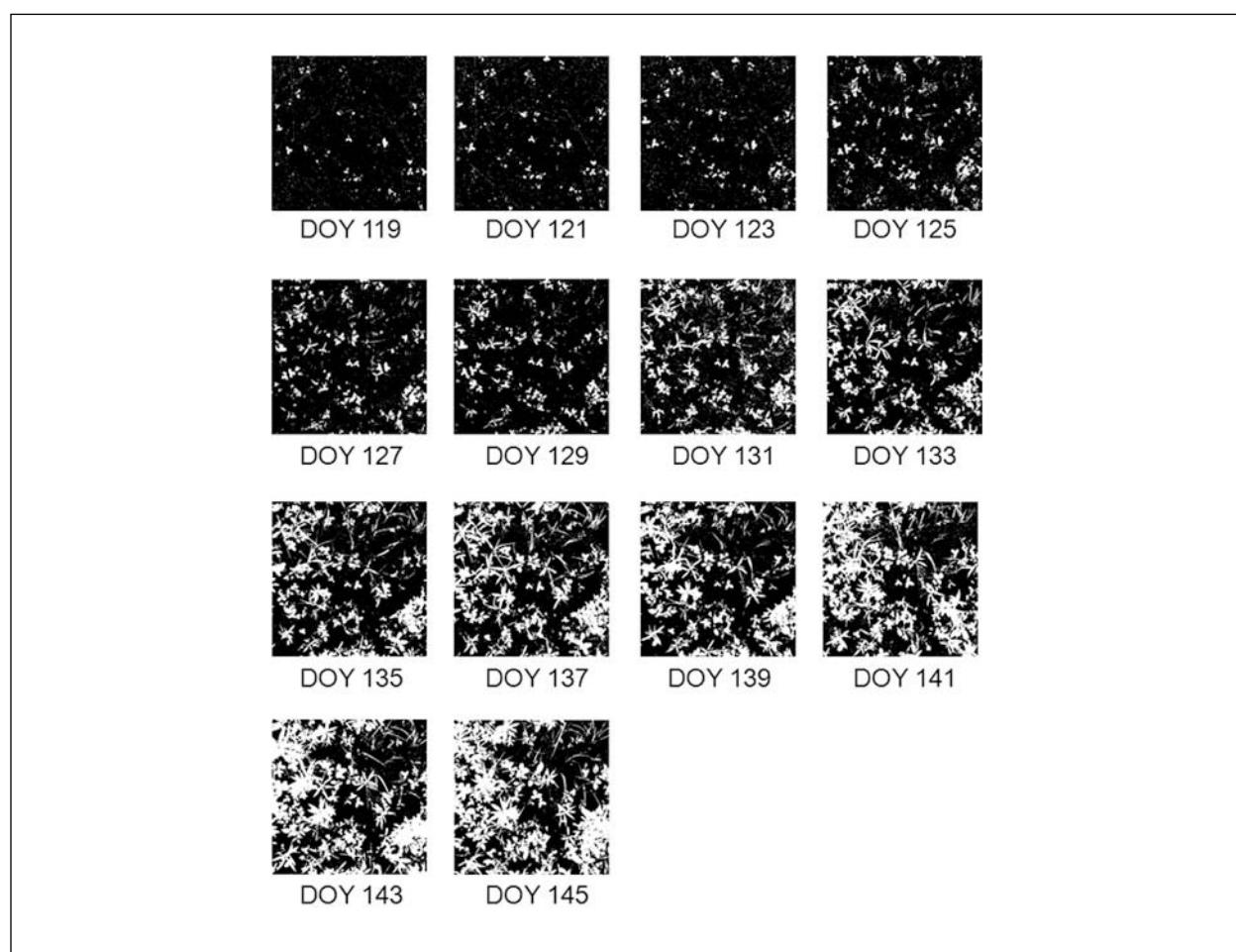


Figure 1.—Binary images of green pixels extracted from digital photos using thresholds based on HSL color space transformed from RGB bands (plot E09, spring 2007).

Each plot yielded a time series of phenological development measured with the photo-based pheno-metrics. The percent green cover averaged across plots showed consistent sensitivity to time and a stable upturning trend, which matches subjective knowledge about the understory greening process in spring (Fig. 2).

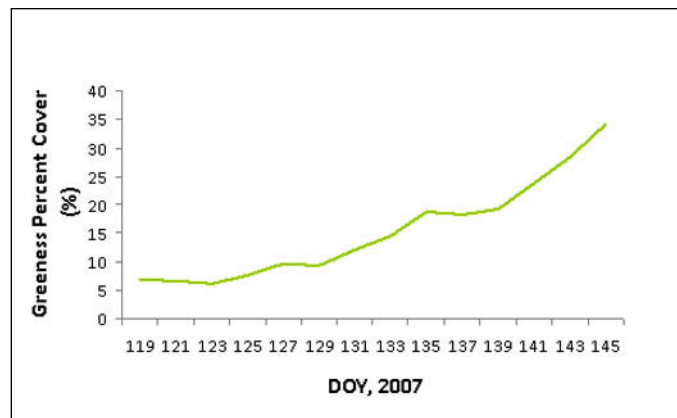


Figure 2.—Time series of averaged pheno-metrics: percent greenness cover of forest understory, spring 2007.

CORRELATIONS BETWEEN PHENO-METRICS AND CLIMATIC VARIABLES

Pearson correlation coefficient and significance values for pairs of daily increment rates of percent green cover and temperature-based climatic indices (AGDHs, growing degree hours [GDHs] over antecedent temporal windows up to 6 days, and daily temperature variables) are provided in Table 1. A strong positive correlation (0.724) was found between green pixel percentage index and AGDHs at the 99-percent confidence level. However, other pairs of phenological and temperature variables did not show significant correlations. A visual comparison between HSL-based pheno-metric advance rates and AGDHs indicated that the AGDH value was not sensitive to daily variation but matched the variation in the height of the peaks of the daily greenness increment, which increased with time (Fig. 3).

Results from the correlation analysis for understory phenology and averaged moisture indices (variables based on WP, RH and AH) are also provided in Table 1. Green pixel percentage measures demonstrated significant correlations with moisture variables at confidence levels of 95 percent or above. Visual comparison of the pheno-metric daily increment rate and mean WP demonstrated a clear correspondence of their temporal variation patterns (Fig. 4).

Table 1.—Correlations (Pearson correlation coefficient and significance values, 2-tailed) between pheno-metrics and climatic variables (significant values < 0.05 with corresponding correlation coefficients are indicated in bold letters).

	GDHs_1d	GDHs_2d	GDHs_3d	GDHs_4d	GDHs_5d	GDHs_6d	AGDHs
Cor.	0.333	0.433	0.542	0.477	0.399	0.320	0.724
Sig.	0.266	0.140	0.056	0.099	0.177	0.286	0.005
	TMean	TMax	TMin	TRange			
Cor.	0.073	0.121	0.339	-0.203			
Sig.	0.812	0.693	0.257	0.506			
	WPMean	WPMax	WPMin	WPRange	RHMean	RHMax	RHMin
Cor.	0.708	0.435	0.732	-0.729	0.611	0.119	0.765
Sig.	0.007	0.137	0.004	0.005	0.027	0.699	0.002
	RHRange	AHMean	AHMax	AHMin	AHRange		
Cor.	-0.799	0.616	0.700	0.500	0.528		
Sig.	0.001	0.025	0.008	0.082	0.064		

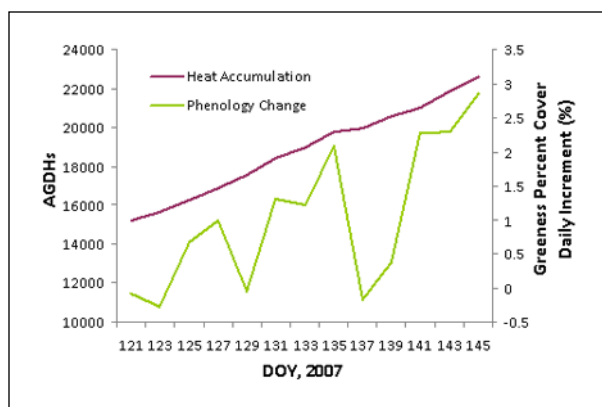


Figure 3.—Time series of daily increment of percentage greenness cover and accumulated growing degree hours (AGDHs), spring 2007.

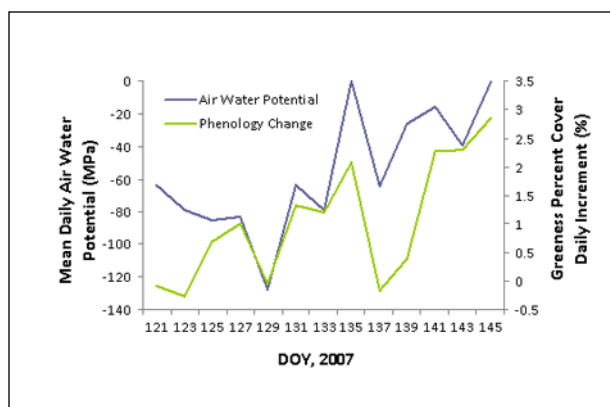


Figure 4.—Time series of daily increment of percent greenness cover and mean daily air water potential, spring 2007.

DISCUSSION

The significant correlation between forest understory phenology and AGDHs indicated a generally accelerated rate of grass flushing with spring warmth accumulation (Fig. 3). In other words, the daily increase of greenness percentage cover was greater in the late spring, associated with the increase of accumulated heat. However, AGDHs and daily GDHs were not correlated with the actual occurrences of understory flush as indicated by percent greenness cover change. Rather, each spur of the understory growth responded to increased moisture content in the air, presumably triggered by springtime precipitation events. Such a water-dependent phenology regime is common in the semi-arid environment. In the mesic forest investigated in this study, understory phenology could be susceptible to minor drought spells, likely due to the relatively shallow root systems of understory species. Actually, a field-observed drought around May 17, 2007 could have significantly reduced the normal growth of understory plants on the forest floor (Figs. 3 and 4). We also noticed in the field that a rapid understory flush occurred following an intense rainfall, confirming the relationship suggested by the data analysis. In summary, precipitation appears to be a more direct driver of understory phenology variation. Heat accumulation plays an underlying role that limits the degree or intensity of each understory flush.

We observed that combinations of understory plant species varied according to different light conditions created by nearby canopies; such heterogeneity may be a major source of variation. Consequently, spatial variations of understory phenology in relation to canopy illumination conditions and plant form variability need to be accounted for in follow-up studies. The strong correspondence between understory phenology and air water potential (derived from temperature and relative humidity) suggests that forest understory phenology could be modeled accurately with both temperature- and moisture-based climatic variables in a temperate forest. Additional efforts are needed to quantify the climate-dependency of forest understory phenology by using multivariate regression models to address questions related to climate change.

Forest understory phenology could be in an efficient vegetation health indicator useful in assessing forest structure and detecting change. The study of forest understory phenology also has broader linkages with forest nutrient and energy cycling and species interactions. Digital photography is an easy and cost-effective tool for observing understory phenology. Given the deepening effects of environmental change on forest resources, tracking the long-term dynamics of the often-overlooked understory phenology could further contribute to our knowledge base in support of management and conservation tasks.

ACKNOWLEDGMENTS

Eric Graham from UCLA provided insight into digital photo processing. Two reviewers' careful edits and constructive comments are sincerely appreciated. The research was partially supported by a Doctoral Dissertation Research Improvement Grant, National Science Foundation, BCS-0703360.

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DEVELOPING DIGITAL VEGETATION FOR CENTRAL HARDWOOD FOREST TYPES: A CASE STUDY FROM LESLIE COUNTY, KY

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William Conner, and Brian J. Williams¹

Abstract.—Digital vegetation is the computerized representation, with either virtual images or animations, of vegetation types and conditions based on current measurements or ecological models. Digital vegetation can be useful in evaluating past, present, or future land use; changes in vegetation linked to climate change; or restoration efforts. Digital vegetation can be spatially explicit at various scales: region, subregion, landscape, landtype, forest, or stand. Advances in computer technology allow us to build digital vegetation based on integrated environmental information (i.e., soils, topography, forest types, and vegetation composition).

INTRODUCTION

With recent advances in technology, three-dimensional visualization of forest landscapes has increased in sophistication and applicability (McGaughey 1998). Supplemented with geographic information systems and remotely sensed data, computer-animated pictures and movies can now be used to represent real-world forests in three dimensions. Using this medium allows us to visualize not only forest structure and composition, but also spatial and temporal dynamics occurring in forests. Visualization has advanced as a very useful tool for comparing management options and for analyzing forest dynamics at stand and landscape levels (Orland 1994, McCarter 1997, Wang and others 2006). By generating virtual forests, it helps forest planners and managers experiment with the consequences of various management scenarios; they can quantify and visually compare the differences. This approach allows comparison of various forests and landscapes without time limitations.

To visualize a forest landscape, the data needed include digital elevation models (DEMs), forest-type maps, tree images, tree size, stand densities, and species composition. Other landscape features such as roads, streams, and buildings can also be included to add to the quality of visualizations. Sometimes, however, it is hard to get the necessary information (e.g., tree size and stand density). An alternative is to use existing public datasets collected in ecosystems similar to the landscape desired for visualization. In this study, Forest Inventory and Analysis (FIA) data from the U.S. Department of Agriculture, Forest Service (Miles and others 2001) were used for the visualization of central hardwood forests. FIA data are a public domain database and have been widely used in scientific investigations and resource management. The specific objectives of this study were to: (1) visualize major forest types of the Central Hardwood Forest Region (CHFR) occurring in Leslie County, KY; and (2) discuss the potential applications of visualizations and the limitations of our visualizations of central hardwoods.

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STUDY AREA

The Central Hardwood Forest Region covers approximately 100 million acres in the eastern United States and is one of the largest forest areas in the country (Clark 1989). Climatic factors (i.e., winter temperature and summer drought) provide boundaries for the CHFR on the north, south, and west. The Appalachian Mountains provide a general boundary on the east (Clark 1989). Based on FIA's forest-type classification (Miles and others 2001), our study area contains the following six main hardwood forest types in Leslie County, KY: oak forest, yellow-poplar forest, mixed deciduous forest, oak-pine mixed forest, other mixed forest, and deciduous woodland (Table 1, Fig. 1). These six forest types occupy 86 percent of the county, and mixed deciduous forest (49 percent) is the main type in our study area. The dominant species in these six forest cover types are American beech (*Fagus grandifolia* Ehrh.), black oak (*Quercus velutina* Lam.), chestnut oak (*Q. prinus* L.), northern red oak (*Q. rubra* L.), white oak (*Q. alba* L.), scarlet oak (*Q. coccinea* Muenchh.), red maple (*Acer rubrum* L.), and pine (*Pinus* spp.). Tree heights range from 7 to 135 feet and tree density varies by forest type.

METHODS

Data for this study are from the FIA database (Miles and others 2001), which provides complete information on plot location, stand density, species composition, and height and diameter at breast height (d.b.h.) for individual trees. The DEMs can be downloaded from The National Map Seamless Server (provided by the U.S. Geological Survey, <http://seamless.usgs.gov/>). They can also be downloaded from The Geospatial Data Gateway (provided by the U.S. Department of Agriculture, Natural Resources Conservation Service, <http://datagateway.nrcs.usda.gov/GatewayHome.html>).

The FIA dataset was designed for determining the extent, condition, volume, growth, and depletion of timber on the nation's forest lands. Prior to 1999, all inventories were conducted every 7-10 years in the South and every 12-15 years in the North. With the passage of the 1998 Farm Bill, the U.S. Forest Service is required to collect data on 20 percent of the plots annually within each state. The research stations of the Forest Service, in cooperation with each state, are responsible for conducting these inventories and publishing summary reports for individual states and survey units (Miles and others 2001). Variables required for visualization (e.g., tree heights and density) are available in the FIA database.

The FIA data were filtered using combinations of qualifiers (e.g., locations, stand type, ownership, slope, species, d.b.h., and physiographic class) so that different forest types were simulated independently. Locations based on land-cover maps, stand type, age, ownership, and species were used as qualifiers to filter the data in this study.

Table 1.—Area and relative percentage of six forest types in Leslie County, KY.

Land cover type	Area (Hectare)	Percentage
Oak forest	73,715.4	28.50
Yellow-poplar forest	13,928.8	5.38
Mixed deciduous forest	126,043.6	48.73
Oak-pine mixed forest	4,060.4	1.57
Other mixed forest	2,865.2	1.11
Deciduous woodland	2,470.5	0.96
Total	258,679.1	86.24

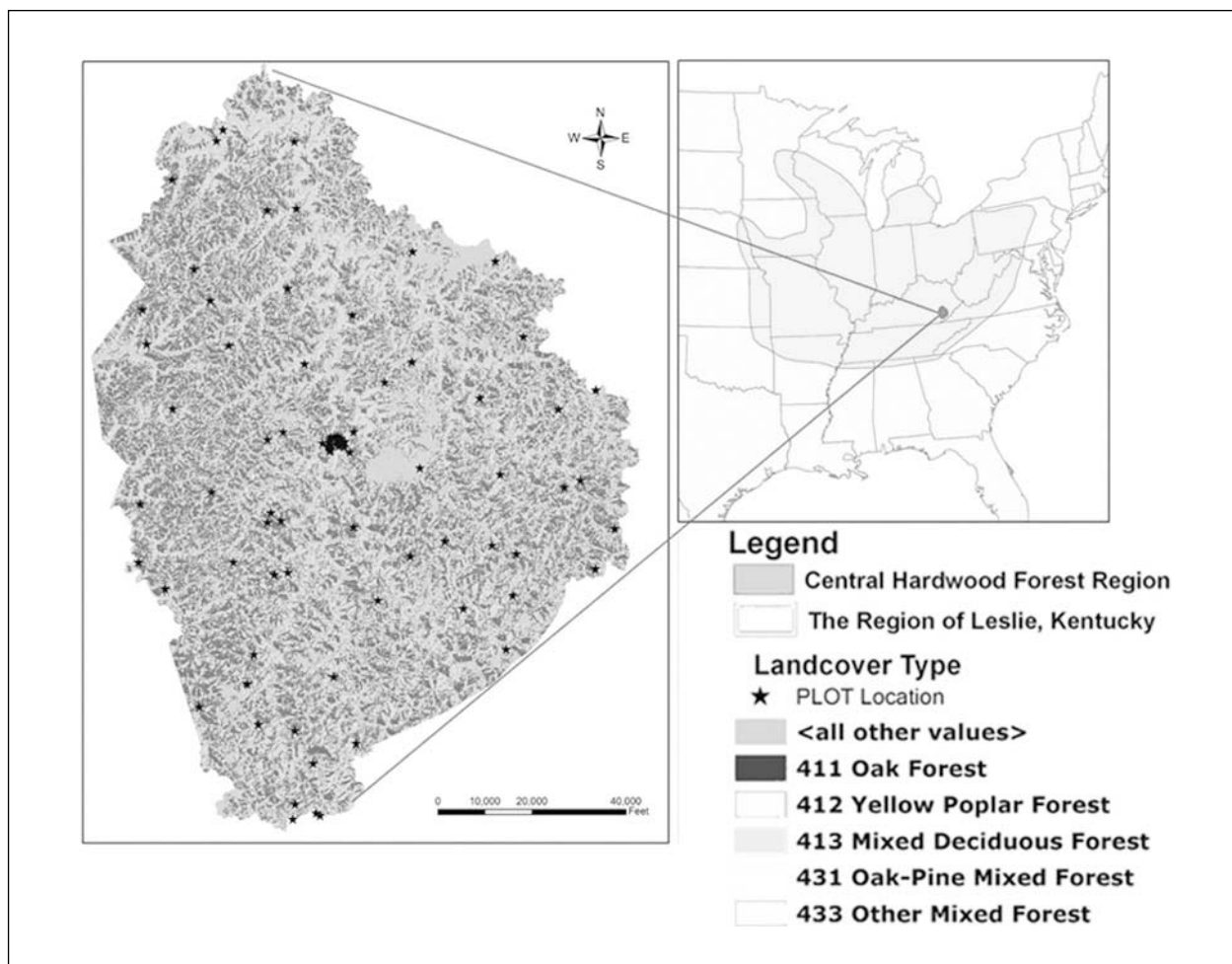


Figure 1.—Central Hardwood Forest Region and forest type map of study area, Leslie County, KY.

The first step of the visualization process is to generate realistic tree images. The detailed images of single trees are a crucial component of high-quality computer visualization. We developed a set of individual tree images of nine species dominant in central hardwood forests (Fig. 2). Different tree species were ranked by their densities in a forest, and the dominant species were chosen based on how many trees existed in this forest type. The individual tree images were generated in a graphics software package called TREE Professional (Onyx Computing Inc.; Fig. 2).

The second step was deriving spatially referenced forest data (tree species, sizes, forest composition, and density) from FIA data (Table 2). This step is critical for designing visualizations of the six major forest types within the study area. Then the vegetation views were visualized by VNS® (Visual Natural Studio; 3D Nature, LLC 2002). A series of images was prepared for each species in each forest type to account for variations in forest age and density. Subsequently, each forest type can be presented by digital vegetation (Figs. 3 and 4).

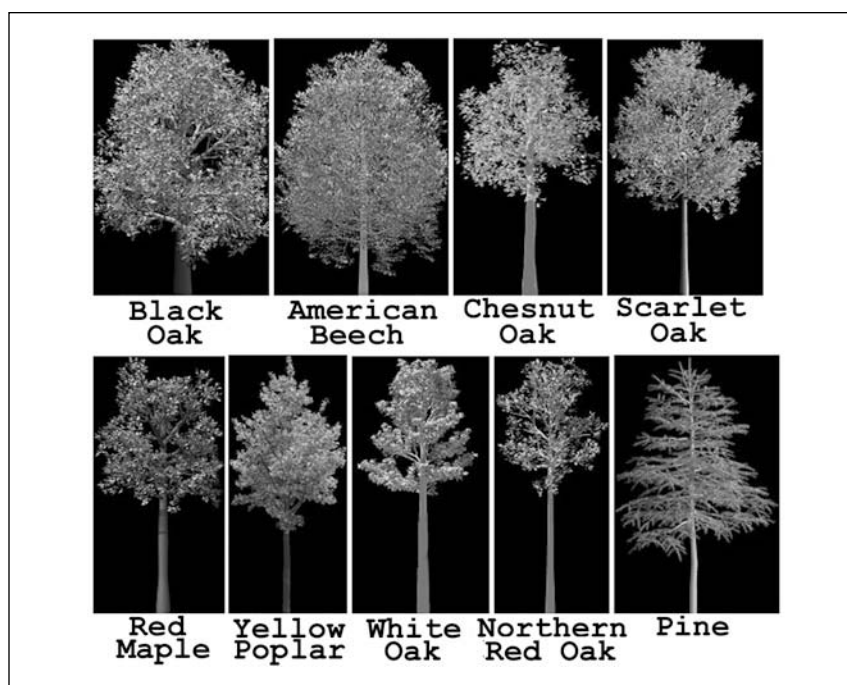


Figure 2.—Individual tree images of dominant species designed in Onyx® for visualization of six forest types in Leslie County, KY.

Table 2.-Species composition of six forest types in Leslie County, KY

Grid Code	Land cover type	Species	Composition	HT min (ft)	HT max (ft)
411	Oak Forest	red maple	26%	9.57	115.65
		yellow-poplar	23%	12.83	112.52
		chestnut oak	17%	8.00	133.00
		American beech	13%	13.78	108.27
		white oak	12%	10.06	77.12
		black oak	9%	26.30	95.04
412	Yellow-Poplar Forest	yellow-poplar	32%	13.00	114.00
		chestnut oak	24%	7.92	131.61
		American beech	17%	14.00	110.00
		white oak	15%	10.16	77.87
		black oak	12%	27.07	97.79
413	Mixed Deciduous Forest	red maple	24%	11.00	133.00
		yellow-poplar	21%	12.83	112.52
		chestnut oak	16%	8.00	133.00
		American beech	12%	13.78	108.27
		white oak	11%	19.00	100.00
		black oak	8%	10.06	77.12
		scarlet oak	5%	22.00	129.00
		northern red oak	4%	26.30	95.04
431	Oak-Pine Mixed Forest	yellow-poplar	35%	7.03	116.84
		chestnut oak	18%	9.97	76.46
		black oak	39%	28.23	102.01
		pine	17%	25.65	84.00
433	Other Mixed Forest	American beech	41%	12.70	111.39
		chestnut oak	37%	10.33	79.20
		black oak	22%	31.00	112.00
441	Deciduous Woodland	chestnut oak	47%	7.18	119.44
		yellow-poplar	38%	10.84	83.09
		black oak	15%	25.59	92.46



Oak Forest



Yellow Poplar Forest



Mixed Deciduous Forest



Oak-Pine Mixed Forest



Other Mixed Forest



Deciduous Woodland

Figure 3.—Visualization of stand view on six forest types in Leslie County, KY.

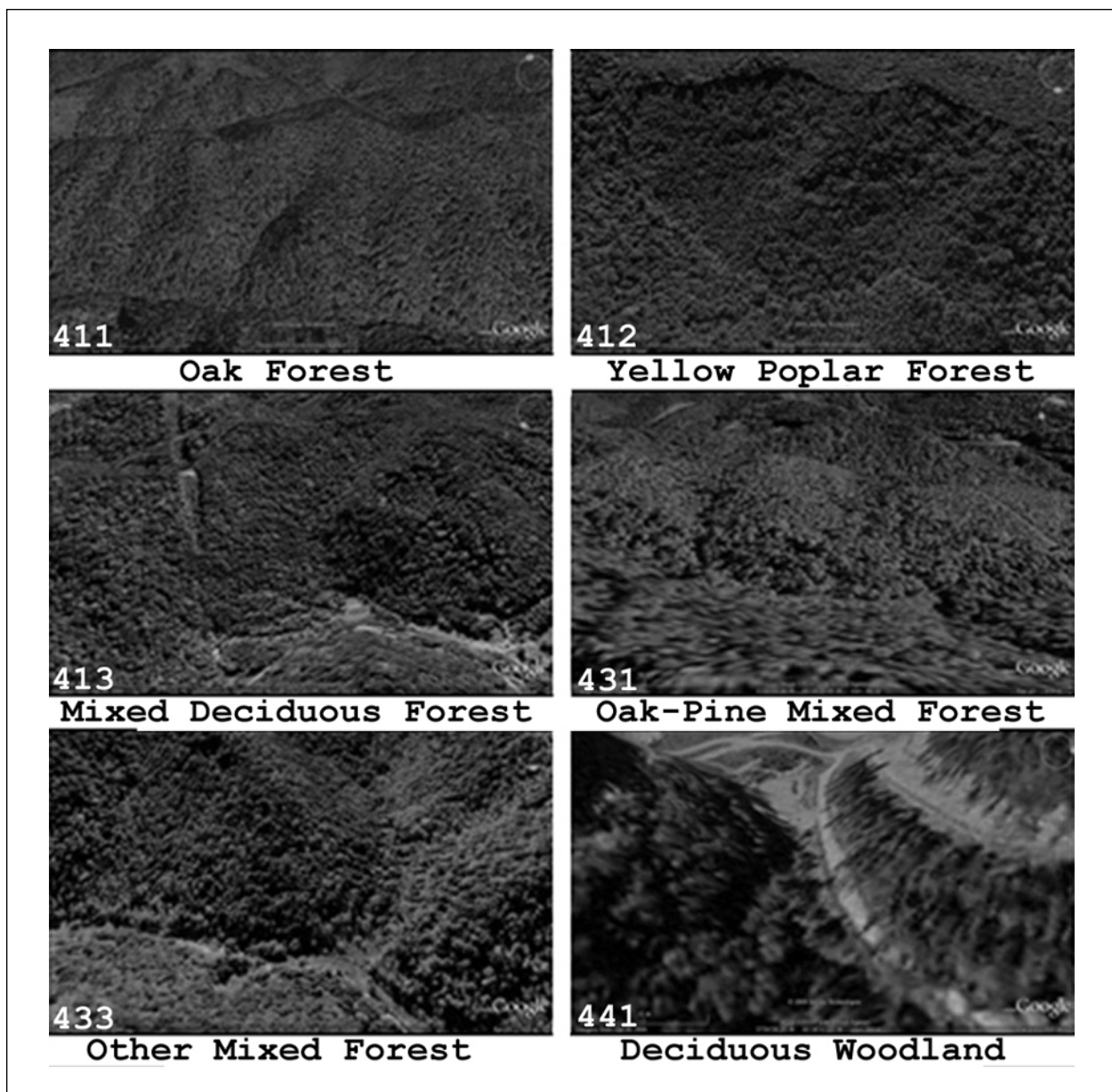


Figure 4.—Visualization of landscape view on six forest types in Leslie County, KY.

RESULTS

Among the six forest types of our study area, mixed deciduous forests occupied the largest area, 49 percent (Table 1, Fig. 1-pink), followed by oak forests, 29 percent (Table 1, Fig. 1-blue). Although the dominant species are similar in oak forests and mixed deciduous forests (e.g., similar percentages for red maple, yellow-poplar, and chestnut oak), the percentage of scarlet oak and northern red oak is higher in mixed deciduous forests (Table 2). According to FIA data, however, the basal area of oak in oak forests (approximately 127,000 sq ft) is almost twice as large as in mixed deciduous forests (approximately 66,000 sq ft). Yellow-poplar forests occupied only 5 percent of our study area (Table 1); dominant species were yellow-poplar (32 percent), chestnut oak (24 percent), and American beech (17 percent, Table 2). The other three forest types—i.e., oak-pine mixed forest, other mixed forest, and deciduous woodland—represent only 4 percent of the area (Table 1). Using the FIA data, virtual forests of the six forest types were visualized using VNS software (Fig. 3). For each type, a location on the DEM was chosen and the FIA data for that forest type were used to distribute the correct tree images to the view of that location. These virtual forest visualizations can be compared to a Google Earth™ aerial view of environments similar to those of the DEM base (Fig. 5).

In addition to production of pseudo-photographic views, the software can produce comparisons or representations that are unavailable in the real landscape. Figure 4 shows a comparison of all six forest types in an overhead view of a single landscape. Such a method would be most appropriate for planning of forest changes.

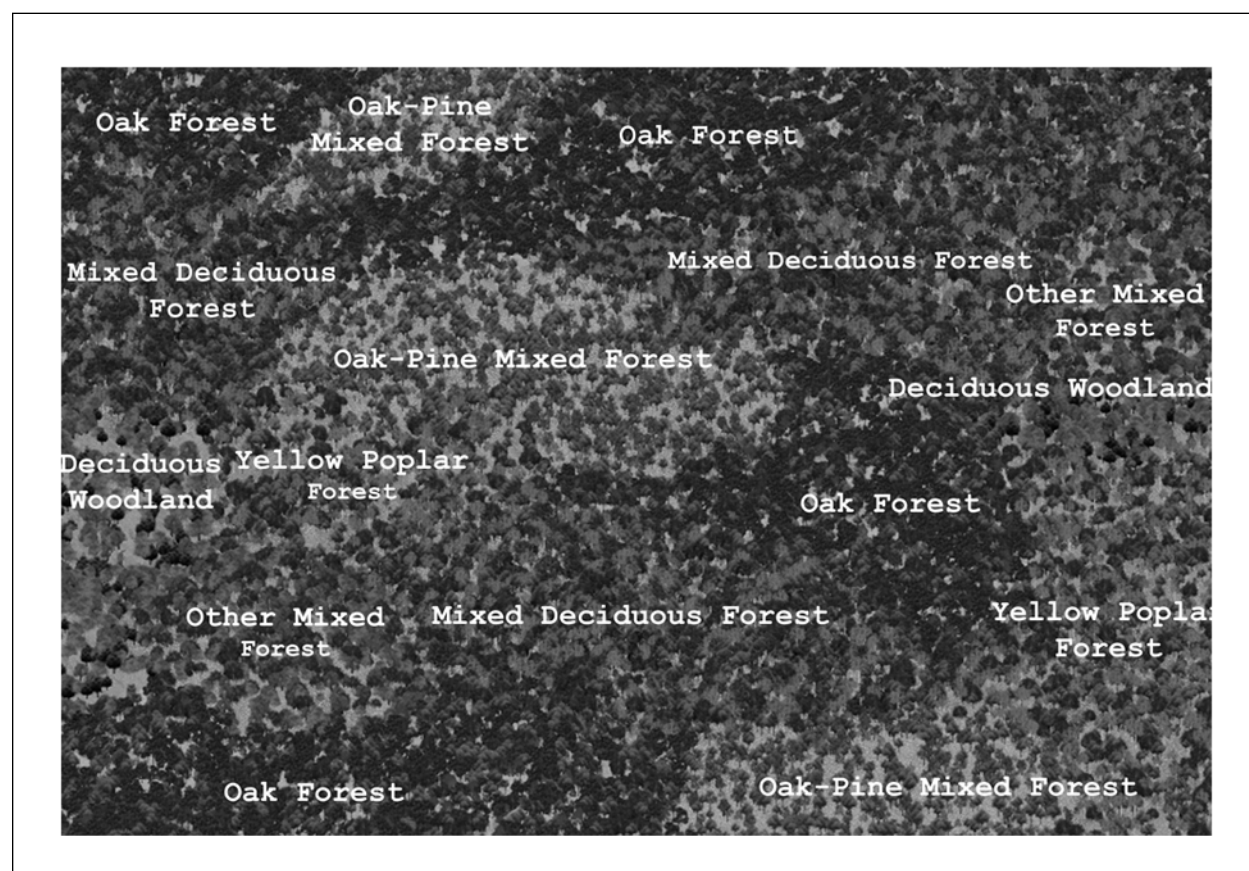


Figure 5.—Google Earth™ aerial views of the positions visualized in Figure 3.

DISCUSSION

These results demonstrate that public datasets can be used for visualization of forests and other landscapes. FIA data are one of the most accessible public forest datasets available, allowing low-cost and convenient visualization anywhere in the CHFR. Likewise, DEM data, at a resolution of 98-ft grid cells, are also freely available for the entire United States. Visualizations could be made using the aggregate data on species, age, size, and density for a randomly chosen county within the CHFR. Publicly available data on terrain shape and forest attributes produced photorealistic views of the two major forest types in this study. Since FIA data are allocated to represent an entire forest area, while the distribution of plots and subsequent representativeness of the data are allocated according to the area of each forest type, minor forest types may have relatively few points and the data may not represent the entire range of tree species, size, and density of the forest type.

Truthfulness and quality of visualizations are essential to forest managers or land owners (Daniel and Meitner 2001). From Figures 3 and 4 it is obvious that visualization based on FIA data is most realistic for the two most common forest types (mixed deciduous forest and oak forest). The quality of realism of the visualization (Figs. 3 and 4) appears to parallel area of the forest type (Table 2). Less common forest types appear much less likely to be visualized satisfactorily for photorealistic depictions. If the goal is depiction of small areas that are in an uncommon forest type, the FIA data do not appear to be sufficient; more detailed field inventory will probably be needed. Overall, the FIA data may be more applicable to non-photorealistic applications such as Figure 4.

This study revealed some of the difficulty that might be encountered when visualization is applied to central hardwoods. Application to a single random location showed that the quality of visualization depended on the representativeness of the original data source. Data collected for determining the extent, condition, volume, growth, and depletion of timber on the nation's forest lands will be concentrated on the most common, abundant species. In this visualization tree images were developed only for tree species that made up more than 5 percent of the overstory. Other minor species are not represented in the visualizations. The range of the CHFR is defined by the joint range of six species (hackberry [*Celtis occidentalis*], swamp white oak [*Quercus bicolor*], pin oak (*Q. palustris*), Ohio buckeye [*Aesculus glabra*], shellbark hickory [*Carya laciniosa*], and bur oak [*Q. macrocarpa*]; Little 1971, Fralish 2003) yet none of these species dominates the common stands of the CHFR. Visualization in this region may well require a very large library of tree images that includes many of the common minor species. Ignoring minor species may have also been a reason for the rather poor representations of four forest types.

Overall, satisfactory visualization can be obtained for major forest types within the CHFR using publicly available DEMs and FIA data. The results were not satisfactory for forest types that were of limited areal extent in this particular county.

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A THREE-DIMENSIONAL OPTIMAL SAWING SYSTEM FOR SMALL SAWMILLS IN CENTRAL APPALACHIA

Wenshu Lin, Jingxin Wang, and R. Edward Thomas¹

Abstract.—A three-dimensional (3D) log sawing optimization system was developed to perform 3D log generation, opening face determination, sawing simulation, and lumber grading. Superficial characteristics of logs such as length, large-end and small-end diameters, and external defects were collected from local sawmills. Internal log defect positions and shapes were predicted using a USDA Forest Service model. A 3D virtual reconstruction of a log and its internal defects was generated using 3D modeling techniques. Heuristic and dynamic programming algorithms were developed for opening face determination and sawing optimization while grading procedures were programmed based on the National Hardwood Lumber Association rules. The system was validated through comparisons of lumber results generated by the system and by sawmills. Our preliminary results indicated that a significant gain in lumber value can be achieved using this optimization system. This study will help small sawmill operators improve their processing performance and understand the impacts of defects on lumber grade, resulting in improved industry competitiveness.

INTRODUCTION

Currently, hardwood sawmills in the central Appalachian region face many challenges, including decreasing log size, low-quality logs, limited availability of logs, and pressure from foreign competition (Milauskas and others 2005). In addition, a weak global economy and the housing slow down have had impacts on the hardwood products industry in the region. All of these factors are pressuring hardwood sawmills to use more efficient processing methods that increase the value or volume of lumber produced.

Studies have shown that the value of hardwood lumber can be increased by 11 to 21 percent through optimal sawing strategies gained through the ability to detect internal hardwood log defects (Sarigul and others 2001). Existing nondestructive methods (x-ray or computed tomography) do not lend themselves to fast, efficient, and cost-effective analysis of logs and tree stems in the mill, especially for small sawmills. A three-dimensional (3D) laser-line scanner has been developed by the U.S. Forest Service in cooperation with Virginia Tech, and it uses relatively low-cost equipment (\$30,000 plus integration labor cost) (Thomas 2006). The Forest Service also has developed models to predict internal defect characteristics based on external defect measurements. A recent study has shown that the models can predict approximately 81 percent of internal red oak knot defects (Thomas 2010).

Log breakdown procedures must be combined with the scanned information to be successful at improving lumber value, otherwise it is not guaranteed (Occeña and others 1997). Software can generate hypothetical logs and defects, and simulate their breakdown. Over the past few decades, many log breakdown models

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and sawing simulation systems have been developed, but most lack practical sawing tools or 3D simulation training tools. The existing advanced sawing systems are not optimized for hardwood log breakdown or are not suitable for the small or portable sawmills typical of the central Appalachian region. To survive in a highly competitive marketplace in today's turbulent economic conditions, small sawmills need appropriate milling technology that will improve their profitability and competitiveness. One strategy is to use a cost-effective log sawing optimization system that combines the log breakdown models with the 3D scanned log data.

The overall objective of this study is to develop an optimal log sawing system for hardwood lumber manufacturing facilities. This paper mainly focuses on: 1) algorithms to determine the opening face and to improve lumber value recovery; 2) system development to implement the optimal computer algorithms; and 3) validation of the optimal sawing system by comparing computer-generated results and sawmill data.

MATERIALS AND METHODS

Twelve hardwood log samples were collected from the West Virginia University Research Forest in north-central West Virginia. Samples were taken from red oak (RO) (*Quercus rubra*) and yellow-poplar (YP) (*Liriodendron tulipifera*). The logs were all 8 feet in length and approximately 10-12 inches in diameter. To remove the influence of log sweep and crook on lumber value yields, the logs were chosen to be as straight as possible. Small- and large-end diameters were as close as possible to reduce the influence of log taper on lumber value yields. Six logs that met the above criteria were selected for each species.

Log profile data, including log length and small-end and large-end diameter, were taken. Other log characteristics such as log taper, sweep, and ovality were recorded (Table 1). Log external defect data was also collected, including: species; defect type, such as adventitious knot (AK), heavy distortion (HD), medium distortion (MD), light distortion (LD), overgrown knot (OK), sound knot (SK), and unsound knot (UK); linear position of defects from small end of log; defect's angle with respect to predetermined initial 0°; defect size; and defect surface rise. Log internal defect location was derived from Thomas' (2008) prediction model. Collected and predicted data were entered into a database.

Once all the logs were numbered and log data collected, the logs were transported to West Virginia University's band saw mill for processing into lumber. Boards sawn from the same log were numbered consecutively to track the source of the lumber, and measured after all sample logs had been sawn. Lumber length in feet, width and thickness in inches, and volume in board foot were then measured (Table 2). The grade and surface measurement of the lumber were determined by a National Hardwood Lumber Association (NHLA)-certified grader. The lumber values were based on the green rough lumber price across the state at the time of the assessment. The collected lumber data were used to compare with the simulation results generated from the optimal sawing system.

Table 1.—Characteristics of the sample logs.

Statistic categories	Small-end diameter (inch)	Large-end diameter (inch)	Taper (inch/foot)	Sweep (inch)	Ovality
Min	10.05	10.16	0.01	0.25	1.05
Max	11.96	12.34	0.07	1.13	1.12
Mean	10.84	11.1	0.03	0.53	1.08
Stdv ^a	0.62	0.7	0.02	0.03	0.03

^a standard deviation.

Table 2.—Characteristics of lumber from the sample logs.

Statistic categories	Width (inch)	Thickness (inch)	SM ^b	Volume (bf)	Value (\$)
Min	3.5	1	2	2.33	0.86
Max	8.44	2.25	6	10.69	5.34
Mean	6.1	1.42	4	5.58	2.27
Stdv ^a	1.12	0.16	0.9	1.44	0.88

^a standard deviation^b surface measure

OPTIMAL SAWING SYSTEM DESIGN

System Structure

The optimal sawing system consists of five major components: opening face, solid internal defect model, optimal log sawing, cant re-sawing, and lumber grading. This system was developed by utilizing object-oriented programming techniques with the Microsoft Foundation Class and Open Graphics Library (OpenGL). A relational database model with four entities (logs, defects, shapes, and lumber), was implemented using an entity-relationship model. The system accesses the database to retrieve data through an Active Data Object.

To provide users with a realistic log, 3D modeling techniques were used, together with OpenGL primitive drawing functions to generate three-dimensional log visualizations. A mathematical model was used to describe the geometry of the internal defects of a log (Thomas 2008). When a log was sawn, the locations and sizes of internal defects exposed on the surface of lumber could be determined using the developed procedures, and the lumber grade could be assigned accordingly. At this point, a cone model was used to represent internal defect, and its apex at the pith (central axis) of the log was assumed. When a sawing plane passed through an internal defect, a two-dimensional defect area was exposed on the surface. The defect area was mapped as rectangle on both sides of the lumber.

Sawing Algorithms

Opening face: Three steps were needed to determine the opening face. Log orientation needed to be determined first. When the defects were scattered over the entire surface of the log, they would be placed on the edge of the cutting plane. When defects were located on one portion of the log, they would be placed in one face. A procedure was developed to identify four log faces after orienting major defects at the edge of the cutting plane or in one face. The next step was to determine the best face. We assumed the opening face was cut from the best face that could be determined, based on log face grade. Each face was graded by a computer procedure based on the U.S. Forest Service hardwood log grading rules. The same grade will occur in more than one face of a log since there are only three grades (F1, F2, or F3). If the log has only one highest grade face, it is chosen as the best face. However, if there is more than one highest grade face, the face with the maximum clear area (curved-surface clear area) will be selected. Finally, the opening face dimension was determined. Lumber length was assumed to be the same as the log. The width of the opening face was determined as follows: if the grade of the best face was F1, the log would be slabbed to a width of 6.25 inches for logs of ≥ 13 inches in small-end diameter. For logs of < 13 inches in small-end diameter, the log would be slabbed to a width of 4.25 inches. For all logs that had a best face with grades of F2 or F3, the slab width was 3.25 inches.

Heuristic algorithm for log grade sawing: A heuristic algorithm was developed based on the Malcolm simplified procedure for hardwood log grade sawing (Malcolm 1965). Cutting from the small end, we set out the opening face full taper by using the small end of the log as the pivot, and the unopened face becomes parallel to the saw line. The log is not rotated unless one of the other log faces could yield a higher grade of lumber than could the current sawing face, in which case the log is rotated to the face with the potential for the highest lumber grade. After a log face has been cut completely, the grade of that face will be recorded, triggering the algorithm to move to the next face. This sawing process repeated until a specified size cant is produced, indicating that sawing is complete.

Mathematically-based algorithm for log grade sawing: In grade sawing, a log is broken into four portions at the small end (Fig. 1). For a given sawing orientation θ , a series of parallel cuts is performed along sawing orientation θ on portion 1 and 3 of the log. For portions 2 and 4, a series of parallel cuts with orthogonal orientation is used. When the first opening face was determined, θ was fixed correspondingly. A mathematical model for maximizing value through grade sawing can be expressed by the following function:

$$F = (L_1^*, L_2^*(L_1^*), L_3^*(L_1^*, L_2^*), L_4^*(L_1^*, L_2^*, L_3^*), V^*(L_1^*, L_2^*, L_3^*, L_4^*), S_1^*(L_1^*), S_2^*(L_1^*, L_2^*), S_3^*(L_1^*, L_2^*, L_3^*), S_4^*(L_1^*, L_2^*, L_3^*, L_4^*)) \quad (1)$$

where L_1 , L_2 , L_3 , and L_4 , and S_1 , S_2 , S_3 , and S_4 are the sawing planes and sawing patterns at each portion, respectively, V is the lumber value, and $*$ indicates an optimal value. This proposed model is based on an optimal log grade sawing procedure described by Bhandarkar and others (2008). The objective of function (1) is to find L_1 , L_2 , L_3 , and L_4 to maximize the total lumber value. For each portion, let $C = \{1, 2, \dots, n\}$ be a finite set of all the potential cutting planes, and $S = \{s_0, s_1, \dots, s_n\}$ is a subset of C that satisfies the following constraints:

$$(s_i - s_{i-1} - \left\lceil \frac{k}{c} \right\rceil) \cdot c \in T, \text{ for } 1 \leq i \leq n \quad (2)$$

$$s_0 = 1, s_n = n \quad (3)$$

where variables used in functions (2) and (3) are defined as following:

$T = (T_1, T_2, \dots, T_m)$ is a set of lumber thickness values (mm), it is an integer, and m is the total number of lumber thicknesses considered.

c is the sawing plane resolution (mm).

k is the kerf value (mm).

$n = \frac{CR}{c}$ is the total number of cutting planes within the cutting range, so the possible cutting planes are enumerated as $1, 2, \dots, n$, while CR is the cutting range between the opening face and central cant (mm).

A sawing pattern satisfying functions (2) and (3) will be seen as a feasible solution for log grade sawing as shown in Figure 2. Optimal sawing patterns can be determined by using a dynamic programming algorithm. Let $v^*(i)$ represent the optimal lumber value for each portion of the log between cutting planes 1 and i and $g(i, j)$ be the lumber value from the cutting planes i through j . Then a recursive mathematical equation for the dynamic programming can be formulated as follows (Bhandarkar and others 2008):

$$v^*(i+1) = \max_{j \in [0, m]} (v^*(i+1 - \frac{T_j}{c} - \left\lceil \frac{k}{c} \right\rceil) + g(i+1 - \frac{T_j}{c}, i+1)) \quad (4)$$

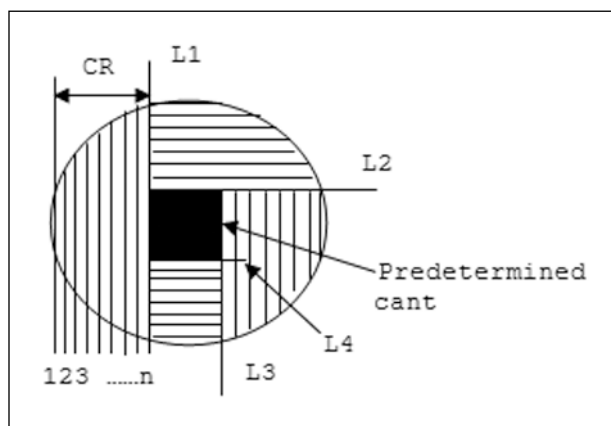


Figure 1.—All potential cutting lines for log grade sawing.

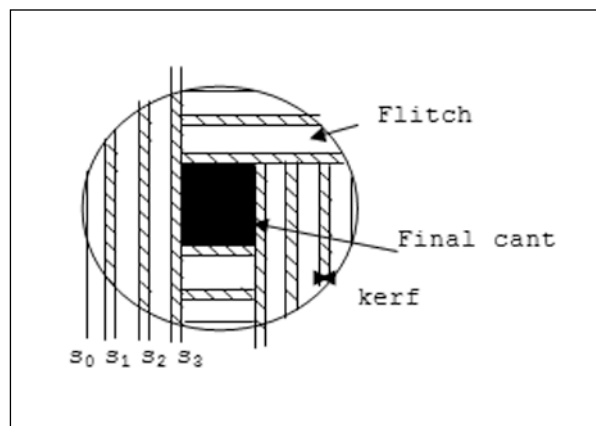


Figure 2.—A feasible solution for log grade sawing.

Cant Resawing

To compare the total lumber value from each log, the central cant must be sawn. Before sawing the central cant, we had to define a usable sawing region, and combine the given lumber thickness to obtain the best cutting solution. Because the taper sawing method was used, the initial cant is not square. So, the taper must first be removed from the cant. We assume the size of the cant at the small end of log to be the whole size of the final cant. Cant re-sawing can be done using live sawing or grade sawing; we used the former method in this study. The system saws the central cant from the best face of the log. Then, the system turns the central cant 90° clockwise and saws the cant again. The final sawing pattern was that which produced the largest total lumber value. During heuristic log sawing, no optimization algorithm is used as it simply makes a series of parallel cuts. While the dynamic programming algorithm was used to saw the log, the problem of central cant re-sawing can also be solved by the optimal algorithm used for live sawing. The cant re-sawing and log grade sawing parameters are the same with the exception of log sawing orientation and cutting range.

Lumber Grading

Before the lumber is graded, the sawn flitch must be re-sawn into lumber. All flitches were edged in this study to remove all wane. To generate this lumber pattern, the width of the lumber from the lowest width end is used as the width. The edged lumber is graded by computer algorithm based on the NHLA grading rules and a hardwood lumber grading program (Klinkhachorn and others 1988). Lumber grades used were First and Seconds (FAS), FAS-One-Face (F1F), Select, 1Common (1COM), 2Common (2COM), and 3Common (3COM).

RESULTS AND DISCUSSION

By specifying some parameters (i.e., kerf width, lumber thickness, cant size), users can interactively simulate the sawing process. In this study, sawing kerf width was chosen as 1/8 inch and lumber thickness was 1-3/8 inch, the same as the parameters used in the sawmill. The cant size was chosen as 4 x 6 inches since it is the most common in West Virginia sawmills (McDonald and others 1996).

OPENING FACE

To determine whether the optimal opening face derived by the procedure was better than other opening faces chosen randomly, we simulated log sawing based on the optimal opening face and other assumed opening faces from 0 to 85° in 5° increments. During the simulation, a series of parallel cuts was made on each of the four faces until a central cant remained, and then the central cant was also sawn by parallel cuts. Table 3 summarizes the results. When a log was cut from the optimal opening face, total lumber value was higher than the mean value by an average of 2.9 percent. At the best rotation angle, the maximum lumber value was 6.2 percent higher than the mean value. At the worst rotation angle, the minimum value was approximately 6.3 percent lower than the mean value.

The results suggested that lumber value can be improved based on opening face chosen (Table 3). However, the lumber value produced from the optimal opening face was not always the maximum. Only 25 percent of the lumber value produced from the optimal opening face cutting equaled the maximum value. It was also noted that the lumber value produced from the optimal opening face derived by algorithm was lower than the average lumber value produced from the 3rd and 10th logs. The reason for this shortfall may be that the log orientation determined by the opening face algorithm does not consider defect types and sizes. For example, the penetration depth and clear area between bark and defect area are different based on defect type. Severe defects may have more significant effects on lumber value than slight defects. Thus the major defects must be considered first and rotated to the edge of the log sawing planes. Therefore, the optimal opening face found was not the best opening face.

ACTUAL VS. OPTIMAL LOG SAWING

Comparisons between the actual lumber value obtained from the sawmill and the simulated solutions are presented in Table 4. In terms of average lumber value recovery, a sawmill could improve lumber value 22.02 percent and 33 percent if they used the heuristic and dynamic programming algorithms, respectively. If the average value of lumber produced were priced at 50 cents per board foot and 1 million board feet were sawn annually, the gain in lumber value could be as high as \$110,000 to \$165,000 per year. Additionally, recovery of high volume does not always lead to recovery of high lumber value. For example, when heuristic and

Table 3.—Lumber value vs. log rotation.

Log No.	Species	No. of defects	Opening ^a (\$)	Max ^b (\$)	Min ^c (\$)	Mean ^d (\$)
1	YP	5	19.79	19.79	18.72	19.55
2	YP	7	19.42	19.91	17.44	18.68
3	YP	11	14.73	17.48	13.73	15.75
4	YP	11	16.08	17.25	12.96	15.54
5	YP	10	23.8	25.38	21.01	22.8
6	YP	9	22	22	19.94	20.53
7	RO	4	28.66	29.28	26.1	27.86
8	RO	6	29.55	30.09	25.14	28.52
9	RO	7	27.36	27.36	23.53	24.89
10	RO	8	25.45	26.75	24.83	25.7
11	RO	6	32.42	32.74	32.04	32.25
12	RO	6	34.5	34.97	31.86	33.34

^a Determined by opening face algorithm.

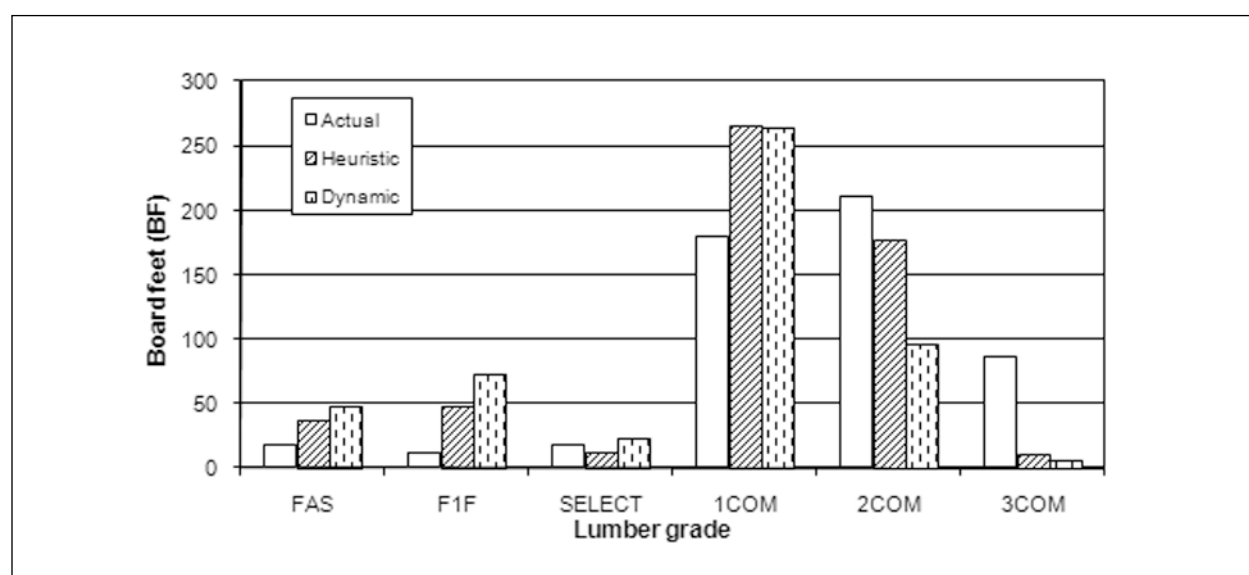
^b, ^c, and ^d are the maximum, minimum, and average of lumber value at 18 log rotations, respectively.

Table 4.—Summary of log sawing simulation.

Log No	Actual			Heuristic			Dynamic		
	Volume (bf)	Value (\$)	\$/bf	Volume (bf)	Value (\$)	\$/bf	Volume (bf)	Value (\$)	\$/bf
1	43.77	17.06	0.39	45.27	20.12	0.44	38.04	20.26	0.53
2	35.56	14.27	0.4	40.08	19.42	0.48	37.65	19.53	0.52
3	36.34	21.67	0.6	38.54	28.66	0.74	38.84	29.26	0.75
4	45.22	26.44	0.58	46.54	32.35	0.7	43.65	32.93	0.75
5	35.69	10.4	0.29	42.69	14.73	0.35	39.94	14.89	0.37
6	37.07	11.29	0.3	41.75	16.08	0.39	39.08	16.53	0.42
7	40.71	20.97	0.52	40.08	27.36	0.68	40.05	28.57	0.71
8	36.92	21.44	0.58	38.75	25.45	0.66	34.06	25.58	0.75
9	56.74	18.9	0.33	57.16	23.8	0.42	53.06	23.8	0.45
10	52.23	18.38	0.35	53.32	22.18	0.42	48.56	22.67	0.47
11	51.91	30.46	0.59	50.01	35.3	0.71	46.53	35.88	0.77
12	52.23	27.68	0.53	52.89	35.06	0.66	48.51	35.79	0.74
Mean	43.67	19.91	0.46	45.59	25.04	0.55	42.33	25.47	0.61

dynamic programming algorithms were used to saw the logs, the average volume of lumber was 45.59 bf and 42.33 bf, respectively. However, the average value of lumber was \$25.04 and \$25.47, respectively.

The distribution of lumber grade produced by using algorithms and actual lumber grade production is shown in Figure 3. We found that the distribution of lumber grades differs among different sawing methods. In the actual log sawing production, 34 percent, 40 percent, and 17 percent of lumber were graded as 1COM, 2COM, and 3COM, respectively. When the heuristic algorithm is used to saw those logs, 48 percent, 32 percent, and 2 percent of lumber were graded as 1COM, 2COM, and 3COM, respectively. When the dynamic programming algorithm was used, 52 percent, 19 percent, and 1 percent of lumber were graded as 1COM, 2COM, and 3COM, respectively. About 8 percent, 18 percent, and 27 percent of lumber grade were Select or higher among the actual production, heuristic algorithm, and dynamic programming algorithm, respectively.

**Figure 3.—Lumber grade distribution.**

The results suggest that the difference in lumber value produced between actual sawmill and optimal sawing algorithm is rather large. One of the possible reasons is that real sawmill production depends on the sawyer's decisions. Such decisions are usually subjective and depend on the sawyer's skills and experience. Another reason is that external defects do not always indicate internal defects, especially for yellow-poplar. For example, only 7 external defects were found for log 2, but there were 10, 14, and 16 defects found in three pieces of lumber sawn from that log. In addition, some parts of the logs have no bark due to inappropriate operation or longtime storage, which affects the identification of external defects and predicted model accuracy. Finally, the precision of the predicted internal defect model also contributed to the difference between real production and simulations.

SPECIES COMPARISONS

In this study, yellow-poplar had more external defects than red oak (Fig. 4). Overgrown knots made up approximately one-third of observed defects for both species. In actual production, the average lumber value per board foot for yellow-poplar and red oak was \$0.35/bf and \$0.57/bf, respectively. However, if the heuristic algorithm is used to saw those logs, the average of lumber value per board foot could be \$0.42/bf and \$0.69/bf, respectively. If the dynamic programming algorithm is used, the average lumber value per board foot could be \$0.46/bf and \$0.75/bf, respectively. The improvement of lumber value per board foot using heuristic algorithm was 7 percent and 12 percent for yellow-poplar and red oak when compared to the actual production, respectively. The improvement of lumber value per board foot using the dynamic programming algorithm was 11 percent and 18 percent for yellow-poplar and red oak, respectively. Because the range of price change for yellow-poplar lumber at different grades is smaller than for red oak, and more defects were found in yellow-poplar logs, it makes sense that the improvement of lumber value recovery for yellow-poplar is smaller compared to red oak.

While lumber value has been improved by using the optimal sawing system, we must note some limitations. When the dynamic programming algorithm is used to optimize lumber value, the best cutting plane resolution is 0.04 in. In this study, the resolution was chosen as 0.16 in because run times for the program

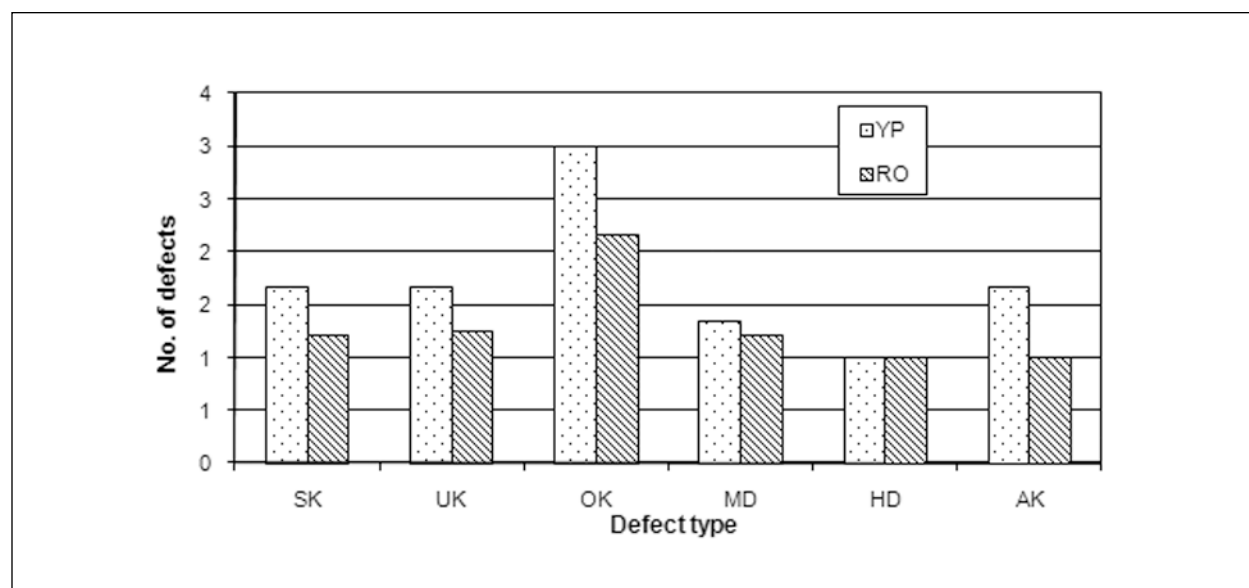


Figure 4.—Average log external defects by species. See page 68 for explanation of abbreviations.

are too great for this complicated algorithm. For example, if there were 75 cutting lines at each face of a log, there were approximately 7.6×10^{18} possible choices to consider. With an increase in the small-end diameter, the calculations are overwhelming. All flitches produced from log breakdown were edged to remove all waste. This processing method does not guarantee achievement of maximum value since flitch edging has significant effects on final lumber value. Using the optimum algorithm to deal with the flitch edging problem is essential to increase total lumber recovery. Additionally, the virtual shape of a log is not uniform. Thus, the precision of log sawing simulation is limited by using a truncated cone shape to represent the logs.

Future improvements that would extend the current findings include: 1) considering the real shape of the log in the sawing optimization system; 2) involving more variables, including external defects type and size as well as internal defects in the opening face cutting to improve the accuracy of the opening face cut; 3) combining the log sawing and flitch edging optimization; and 4) using multiple lumber thicknesses during the log sawing process.

CONCLUSIONS

A 3D log sawing optimization system was developed to perform 3D log generation, opening face determination, sawing simulation, and lumber grading. The system was then applied to 12 sample sawlogs at a sawmill in West Virginia. Comparisons between the actual lumber values and the simulated results from the same log were presented.

Preliminary results have shown that a typical hardwood sawmill potentially can increase lumber values by rotating logs and optimizing sawing patterns. In this study, we found that lumber value can increase 2.9 percent when opening face cutting is used, as compared to the average of lumber value produced from the 0 to 85° rotation. In terms of the average of lumber value recovery, sawmills have the potential to improve lumber value by 22 percent or 33 percent with use of the heuristic or dynamic programming algorithm, respectively. The lumber grade was improved from 2COM or 3COM to 1COM or higher grade by using the optimal algorithms. The results also indicated that high lumber value recovery does not necessarily mean high volume recovery.

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

VALIDATION OF AN INTERNAL HARDWOOD LOG DEFECT PREDICTION MODEL

R. Edward Thomas¹

Abstract.—The type, size, and location of internal defects dictate the grade and value of lumber sawn from hardwood logs. However, acquiring internal defect knowledge with x-ray/computed-tomography or magnetic-resonance imaging technology can be expensive both in time and cost. An alternative approach uses prediction models based on correlations among external defect indicators and internal defect features. Using external defect feature measurements, a prediction of internal defect size, shape, and depth can be generated. This paper examines the accuracy of the prediction models by comparing defect attributes on actual sawn board faces with the predicted defect attributes on virtual sawn boards. Although the models showed significant correlations in the model-testing dataset for most defect types and features, they were never tested by the sawing of new samples and comparing predicted to actual defect attributes. Results for a test sample of 41 boards with 83 observed knot defects show that the prediction models can predict the occurrence of approximately 80 percent of all knot-type defects.

INTRODUCTION

The grade and value of lumber sawn from hardwood logs depend on the type, size, and location of internal defects. The hardwood log grading rules are based on the relationship between defect occurrence and lumber recovery (Hanks and others 1980). The ability to use internal defect information such as location, size, and shape during the sawing process has been shown to significantly improve the value of hardwood lumber sawn by as much as 10 to 21 percent (Wagner and others 1990, Steele and others 1994). X-ray/computed-tomography (CT) and magnetic-resonance imaging (MRI) methods have been the traditional research approach to obtaining internal log defect information (Chang 1992). However, there are several problems associated with acquiring internal hardwood log defect data using these methods. For example, scanning a log can be costly due to equipment expenses and time-consuming, depending on resolution and processing speed (Schmoldt and others 2000).

Because of the importance of internal defect information to the sawing process, several studies have sought to link external features to internal defect characteristics. Schultz (1961) studied German beech (*Fagus sylvatica* L.) and found that the ratio of the bark distortion width to length was the same as the ratio of the stem when the branch stub was encapsulated to the current stem diameter. For species with heavier, irregular bark, such as sugar maple (*Acer saccharum* L.), however, Schultz (1961) found that it was difficult to judge the clear area above the defect using this method. Like Schultz (1961), Shigo and Larson (1969) found that for many hardwoods the ratio of defect length to width indicated the depth of the defect with respect to the radius of the stem at the defect. Hyvärinen (1976) studied external bark distortion measurements and defect encapsulation depth for sugar maple and found strong correlations among encapsulation depth and distortion

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width, length, and rise using linear regression methods. More recently, Lemieux and others (2001) examined knot defect correlations among external indicators and internal defect features for a sample of 21 black spruce (*Picea mariana* L.) trees collected in northern Quebec. They found correlations ($r > 0.89$) among the width and length of internal defect areas and external features such as branch stub diameter and length. The Lemieux and others (2001) study examined only branches that had not been dropped or pruned, thus preventing an examination of encapsulation depth.

An alternative approach to the x-ray/CT and MRI methods has been developed that uses high-resolution laser surface scanning. This approach uses intelligent software to locate and classify external log defects in the three-dimensional (3D) laser data (Thomas and others 2006). Once the external log defects are located, predictions of internal characteristics are made which estimate internal defect position, shape, and size based on measurements of the external indicator and log diameter (Thomas 2008, 2009). These models are based on correlations among internal and external defect features. Different models were created for different species, defect types, and defect type groupings. The models were created by statistically analyzing 842 red oak (*Quercus rubra* L.) and 1,000 yellow-poplar (*Liriodendron tulipifera* L.) defect samples collected from three sites in the central Appalachian region. These sites were the West Virginia University Experimental Forest; a forest near Rupert, WV; and the Camp Creek State Forest in West Virginia.

METHODS

A red oak tree was randomly selected from a site on the Fernow Experimental Forest near Parsons, WV. This experimental fire site had been subjected to fire twice in the last 15 years. The selected tree was bucked into four logs and a forked jump cut, which was discarded. Due to the site's burn history, the base of the tree had a large burn and decay area measuring approximately 66 x 26 inches with a maximum depth of 3.625 inches. The logs were graded to U.S. Department of Agriculture, Forest Service, log grades (Rast and others 1973). Despite the burn scar, the butt log was a grade 1 log. The second and third logs were grade 2 logs, and the fourth was a grade 3 log. Table 1 contains more detail about the size and defect characteristics of these logs. Because the butt log had no surface knot defects, it was omitted from the study.

Log surface data were collected using a high-resolution laser log scanner. The scanner is composed of three laser scan heads positioned at 120-degree intervals around the log. Each log is held on steel supports as the scanner mechanism traverses the length of the log. Each log scan is composed of series of scan lines around the circumference of the log. The distance between scan lines is 1/16 (0.0625) inch and each line is composed of approximately 300 to 600 points, depending on log size. The scan data, in their raw format, can be viewed as a dot cloud image (Fig. 1).

Table 1.—Characteristics of defect study logs.

Log	Grade	Length (feet)	Small end diameter (inches)	Large end diameter (inches)	Defect characteristics			
					Num. of knots	Num. of bark distortions	Num. of wounds	Num. of other defects
1	1	12	19.50	27.50	0	0	2	0
2	2	16	17.25	19.50	3	10	4	0
3	2	16	15.00	17.25	7	4	0	2
4	3	8	9.75	9.75	9	2	1	4

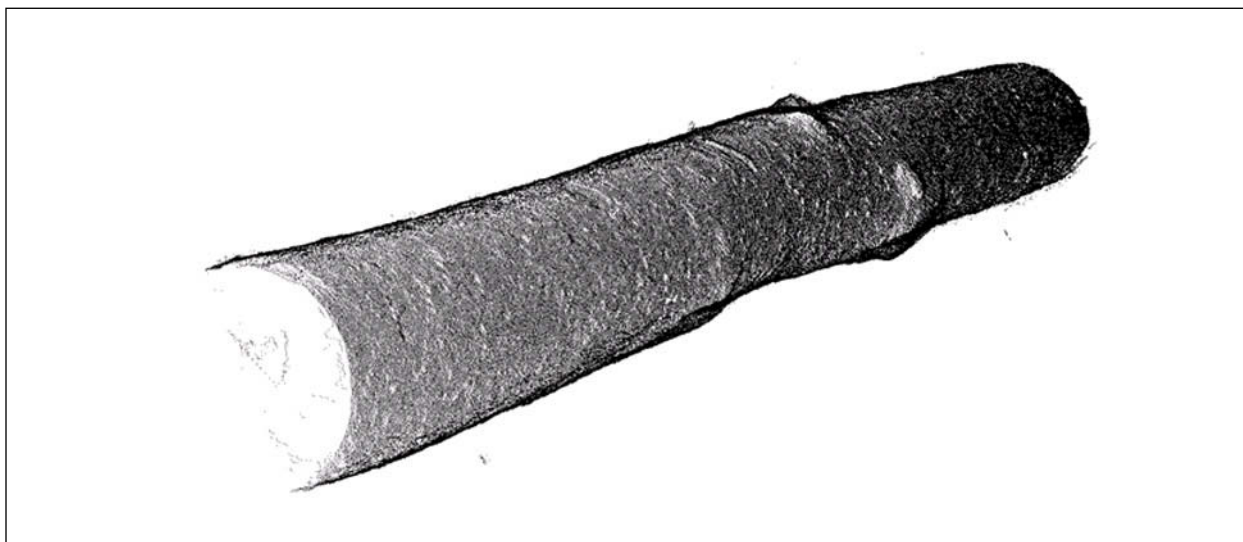


Figure 1.—Dot cloud image of a laser scanned log (Log 2).

After each log was scanned, all defect sizes and positions were recorded. This provided an accurate map of external defects and was used by the defect prediction program to generate internal defect size, shape, and position data. Defects are modeled as elliptical cones with two regions centered about the line from the center of the surface indicator to the predicted innermost endpoint determined using the predicted depth and penetration angle. The inner region extends from the predicted midpoint. The penetration angle is the angle at which a line through the center of the defect intersects the log surface. For some types of defects, most notably the bark distortions, the defects are grown over or encapsulated. Encapsulation depth is the amount of clear wood that has grown over the internal defect.

The internal and surface defect structure coordinates were overlaid with the scan data, creating a completely defined log dataset. These data were processed using the RAYSAW sawing program² (Thomas 2009), which reads a sawing pattern file and generates digital boards. Defects on the virtual boards are represented by boxes. The width and length of a box represent the size and approximate shape of the defect as it intersects the board face. To allow comparison of knots between actual and digital board faces, each log was sawn using the same sawing pattern as was used with its digital version.

The sawing patterns were designed to maximize the number of defects on the board faces rather than on the edges. This step was taken to simplify the comparison of defects on actual and digital board faces. To perform the comparison, defects were tracked from the surface indicator to their termination in both the digital and real-world logs. Only the knot-type defects on the board surfaces facing outward from the center of the log were compared. The specific defect types traced were:

Light Distortion	Medium Distortion	Heavy Distortion
Sound Knot	Sound Knot Cluster	Overgrown Knot
Overgrown Knot Cluster	Adventitious Knot Cluster	Unsound Knot

Given the 3/32-inch kerf, there is little difference in defect characteristics from the back side of the outermost board to the face side of the next board sawn.

²Thomas, R.E. 2009. RAYSAW: Ray-tracing based log sawing program for 3D laser profile data. Unpublished computer program. U.S. Forest Service, Northern Research Station, Princeton, WV.

RESULTS

Forty-one boards were sawn from three sample logs. Three categories of results were observed: 1) defect was observed on the board face, but not predicted by the models; 2) defect was predicted but was not observed on the board face; and 3) the observed defect was predicted by the models. If no defects were observed on the actual board face and no defects were predicted, this was counted as a match. This situation occurred on two boards.

The outermost or first face of the boards contained 83 knot defects. The length and width of all observed knots were measured and knots traced to their surface indicator. In addition, the lineal position of each defect was determined by measuring the distance from the center of defect to the end of the board. Of the 83 knot defects observed on the board faces, the model predicted the appearance of 67, or 80.7 percent of the population. The median of the absolute value of the difference between actual defect surface size versus predicted size was 5.6 square inches. However, in many cases the actual error was much smaller. In 15 instances (22.4 percent) the actual versus predicted size error was less than 1 square inch. In 22 instances (32.8 percent) the error was less than 2.5 square inches. The minimum size error observed was 0.02 square inch, and the maximum was 42.81 inches. The large maximum error size was due to a single overgrown knot cluster that appeared on four board faces which had a large surface indicator but a small internal defect. If these samples are omitted from the analysis, the median absolute size error becomes 4.94 square inches. This size of error corresponds approximately to 2.22 inches in defect width and length.

For the 67 defects which were predicted by the models, the median of the absolute value of the lineal position errors was 0.875 inch (measured from the center of the actual defect to the predicted location). In 35 instances or 52.2 percent of those predicted, the mean error was 1.0 inch or less. In 49 instances, or 73.1 percent of those predicted, the mean error was less than 2.5 inches. The minimum observed error was 0 inches while the maximum was 21.25 inches. The maximum error occurred on a sound knot which had a very slight rake. Here the penetration angle into the log was nearly vertical. The model predicted a more normal occurrence of the rake angle, thus the error in position.

The other types of errors in the analysis were instances where there was a defect predicted but not an actual defect, or vice-versa. These are summarized by defect and prediction error types in Table 2. In 14 instances, a defect was found on the board face but was not predicted. Ten of these occurrences were with distortion defects while the remainder occurred with sound and overgrown knots and clusters. A greater number of defects, 49 occurrences, were predicted but were not found on the actual board faces. Thirty-one instances of these errors were with the distortion-type defects; the model often predicted that the defect penetrated more deeply or that the defect was not encapsulated as deeply as it was in actuality.

DISCUSSION

This was the first test of the models in predicting the location and size of knot defects on a randomly selected tree. Overall, the models predicted the location of 67 of 83 board defects for an accuracy rate of 80.7 percent. While missing less than 20 percent of all defects, the models predicted 49 more defects which did not appear on the board faces. In these cases the models over-estimated the severity of the surface indicators and under-estimated the encapsulation depth of the defects, making the model predictions a conservative estimate of potential lumber quality and value.

Table 2.—Prediction error counts by defect and error type.

Defect type	Number of defects observed but not predicted	Number of defects predicted but not observed
Heavy distortion	6	19
Light distortion	4	3
Sound knot	1	1
Overgrown knot	2	11
Overgrown knot cluster	1	1
Medium distortion	0	9
Adventitious knot cluster	0	3
Unsound knot	0	2
Total	14	49

In general, the models performed well at predicting the location of knots within the tree, that is, the location of the defects on the resulting sawn board faces. The median error of the measured distance between the actual defect location and the predicted defect location was 0.875 inch. Given the sizes of the knot defects, the predicted and actual defect areas often overlapped. Thus, the impact of defect location error on predicted grade or yield should be small.

The models's predictions for internal defect size were not as accurate, but still acceptable in most instances. In 22 cases (32.8 percent) the error was less than 2.5 square inches. This error approximately corresponds to an error of 1.5 inches in defect length and width. The minimum size error observed was 0.02 square inch, and the maximum was 42.81 inches. Overall, the median error for predicted defect area on the board faces was 5.59 square inches. This size of error is approximately 2.4 inches in defect width and length.

The mean absolute error for all internal defect feature predictions was determined during model development. One question arises: Given these errors, what impact could they have on the predicted grade or value of the lumber sawn from logs with predicted defect sizes and locations? This question is currently being addressed and preliminary results indicate that in almost all cases the current prediction error level does not have a significant impact on lumber recovery value. In addition, this initial study needs to examine a greater sample pulled from a wider geographic location. This research will further test the prediction models and compare the value and yield of actual boards to predicted boards.

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INTEGRATED LEAST-COST LUMBER GRADE-MIX SOLVER

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Abstract.—Hardwood lumber costs account for up to 70 percent of the total product costs of U.S. secondary wood products producers. Reducing these costs is difficult and often requires substantial capital investments. However, lumber-purchasing costs can be minimized by buying the least-cost lumber grade-mix that satisfies a company's component needs. Price differentials between National Hardwood Lumber Association lumber quality classes (grades) change based on market pricing and thus a different least-cost lumber grade-mix solution may be obtained at different times. The U.S. Forest Service's ROMI free rough mill simulation package contains a least-cost grade-mix solver that allows hardwood lumber processors to find the least-cost lumber grade mix. This software, however, currently requires calculations performed on a Forest Service server using the SAS statistical package. R, a new open-source statistical software package similar to SAS, allows the least-cost lumber grade-mix solver to run on the users' computer without a connection to a Forest Service server. Use of the program is thus simplified, and company information is kept confidential.

INTRODUCTION

The U.S. manufacturing sector has seen challenging years over the past decades, with manufacturing employment reaching its lowest level since 1950 (Forbes 2004). The major driver behind the decline of U.S. manufacturing prowess is the ongoing globalization of trade, which has brought the comparative disadvantage of the United States as a manufacturing location to the forefront. Unfortunately, many U.S.-based hardwood lumber-consuming industries, such as furniture, flooring, and millwork industries, are hard hit by the worldwide opening of markets for goods and services (Schuler and Buehlmann 2003, Buehlmann and others 2004a, Buehlmann and Schuler 2009). With the decline in manufacturing operations producing hardwood-based products comes a decline in the demand for hardwood lumber, affecting the industries and forest landowners upstream from the manufacturer (Grushecky and others 2006).

According to U.S. Forest Service calculations, approximately 5.3 billion board feet of hardwood lumber is consumed annually in the United States for the production of wooden products other than pallets (i.e., appearance-grade lumber). Assuming that the average cost of this lumber is \$750 per thousand board feet (MBF), the nationwide total lumber expense to rough mills is approximately \$4 billion. An increase in operating efficiency (yield) of 1 percent would reduce the amount of lumber required by 53 million board feet with potential lumber-cost savings of \$40 million, on an aggregate level. On an individual manufacturing location level, Kline and others (1998) estimated that a 1- to 2-percent yield increase in an average rough mill saves \$150,000 to \$300,000 annually. However, increasing operation efficiency often requires investment before savings can be captured. One immediate step to increase the competitive position of producers using domestic hardwoods is to enable them to find the lowest raw material cost solution possible, a problem

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referred to as the least-cost lumber grade-mix problem. This solution, unlike the quest for increased efficiencies mentioned above, requires only minimal capital outlays, can be implemented immediately, and can save significant amounts of capital.

In the past decades, researchers have developed several programs to solve the least-cost lumber grade-mix problem (Englerth and Schumann 1969, Hanover and others 1973, Martens and Nevel 1985, Carino and Foronda 1990, Steele and others 1990, Timson and Martens 1990, Harding 1991, Fortney 1994, Suter and Calloway 1994, Lawson and others 1996, Hamilton and others 2002). Buehlmann and others (2004b, 2010) and Zuo (2003) created the least-cost lumber grade-mix solver, a statistical solution that answered several shortcomings of older models (Zuo and others 2004). Buehlmann and others' (2004b, 2010) solution has been shown to perform better than other optimizers (Buehlmann and others 2008). The U.S. Forest Service has incorporated Buehlmann and others' (2004b) least-cost lumber grade-mix solver solution into the widely used ROMI rough mill simulation software (Weiss and Thomas 2005, Thomas and Weiss 2006). However, Buehlmann and others' (2004b, 2010) least-cost lumber grade-mix solver relies on advanced statistical software (SAS Institute 2009) to build a polynomial response surface to find the least-cost solution. Thus, a user employing ROMI and running the least-cost lumber grade-mix solver needs a working internet connection to run the least-cost optimization hosted on a U.S. Forest Service server equipped with the SAS statistical package (SAS 8.2, 2002, SAS Institute, Cary, NC). This set-up forces industry users to be connected to the Internet and the U.S. Forest Service to maintain a server running the SAS software; industry users also must pay licensing fees to SAS. The industry participant sends yield data derived on the user's local computer to a Forest Service server. The Forest Service computer then derives the least-cost lumber grade-mix solution and returns it to the user.

However, thanks to efforts within the open-source software movement (Anonymous 2009), R 2.7.2, a statistical software package (Venables and others 2008) with capabilities similar to SAS, has become available for free and can be installed on individual computers at will. The research reported below investigates the replacement of SAS 8.2 with R 2.7.2 (Venables and others 2008) in the least-cost lumber grade-mix solver software (Buehlmann and others 2004b, 2010) so that users no longer need to connect to a Forest Service server running SAS.

MATERIALS AND METHODS

This project required the use of the U.S. Forest Service's rough mill simulator, Romi-Rip 2.0 (Thomas 1999a,b) and ROMI 3.0 (Weiss and Thomas 2005, Thomas and Weiss 2006), SAS 8.2, and R 2.7.2 (Venables and others 2008), the open source statistical package freely available on the internet.

ROUGH MILL SIMULATOR

The original least-cost lumber grade-mix solver (Buehlmann and others 2004b, 2010) relied on the U.S. Forest Service's ROMI-RIP 2.0 (Thomas 1999a,b) rough mill rip-first simulator. Thus, ROMI-RIP 2.0 was used to find the least-cost lumber grade-mix when using the SAS 8.2 statistical analysis package. With the R 2.7.2 (Venables and others 2008) statistical analysis package, however, it was decided to use ROMI 3.0 (Weiss and Thomas 2005, Thomas and Weiss 2006). While this approach may introduce variability not associated with the switch of statistical software package (from SAS 8.2 to R 2.7.2) because of improvements made to the rough mill simulator (Weiss and Thomas 2005), the research team first wanted to know whether a user can expect comparable results from the older (ROMI-RIP 2.0, SAS 8.2-based) and the newer

(ROMI 3.0, R 2.7.2-based) version of the least-cost lumber grade-mix solver. If needed, future research could resolve any source of variability observed. The simulators, ROMI-RIP 2.0 and ROMI 3.0, employ the same operational settings, i.e., use the all blades movable arbor, obtain parts salvaged through additional processing, use complex dynamic exponential component prioritization (Thomas 1996), and produce no excess and random-length or random-width components.

LUMBER

Data from Gatchell and others' (1998) Kiln-Dried Red Oak Data Bank containing digitized representations of red oak boards (Anderson and others 1993) were employed for this study. Using the Gatchell and others (1998) database, we created randomly composed lumber datafiles for each grade-mix needed in the experiment. To create the necessary lumber datafiles, the MAKEFILE tool in ROMI-RIP 2.0 and ROMI 3.0 was employed (Thomas 1999, Weiss and Thomas 2005). The lumber grades involved were First and Seconds (FAS), Selects (SEL), 1 Common (1C), 2 Common (2C), and 3A Common (3AC; National Hardwood Lumber Association 1998). To assure consistency with Buehlmann and others' (2004b, 2010) and Zuo's (2003) work, the following 2002 prices for red oak lumber found in Hardwood Review Weekly (2002) were used: \$1,570 per MBF for FAS, \$1,350 per MBF for SEL, \$1,000 per MBF for 1C, \$748 per MBF for 2AC, and \$500 per MBF for 3AC.

CUTTING BILL

For these initial performance assessments, a cutting bill (cutting bill A) used by Thomas (1996) was selected. Table 1 shows details of the cutting bill requirements.

Table 1.—Part sizes and quantities required by cutting bill A.

Width (inches)	Length (inches)	Quantity (units)
2.00	25.50	625
2.25	20.75	750
2.25	27.00	2,250
2.75	16.25	1,500
2.75	32.50	1,500

STATISTICS

A mixture design, a special response surface methodology typically used for optimizing multiple factors simultaneously, was employed (Giesbrecht and Gumpertz 2001). Each individual lumber grade (FAS, SEL, 1C, 2AC, 3AC; National Hardwood Lumber Association 1998) can contribute between zero and 100 percent of the total lumber used and the sum of all lumber grade proportions is one. Since each grade is an individual factor, a five-factor mixed design is established. The 3AC lumber grade was restricted to a maximum contribution of 80 percent since 3AC rarely is able to contribute all the large parts required by a typical cutting bill, resulting in skewed yield results (Zuo 2003, Buehlmann and others 2004b).

Minimizing costs and not optimizing yields is the objective of the optimization. Thus, the least-cost lumber grade-mix solver converts the yields obtained from 25 test runs conducted by the rough mill simulator (Table 2; Buehlmann and others 2004b, 2010) to lumber costs plus processing costs using Equation 1 (Zuo 2003). Equation 2 then creates a total cost (lumber plus processing costs) response surface employing a second order polynomial equation (Zuo 2003).

$$COST_j = \frac{\sum_i^5 G_i * (M_i + P_i)}{YIELD_j} \quad (1)$$

where: G_i - the proportion of each lumber grade
 M_i - the market price per thousand board feet of each lumber grade
 P_i - the processing cost per thousand board feet of each lumber grade
 i - 1 for FAS, 2 for SEL, 3 for 1 Common, 4 for 2A Common, and 5 for 3A Common
 j - observation of a grade combination run

Table 2.—Twenty-five lumber grade-mix combinations used to establish cost response surface.

RUN	FAS	SEL	1C	2AC	3AC
1	0	0	0	20	80
2	0	0	0	60	40
3	0	0	0	100	0
4	0	0	20	0	80
5	0	0	50	50	0
6	0	0	50	50	0
7	0	0	60	0	40
8	0	0	100	0	0
9	0	20	0	0	80
10	0	50	0	50	0
11	0	50	0	50	0
12	0	50	50	0	0
13	0	50	50	0	0
14	0	60	0	0	40
15	0	100	0	0	0
16	50	0	0	50	0
17	50	0	0	50	0
18	50	0	50	0	0
19	50	0	50	0	0
20	50	50	0	0	0
21	50	50	0	0	0
22	60	0	0	0	40
23	60	0	0	0	40
24	100	0	0	0	0
25	100	0	0	0	0

$$\mu_y = \beta_0^* + \sum_{i=1}^5 \beta_i^* x_i + \sum_{i < j} \beta_{ij}^* x_i x_j \quad (2)$$

where: μ_y - the cost of satisfying a given cutting bill
 x_i, x_j - the proportions of each lumber grade
 β_0^* - the intercept
 β_i^* - the coefficient of linear terms
 β_{ij}^* - the coefficients of the interaction terms
 i, j - 1 for FAS, 2 for SEL, 3 for 1 Common, 4 for 2A Common and 5 for 3A Common

Using the cost response surface created with Equations 1 and 2, the minimum cost location is identified and the respective grade or grade-mix is returned. While SAS 8.2 and R 2.7.2 (Venables and others 2008) use different commands for this purpose, both statistical packages use the same underlying methodologies.

EQUIVALENCE TESTING OF SOLUTIONS

In a first set of tests, it was decided to compare least-cost lumber grade-mix solutions using ROMI-RIP 2.0 (Thomas 1999a,b) and SAS 8.2 with solutions from using ROMI 3.0 (Weiss and Thomas 2005, Thomas and Weiss 2006,) and R 2.7.2 (Venables and others 2008). For both tests, cutting bill A was employed and three repetitions were conducted using the same three lumber datafiles. No statistical comparison of results was

planned, as future tests are currently under way that will add to the insight gathered from this study and allow more encompassing conclusions to be made.

RESULTS AND DISCUSSIONS

The least-cost lumber grade-mix solution derived with the ROMI-RIP 2.0 and SAS 8.2 found a least-cost lumber grade-mix with a cost of \$2,101 per MBF with an average yield of 57.11 percent (three repetitions) (Table 3). Using the same datafiles, the ROMI 3.0 and R 2.7.2 found a least-cost lumber grade-mix with a cost of \$2,096 per MBF with an average yield of 38.18 percent (three repetitions). The cost difference between these two tests was \$5 per MBF, a difference of less than 1 percent. The yield difference between these two solutions, however, was more than 18 percent: ROMI-RIP and SAS 8.2 generated an average yield of 57.11 percent and ROMI and R 2.7.2 generated an average yield of 38.18 percent.

Table 3.—Least-cost lumber grade-mix solutions and yields for ROMI-RIP 2.0 and SAS 8.2, and ROMI 3.0 and R 2.7.2.

Test	Yield (%)	Minimum Cost (\$ per MBF)
ROMI-RIP & SAS 8.2	57.11	2,101
ROMI & R 2.7.2	38.18	2,096
Difference	18.93	5

The results derived by the two least-cost lumber grade-mix models using the same underlying statistical methodology resulted in two fundamentally different solutions to the problem. Despite the different solutions, however, the cost difference was less than 1 percent (Table 4). The least-cost lumber grade-mix solution from the ROMI-RIP 2.0 and SAS 8.2-based solver relied on 1C for obtaining all the parts. The ROMI 3.0 and R 2.7.2-based least-cost lumber grade-mix solver solution found an almost equivalent least-cost lumber grade-mix solution (difference of less than \$5 per MBF or less than 1 percent) by using 80 percent 3AC and 20 percent 1C lumber, creating a solution with lower yield but also lower costs (Table 4). This solution requires the use of 20 percent higher grade lumber (i.e., 1C) to obtain the larger parts (long and/or wide) required by cutting bill A and cuts the remaining parts from 80 percent of the lowest-grade lumber, 3AC. These results indicate that the two least-cost lumber grade-mix solvers perform equally well in finding a least-cost lumber grade-mix solution. However, questions as to why the two solvers did not produce identical solutions remain. Thus, future research will have to identify the reason for this difference in solutions between set-ups (ROMI-RIP 2.0 vs. ROMI 3.0 and SAS 8.2 vs. R 2.7.2). With solutions between the two least-cost lumber grade-mix solvers differing less than 1 percent, however, Buehlmann and others' (2004b, 2010) and Zuo's (2003) statistical solution to the least-cost lumber grade-mix problem has shown its ability to identify

Table 4.—Details of the least-cost lumber grade-mix solutions and yields for ROMI-RIP 2.0, and SAS 8.2 and ROMI 3.0 and R 2.7.2.

ROMI-RIP 2.0 and SAS 8.2

FAS	SEL	Lumber Grade			Average Yield (%)	Average Cost (\$ per MBF)
		1C	2AC	3AC		
		100			57.11	2,101

ROMI 3.0 and R 2.7.2

FAS	SEL	Lumber Grade			Average Yield (%)	Average Cost (\$ per MBF)
		1C	2AC	3AC		
		20		80	38.18	2,096

least-cost solutions from diverging initial yield results. These initial results indicate that the Forest Service can incorporate the free, open-source R 2.7.2 statistical software analysis package into a new version of ROMI 3.0. Industry participants will no longer need to use the internet to access SAS 8.2 on a server for their least-cost lumber grade-mix search.

CONCLUSIONS

Hardwood lumber is the largest expense incurred by U.S. manufacturers of secondary solid hardwood products. Thus, efforts are made to process this material in the most efficient manner, sometimes requiring significant investments. Minimizing cost by buying the least-cost lumber grade-mix does not require any investment, as software for solving this problem is available for free.

The solutions derived by the two least-cost lumber grade mix solvers using the same underlying statistical models but relying on different versions of the USDA Forest Service's rough mill simulation software (ROMI-RIP 2.0 and ROMI 3.0) and two different statistical analysis software packages (SAS 8.2 vs. R 2.7.2) are similar with respect to cost, yet different with respect to the suggested least-cost lumber grade-mix to be used. The minimum cost solution found by both solvers differed by less than 1 percent or \$5 per MBF. The least-cost lumber grade-mix solution from the ROMI-RIP 2.0- and SAS 8.2-based solver relied on 100 percent 1C lumber for obtaining all its parts requirements. The ROMI 3.0- and R 2.7.2-based least-cost lumber grade-mix solver solution asked for using 80 percent 3AC and 20 percent 1C lumber.

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

A GRAPHICAL AUTOMATED DETECTION SYSTEM TO LOCATE HARDWOOD LOG SURFACE DEFECTS USING HIGH-RESOLUTION THREE-DIMENSIONAL LASER SCAN DATA

Liya Thomas and R. Edward Thomas¹

Abstract.—We have developed an automated defect detection system and a state-of-the-art Graphic User Interface (GUI) for hardwood logs. The algorithm identifies defects at least 0.5 inch high and at least 3 inches in diameter on barked hardwood log and stem surfaces. To summarize defect features and to build a knowledge base, hundreds of defects were measured, photographed, and categorized. Our cost-effective, reliable, and robust statistical method locates surface defects using a commercially available high-resolution laser scanner. The scanned data capture three-dimensional (3-D) images that illustrate log shapes—protrusions and depressions—and portray bark patterns. Most available optimization systems were designed to determine gross external characteristics (e.g., shape, diameter, sweep) of softwood logs. They then better position the log with respect to the saw and improve the sawyer's decision-making ability. Adding external defect information to the optimization process is a natural extension of current technology. Our method applies decision rules obtained from the knowledge base and identifies severe defects. Using the defect detection GUI, we tested our system. Using 14 randomly selected logs, we detected 53 severe defects out of 64. Nine nondefective regions were falsely identified. When defects were calculated as areas, 90.2 percent of defects were located.

INTRODUCTION

For decades hardwood log processing optimization methodologies and techniques have been one of the top priorities in the wood products industry (Chang 1992). After harvesting, a tree undergoes a rough manual estimation of potential yields and is bucked into several logs. Before a log is sawn into boards, the operator quickly examines it and picks the best opening or highest-quality face. During the inspection, surface defects are identified and types, severity, and topological distribution of defects determined. Thus, lumber yield is highly subject to time constraints and the operator's skills and conditions at the time. Defects, such as sawn-knot stubs, rotten knots, surface distortion, holes, cracks, branch sprouts, swelling, worm holes, and animal damage, are imperfections that decrease wood value.

There are two types of wood: softwood and hardwood. Most softwoods have a fast growth rate and identical, clustered defects mostly caused by branch pruning. Hardwood trees generally grow more slowly and the distribution of surface defects is random. Hardwood defects vary in size, shape, and type, making them difficult to identify. Internal defects exist inside the log, while external defects are visible on the tree or log surface.

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With the advance of scanning equipment, it is possible to develop automated systems that aid bucking and sawing processes. Researchers have experimented with technologies and methods that locate and classify internal as well as external defects on either softwood or hardwood. Systems including those using X-ray/Computerized Tomography (CT), X-ray tomosynthesis, magnetic resonance imaging (MRI), microwave scanning, ultrasound, and ground-penetrating radar have been researched and developed to detect internal defects (Wagner and others 1989, Chang 1992, Guddanti and Chang 1998, Devaru and others 2005). Advanced computer vision and image-processing methods and algorithms were proposed for internal defect inspection on hardwood (Li and others 1996; Zhu and others 1996; Tian and Murphy 1997; Bhandarkar and others 1999). Some of these methods and technologies provide high detection rates as their data have extremely high resolution. However, issues such as high cost, low processing speed, data instability, and environmental safety keep them from being commercially available. Another approach is the use of laser scan technologies to collect profile data of softwood and hardwood logs. High-resolution 3-D laser-scanned data of hardwood logs show minute surface variations, making it feasible to detect external defects.

Compared to other systems such as X-ray/CT, laser-scanning equipment is more economical, safer, faster, and more suitable for log sizes (Thomas and others 2007). X-ray based systems might be appropriate for high-end veneer mills, if practical problems such as log size limitations can be resolved. External defect indicators are highly correlated with internal defect characteristics in hardwoods (Thomas 2008). This high correlation strongly suggests that automatically locating external defects based on laser-scanned data can help stem bucking and lumber sawing. In 2001, as the first group attempting external defect detection using laser-scanned log data, we began the development of a software system to identify and locate severe surface defects. Two batches of data were collected from two commercially important northeastern-American hardwood species, yellow-poplar (*Liriodendron tulipifera*), and northern red oak (*Quercus rubra*) (Fig. 1). To build a database, we measured, photographed, and categorized more than 200 external defect samples. Information was extracted from the database to develop the expert system in our defect detection algorithm.

A new robust statistical estimator was created to fit 2-D circles to data cross sections in the presence of severe outliers (Thomas and Mili 2007). Radial distances

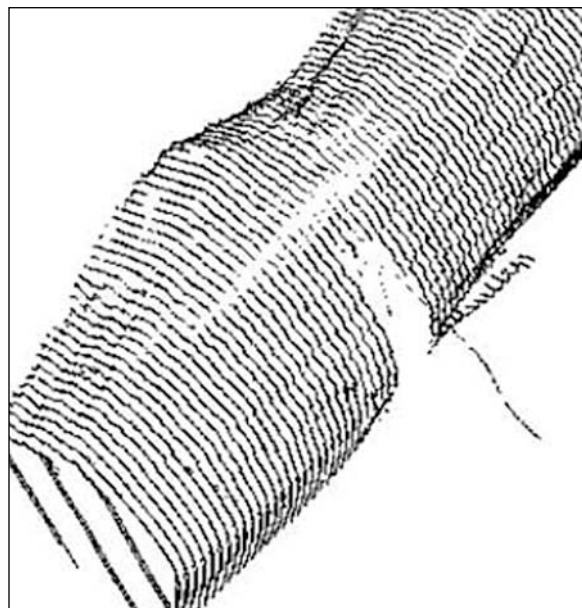


Figure 1.—Portion of the 3-D projection of laser-scanned range data for a red oak log.

are computed between the virtual log made of fitted 2-D circles and log data, based on which computer vision techniques were applied to generate a gray-scale height image of the log surface. Contour curves also are generated that display ups and downs. The defect detection algorithm analyzes both contours and 3-D data to locate severe defects at least 0.5 inch high and 3 inches wide. Defective regions are selected and framed in rectangles over the contour plot. The algorithm was implemented in Java and Matlab on a Windows XP operating system (Thomas and others 2006) and has been merged in Java. A state-of-the-art Java GUI was developed that allows users to control the log scanner, to scan log data, to execute our detection system, and to display detection results. Using the defect detection GUI, we tested the system.

STUDY AREAS

Our log-surface defect detection algorithm processes gray-scale imagery and 3-D range data by using a new robust generalized M-estimator, or GME (Thomas and Mili 2007). It extracts radial distances between the virtual log formed by a series of fitted circles and log data points. The radial distances provide a height map of the log surface with respect to the virtual log. This mapping enables our system to locate defects associated with unique characteristics and height change. Wagner and others (1989) experimented with a CT scanner operating at a speed acceptable to mills that collected internal log images for automatic defect detection. Using X-ray/CT images, Guddanti and Chang (1998) constructed virtual boards to resemble real sawn boards that demonstrated the feasibility of applying the technique. Zhu and others (1996) developed an inspection algorithm to identify internal log defects in CT images. It comprises an adaptive filter, a multi-threshold 2-D segmentation scheme, and a 3-D 8-neighbor connectivity analysis mechanism.

Using CT technology, Li and others (1996) investigated internal log inspection and developed a feed-forward multilayer Artificial Neural Network (ANN) system, trained by back propagation. It can classify three common defect types—knots, splits, and decay—and can accommodate several hardwood species, including northern red oak, water oak (*Q. nigra* L.), yellow-poplar, and black cherry (*Prunus serotina*). Local 3-D data are used to extract defect features, and a pixel-by-pixel classification accuracy of 95 percent was achieved. Sarigul and Abbott (2000) further refined these classifiers in a subsequent post-processing step, and developed the Intellipost system using domain-dependent rules. It improved segmentation accuracy of Li and others (1996) ANN by 1.92 percent, 9.45 percent, 4.22 percent, 0.33 percent for a red oak dataset, datasets provided by Forintech Canada, Inc., and two medical ones, respectively. Bhandarkar and others (1999) developed the CATALOG system for internal defect detection and classification via CT image analysis, consisting of segmentation and correlation phases. Hardwood species studied were red oak, black walnut (*Juglens nigra*), white ash (*Fraxinus americana* L.), and hard maple (*Acer saccharum*). Recent research activities in internal hardwood defect detection by X-ray/CT imaging include a group in Purdue University (Callahan 2007). Tian and Murphy (1997) proposed an automated digital camera-based texture analysis system, KnotVision, to locate and identify defects on freshly harvested pine logs (softwood).

Strong relationships among surface indicators and internal defect features have been discovered for severe defects, such as overgrown knots, sound knots, knot clusters, and unsound knots (Thomas 2008). For example, the strength of the correlations (adjusted multiple R^2) between interior halfway-point width measurement and exterior features ranged from 0.48 to 0.75. Similar results were found between external features and the halfway-point length measurement (adjusted multiple R^2 from 0.45 to 0.75). All correlations between external indicators and internal features were significant at the 99-percent level. Further, the low mean absolute errors (0.25 to 0.70 inch) indicate that internal features can be reliably predicted using external defect measurements. The defect detection algorithm discussed here identifies the size and location of surface defects. When this information is passed to the internal defect prediction model (Thomas 2008), a complete set of defect information for a log, internal and external, is generated. This is the data necessary to optimize the processing of the log to yield maximum product value. A recent test of the internal prediction models has shown that they can accurately predict the occurrence of 80 percent of the knots on sawn board faces.

METHODS

The fact that external defects corresponding to rises or depressions on log surface can be observed from 3-D range images suggests that defect locations can be determined by extracting surface height from the images. Three major steps are involved.

- 1) Fit 2-D circles to log cross-sections. Because datasets might have missing data or include irrelevant, deviant data points, a new, robust estimator was developed to estimate in a reliable manner the centers and radii of the fitted circles.
- 2) Extract radial distances between circles and log data to indicate local height changes. Radial distances refer to height values on the surface with respect to the reference.
- 3) Locate defects characterized by significant surface rises or depressions.

We experimented with methods that fit circles and ellipses to log data to create a reference surface, or virtual log, for converting 3-D data to 2-D images for processing. Our GM estimator that fit circles to the cross-section data showed little to no influence from the outlier data (approximately 3 percent of the dataset) or the missing data (Fig. 2). Our process converts 3-D range data to radial distances and uses image-processing methods commonly applied to gray-scale values (Fig. 3a). In this process, the log surface is unrolled onto a 2-D coordinate space to create a “skin” representing the pattern of log bark along with bumps and bulges associated with most defects. Radial distances are calculated between circle and log surface points, typically ranging from -0.5 to 0.5 inch, scaled to range from 0 to 255 gray-levels, and mapped to create a 2-D image. Log data are processed and interpolated linearly to fill any gaps between data points to form a grid. In our original data set, the cross-sectional density is nearly 20 times the longitudinal density, and linear interpolation generates an image proportional to the real log surface.

Six contour levels are computed from the orthogonal distances between a virtual log and any point of the cross section, generating a topographical exterior log map. Examining only the highest-level and lowest-level contours, we can detect defects corresponding to surface rise or depression. Log data also are referenced to find

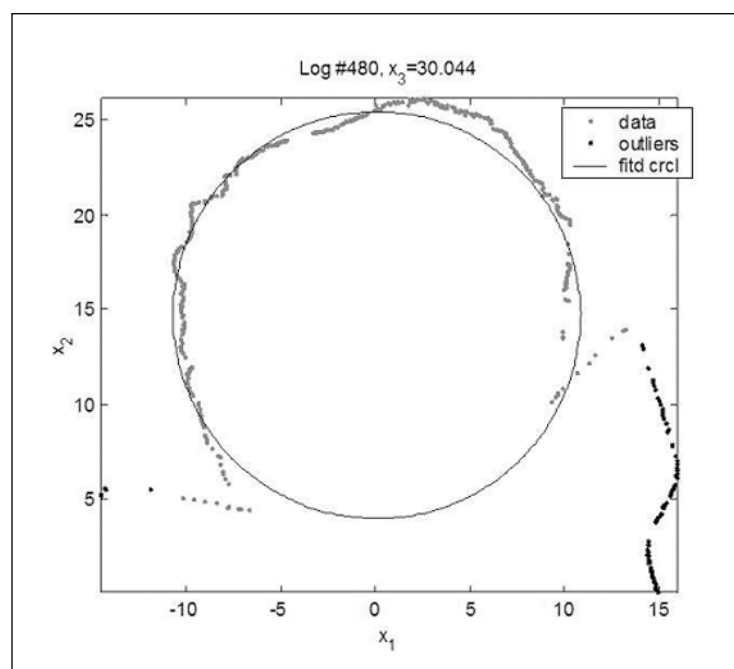


Figure 2.—Circle fitting to a cross section with severe outliers as a portion of the log support.

defects. Expert knowledge, in the form of defect shape and size characteristics, is applied in a stepwise fashion to rule out regions as potential defects, including regions with sizes smaller than a given threshold, nested in other curves, or long and narrow (determined by the actual width-to-length ratio of real log dimensions). The algorithm was created under the assumption that data resolution is 0.8 inches per cross section, so only those with a relatively significant height change on the surface (at least 0.5 inch), and/or a relatively significant size (at least 3 inches in diameter) are examined. Using radial distances visualized by the gray-level image in Figure 3a, the algorithm generates a contour plot as depicted in Figure 3b,

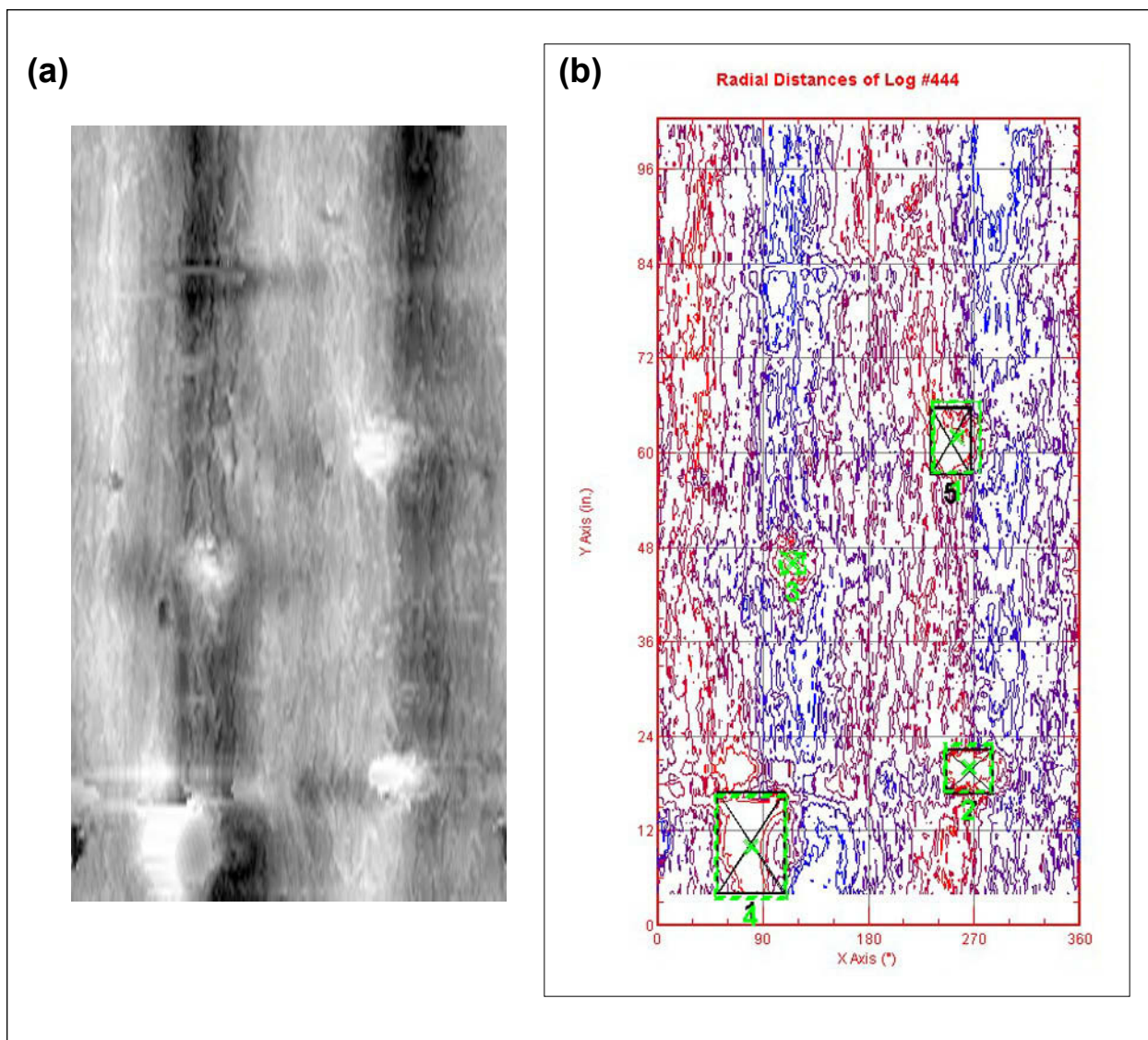


Figure 3.—(a) Radial distances generated by the log-unrolling process presented as a gray-level range image. Lower-left corner has a sawn knot, where sawn pattern can be observed. (b) Three out of four defects are automatically detected as displayed in the contour plot, enclosed in black crossed rectangles. The only exception is in the top left corner. Also shown is a manually recorded map of defects on the same log, enclosed in light green rectangles centered by a small cross.

and determines rectangle-enclosed regions bounding contour curves at the highest level. Some regions are selected if they are big enough or with a significant height.

The detection method uses a statistical procedure to examine the region surrounding a selected small region for relatively straight-line segments (Hampel et al. 1986). If the coverage of straight-line segments is sufficient, the defect region is adjusted to cover the entire defect surface, rather than just a corner. Such a region is identified as a flattop, which is likely a sawn top, either sound (not rotten), or unsound (rotten). Region-removal rules are given as follows: regions smaller than a given threshold; regions enclosed in curves nested in other curves, as there will be at most one defect in the same location; long and narrow regions (because they are normal bark regions); regions that are smaller than 50 in² and are too close to the selected large ones. Some

regions are removed for further consideration if they contain a large portion of missing data. The algorithm extends boundaries of an identified region if only a relatively high standing corner of the defect is located. Regions may include elevated yet nondefective log surface. Typically they are covered with tree bark and are thus associated with distinctive bark patterns.

RESULTS

Two groups of data were collected for our experiment. The first, scanned using a Perceptron scanning system (Perceptron Inc., Plymouth, MI), consists of more than 160 log data samples from both species, each with cross sections along the log length at 0.8-inch intervals and a resolution of about 0.04 inch within a cross section, shown in Figure 1. Each log data sample in this group consists of approximately 120,000 to 240,000 points (3-D coordinates). The second group was scanned using a JoeScan system (JoeScan, Vancouver, WA). It contains more than 140 red oak data samples, with a resolution of 0.06 inch between cross sections, and 0.09 inch within a section. The number of cross sections per log ranges from 1,600 to 3,200, and each cross section has 400 to 800 points. Thus, the second data group has much higher resolution and number of data points varies from 0.64 to 2.56 million per log sample. The prototype of our defect detection system was created and tested using the first data. The circle-fitting part in the algorithm was implemented using Java. The rest of the algorithm that detects defects based on radial distances was written in MATLAB 7.0 (MathWorks, Natick, MA; MATLAB 2008) for its readily available and easy-to-use graphical capabilities.

To make our system easier to use, a sophisticated graphical interface was developed that allows users to enter various information by selecting items through pull-down menus, then to run it by pressing a button (Fig. 4). Through the interface, a user may control the scanning process of the log scanner and launch the log

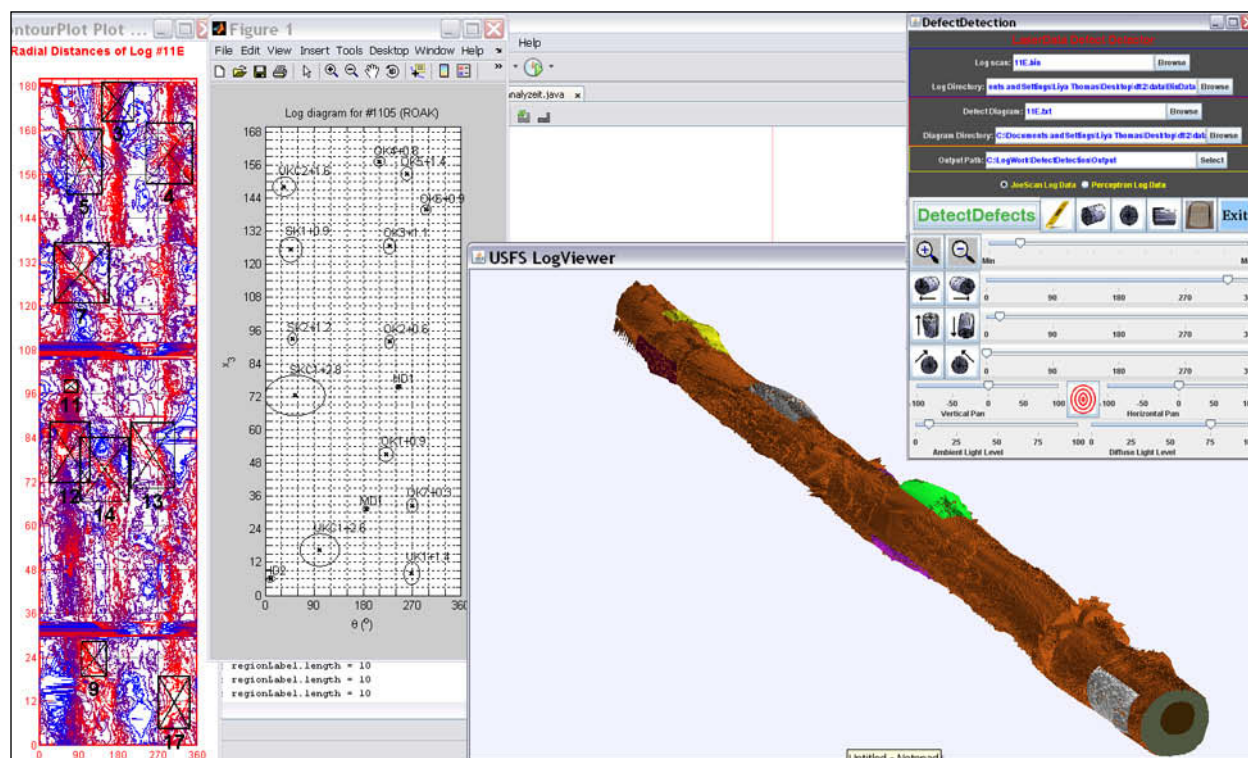


Figure 4.—Partial screen shot of GUI (Graphical User Interface) for the defect detection system. On the left side is the contour plot with automatically detected regions enclosed in black crossed rectangles. To the upper-right is the control panel.

data scan. Once the data are collected, the user may press the corresponding button to execute our detection system. Detection results are automatically displayed in three different forms: a 3-D projection of the log data with defective regions marked; a contour plot tiff image of the log surface height change, also with defective region marked; and a text file of residual values. The 3-D log data projection can be viewed interactively. The user can zoom in/out, rotate, or pan it. Lighting also can be adjusted. These capabilities make our system both interactive and real-time.

We used an open-source Java package to generate 2-D contour plots of data (Huwaldt 2002). Figure 3b presents a contour plot with automated defect detection results overlaid by observed severe defective regions. Changes were made to the Huwaldt package to accommodate our detection system and fix several incompatibilities. Huwaldt's package was developed for relatively small datasets, while the log data can have as many as 2.5 million 3-D points. Thus, we reduced the data to 50 percent for the first group and 17 percent for the second one; that is, 90,000 and 272,000 points on average, respectively. Resolutions for the two groups are 0.8×0.08 in²/pixel, and 0.2×0.2 in²/pixel, respectively. To get a uniform resolution along both the log length and cross section in the first batch, linear interpolation was performed in the direction of log length. Data reduction ensures that the current version of our system will complete the analysis within a reasonable amount of time. This processing speed is possible because a data mesh with these densities is fine enough for a contour-curve based detection algorithm such as ours that aims only at the most protruding regions.

Tables 1 and 2 show simulation results from 14 log samples from the Perceptron data using our new Java implementation. Out of 64 severe defects, we detected 53. We falsely identified nine nondefective regions. When defective regions are calculated as areas, 90.2 percent of the defects were located and 1.8 percent of the total clear area falsely identified as defective. The algorithm made the assumption that data resolution is no more than 0.8 inch per cross section (along log length), which confines the defect size that can be detected to be no more than 3 inches wide, and at least 0.5 inch high. Otherwise there would be fewer than four cross sections spanning a defect, and low-lying defects are not enclosed in the highest-level contours, usually at

Table 1.—Defect detection simulation results in terms of number of rectangle regions for 14 log samples.

Log #	Total	Detected	Missed	False
444	4	3	1	-
448	8	7	1	1
450	4	3	1	-
453	7	6	1	-
468	3	3	-	-
480	5	5	-	1
493	5	3	2	-
501	3	3	-	-
508	5	4	1	1
521	6	5	1	4
537	4	3	1	-
441	2	2	-	-
485	6	4	2	1
520	2	2	-	1
Sum	64	53	11	9

Table 2.—Defect detection simulation results in terms of area (in²) for 14 log samples.

Log #	Surface	Observed	Detected	Missed	False
444	5,797	215	187	8	-
448	7,284	805	812	24	217
450	7,278	235	326	15	-
453	6,301	1,168	928	36	-
468	5,453	812	949	-	-
480	7,486	798	750		166
493	8,551	255	299	42	-
501	3,916	466	387	-	-
508	4,031	459	532	30	295
521	8,560	720	596	24	414
537	6,414	389	263	23	-
441	4,645	202	207	-	-
485	9,352	1,126	715	288	272
520	6,188	368	282	-	159
Total	91,256	8,018	7,233	490	1,523
%	-	-	90.2	6.1	1.8

about 1 inch. The resolution of the second batch is five times higher than that of the first one— 0.06×0.1 in²/pixel vs. 0.8×0.04 in²/pixel. Thus, it is possible to identify defects at the higher resolution that were not previously identified.

We are testing our detection system with both groups of data. During testing with JoeScan data, several problems appeared. First, the algorithm is an expert system that uses many parameters, for example, region width and height, as a threshold to rule out selected regions that are not defective. At places, these parameters are examined using number of points within a cross section, which depends on resolution. Second, due to a data collection issue, noise appears frequently with the higher-resolution data. Third, data resolution along cross sections in the second batch appears to vary. These shortcomings interfere with detection results and more tuning is needed to improve detection capability and reduce false rates. Execution time per log sample is less than 1 minute with data from the Perceptron scanner, and about 1.2 minutes with data from the JoeScan scanner on a desktop computer with an Intel 2.6 GHz Core2 processor.

DISCUSSION

Other than the severe defects that our system is able to identify, there are other groups of severe defects such as heavy distortion (HD) and mild distortion (MD). They are not associated with significant size (mostly about 2-3 inches in diameter), or height change (less than 1 inch high). In a 2007 study of red oak, we counted defect numbers for all defects, all logs, and all trees (Thomas et al. 2007). We identified 1,376 surface defects, among which 60 are HDs, and 103, MDs. They represent 4.4 percent and 7.5 percent of all defects, respectively. Although they lack the topological characteristics of other severe defects, HDs and MDs typically present unique bark patterns. We are looking into technologies such as computer pattern recognition to locate them. From the gray-scaled surface-rise images of 64 third and higher red oak logs with which we experimented, only 12 HDs and 1 MD were at all visible to the human eye. Locating HDs and MDs by computer software is likely difficult, as software will be developed by researchers, in a sequence of programming commands. Obviously it is going to be much more limited than the processing power of

the human brain. Further, the only thing our detection system has to work with is surface height images. Currently it cannot get help from digital color images or from ground truth. With limited input information and limited capability, it remains unknown how effective the detection module can be for defects that include HDs and MDs.

Much can be improved in our defect detection algorithm. Problems arose due to different data resolution. Parameters should be redefined to accommodate different data resolutions—although given the current structure, such modification to the system likely requires a complete overhaul. Currently the contour plot package is built upon JFrame, a Java primitive class. The quality of the resulting plots is not desirable. Adjusting, editing, and storing the graphs are inconvenient, making analysis of results difficult. We shall experiment with other graphical representation methods to improve graphical quality. Further, pattern recognition and neural network methods will be investigated for detecting severe defects without significant height changes, but with unique bark patterns. This application can be observed in Figure 3a where, in the lower-left corner, the sawn pattern is displayed in the radial distance image.

We have tested our system with more data. Our new GUI is easier to navigate and more user-friendly to sawmill operators. The detection software is integrated with the laser scanning system. Lab experiments were conducted to integrate and test the system using sample logs. We are collaborating with a sawmill located in West Virginia to perform real-time industrial testing. We hope our defect detection system will eventually provide a tool that is both effective and affordable for the forest products industry.

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

FOREST ECOLOGY

FOREST COMPOSITION CHANGE IN THE EASTERN UNITED STATES

Songlin Fei and Peilin Yang¹

Abstract.—Forest ecosystems in the eastern United States are believed to be experiencing a species composition change, but most evidence is anecdotal or localized. We used U.S. Forest Service Forest Inventory and Analysis data to quantify the annual changes of three common genera: *Acer* (maple), *Carya* (hickory), and *Quercus* (oak) with survey periods between 8 and 20 years across the eastern United States. The majority of the region had an increase in maple abundance. At the same time, hickories and oaks had a near universal decrease in the Central Hardwood region.

INTRODUCTION

Forest ecosystems in the eastern United States are believed to be experiencing a species composition change, where shade-tolerant species are becoming more abundant largely in response to forest maturity and change of disturbance regime (Nowacki and Abrams 2008). Many studies have pointed to significant declines in disturbance-dependent species such as oaks (Abrams and Nowacki 1992, Johnson 1976), with significant increase in shade-tolerant species such as red maple (Fei and Steiner 2007). However, most evidence is anecdotal or localized, and often lacking of quantitative information. To quantify the change at regional scales, we used U.S. Forest Service Forest Inventory and Analysis (FIA) data (USDA FS 2008) to describe the average changes in relative abundance of three common genera: *Acer* (maple), *Carya* (hickory), and *Quercus* (oak) during the period of 1980-2008 across the eastern United States.

METHODS

Data from FIA program were used in this study (USDA FS 2008). All states were inventoried periodically, but at irregular and asynchronous intervals before 2000. Most states have been inventoried annually, but partially, since 2000. For this study, we used FIA Mapmaker 3.0 (Miles 2008) to capture county-level information on the total number and volume of all live trees for all species in the eastern 37 states of the United States, defined here as North Dakota south to Texas and all states to the eastward. For most states we obtained data from two or more completed inventories, beginning with the first available measurement after 1980. The first inventory for each state (T_1) was a periodic survey conducted between 1980 and 1995 depending on the state. The second inventory (T_2) was defined as the latest available periodic survey or full-cycle, annual survey (all plots) as of June 2008. The interval between the two inventories ranged from eight to 20 years with an average of 16.4 years. Because a full-cycle, annual survey takes 5 to 7 years to complete, the median year was used to calculate the interval. In total, we used data from all 2,625 counties in 37 eastern states in the United States.

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Importance value (IV) was used to describe relative abundance of each species group for each inventory. In this study, we define IV as the mean of 1) relative density (*total number of oaks / total number of all live trees* x 100) of trees with a diameter of at least 1.0 inch; and 2) relative volume or dominance (*total volume of a species group / total volume of all live trees* x 100). To understand the general trend of current abundance for each genus, a trend surface was developed. Species group abundance from the most recent inventory within each county was first assigned to the county centroid, and extrapolated to the entire region using the Ordinary Kriging (12 point search radius) method in ArcGIS (ESRI Inc., Redlands, CA). To eliminate the distraction from nonforested areas, only forested areas (based on the 2001 National Land Cover Dataset, USGS 2008) were extracted. Changes in IV were further compared between T_1 and T_2 for counties that had data for both inventories. Because the intervals between T_1 and T_2 are different for each state, annual change rate was calculated by using the overall difference between T_1 and T_2 divided by interval length to eliminate the difference caused by different sampling intervals. Forest area constrained trend surfaces were also developed for changes of the above measures.

These developed trend surfaces were then superimposed by map of the ecoregions in four major regions in the eastern United States: Northern Hardwood Region, Central Hardwood Region, Southern Pine-Hardwood Region, and Forest-Prairie Transition Region (Bailey 1997) (Figs. 1-3). Percentages of areas within each ecoregion and across the eastern United States that experienced an increase or decrease in species group abundance were then tabulated in ArcGIS.

RESULTS

MAPLE

Spatial Distribution of Maple Abundance

Among the four major regions in the eastern United States, the Northern Hardwood Region had the highest maple abundance, followed by the Central Hardwood Region (Fig. 1a). The majority of the Northern Hardwood Region contained maple, and over 97 percent of the areas had maple IV > 5 percent. New York, northern Pennsylvania, and upper Michigan had the highest maple abundance (> 25 percent in IV). About 90 percent of the Central Hardwood Region had maple IV > 5 percent. However, maple is more concentrated in the eastern portion of the Central Hardwood Region, especially in the Allegheny Mountains in western Pennsylvania and West Virginia and the Cumberland Plateau regions. Maples in the remaining two regions were also widely distributed. The overall trend is that maples, as a group, are more abundant in the Appalachian Mountains and Southern Superior Uplands.

Changes in Maple Abundance through Time

More than 69 percent of the eastern forest has experienced some increase in maple abundance (Fig. 1b, Table 1). The Central Hardwood Region had the highest percentage of forested areas that experienced maple IV increase (>81 percent), followed by the Southern Pine-Hardwood Region (69 percent), and the Northern Hardwood Region (62 percent). In the Forest-Prairie Transition Region, more areas experienced a decrease (54 percent) in maple IV than areas increased. The majority of the Central Hardwood Region had an average annual maple IV increase > 0.1 percent, while areas in the southern unglaciated Allegheny Plateau in southeastern Ohio and western West Virginia had an average annual maple IV increase > 0.3 percent.

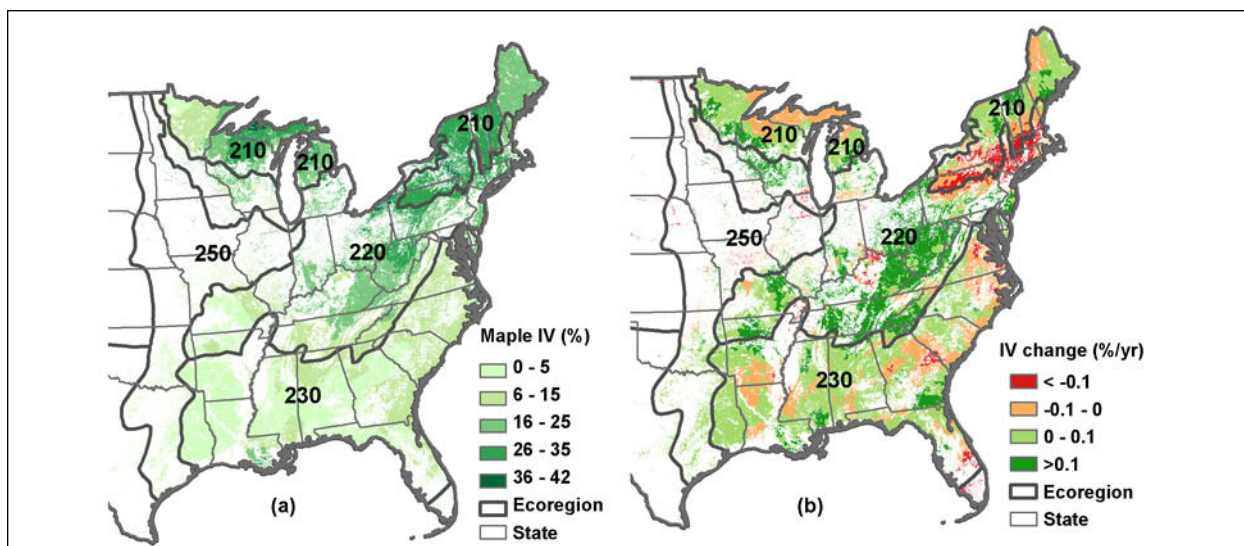


Figure 1.—(a) Current abundance, as measured by importance value ($[(\text{relative density} + \text{relative volume}) / 2]$), for all maple species in the eastern U.S. forested areas; and (b) mean annual change of importance value for all *Acer* species during the last two decades. Numbers correspond to Ecoregion Divisions (Bailey 1997): 210 – Northern Hardwood Region, 220 – Central Hardwood Region, 230 – Southern Pine-Hardwood Region, and 250 – Forest-Prairie Transition Region.

Table 1.—Percentage and acreage of forested area and decrease of the abundance for maple, hickory, and oak in percent of forested areas by ecoregion.

Ecoregion ^a	Total area (Million acres)	Forested (%)	Forested area (Million acres)	Areas decreased in abundance (%)		
				Maple	Hickory	Oak
210	122	78.5	96	38.1	43.6	27.9
220	287	42.8	123	18.7	68.8	81.0
230	270	58.9	159	31.2	80.5	53.0
250	191	8.5	16	53.6	59.1	76.5

^a Ecoregion: 210 – Northern Hardwood Region, 220 – Central Hardwood Region, 230 – Southern Pine-Hardwood Region, and 250 – Forest-Prairie Transition Region

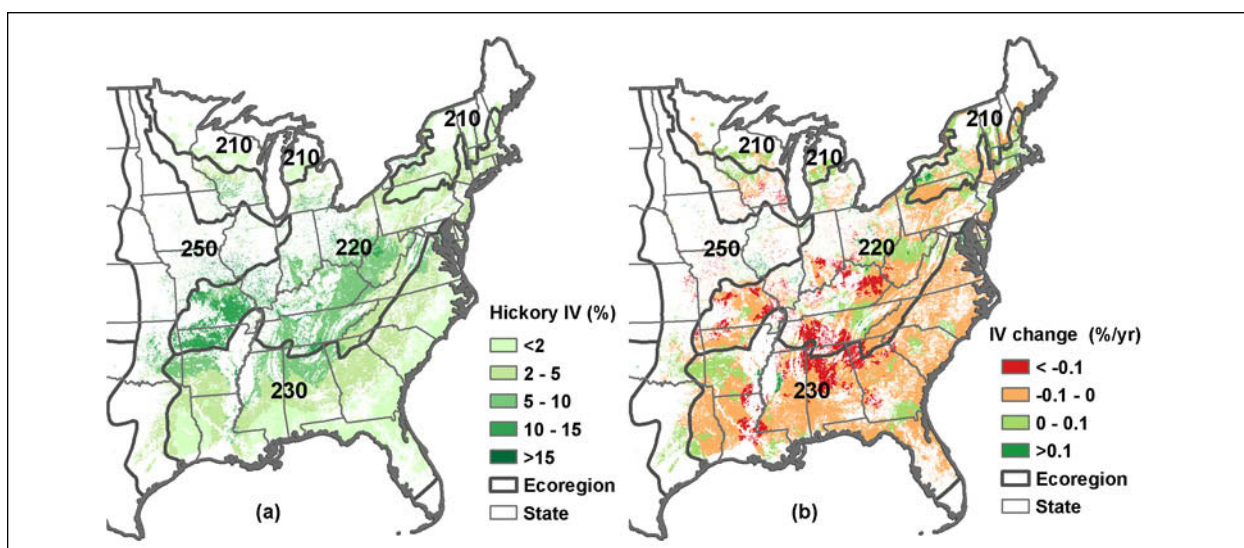


Figure 2.—(a) Current abundance for all hickory species in the eastern U.S. forested areas; and (b) mean annual change of importance value for all *Carya* species during the last two decades.

HICKORY

Spatial Distribution of Hickory Abundance

Hickory species are mainly distributed in the Central Hardwood Region and the Southern Pine-Hardwood Region (Fig. 2a). They also have a limited distribution in the southern portion of the Northern Hardwood Region and the Forest-Prairie Transition Region. The Central Hardwood Region had about 60 percent of its forested areas with > 5 percent of IV for hickory species. The highest hickory abundance was observed in the Ozarks in the Central Hardwood Region, followed by southern portion of the southern unglaciated Allegheny Plateau and the Northern Cumberland Plateau regions. Hickory in the Southern Pine-Hardwood Region is more concentrated in the northern boarder adjacent to the Central Hardwood Region.

Changes in Hickory Abundance through Time

More than 72 percent of the eastern forest has experienced some decrease in hickory abundance (Fig. 2b, Table 1). The Southern Pine-Hardwood Region had the highest percentage of forested areas that have experienced IV decrease (>80 percent), followed by the Central Hardwood Region (69 percent). A sharp hickory abundance decline was observed in the area of central Tennessee, northern Alabama, and northeastern Georgia. The Ozarks in Missouri and the Cumberland Plateaus in eastern Kentucky also experienced a prominent decline in hickory abundance. On the other hand, hickories increased in abundance in the northern portion of their distribution in Wisconsin, Michigan, New York, and other states in northern New England.

OAK

Spatial Distribution of Oak Abundance

Among the four regions in the eastern United States, the Central Hardwood Region had the highest oak abundance, followed by the Southern Pine-Hardwood Region, the Northern Hardwood Region, and the Forest-Prairie Transition Region (Fig. 3a). Within the Central Hardwood Region, the Ozarks had the highest oak abundance (> 40 percent in IV), and portions of the Appalachians from Pennsylvania southward and the adjacent Cumberland Plateau have significant concentrations of oak (IV > 20 percent). Within the Southern Pine-Hardwood Region, oak was most dominant in northern Georgia and Alabama and, eastern Texas and adjacent parts of Louisiana and Arkansas. Oak was rather sparsely but widely distributed across most of the Northern Hardwood Region, and the greatest concentrations occurred along the southern edge of the region where it borders the Central Hardwood Region. For the Forest-Prairie Transition Region, only limited area had notable oak distribution.

Changes in Oak Abundance through Time

The majority of the eastern forest has experienced a decrease in oak abundance (Fig. 3b, Table 1). In the Central Hardwood Region, oak abundance has decreased in over 81 percent of the forested areas that had decreases, with the sharpest declines occurring in the Ozarks, the Highland Rim in Tennessee, the Allegheny Plateaus in southwestern Pennsylvania and eastern Ohio, and the Driftless section in Wisconsin and Minnesota. The only area with prominent increase in oak abundance in this region was eastern Tennessee and Northern Georgia. In contrast, oak abundance increased in over 72 percent of the Northern Hardwood Region. Increases and decreases in oak abundance in the Southern Pine-Hardwood Region were about evenly split by area. The majority of the area in the Coastal Plains in eastern Texas, Louisiana, Alabama, and Florida had considerable increases in oak abundance.

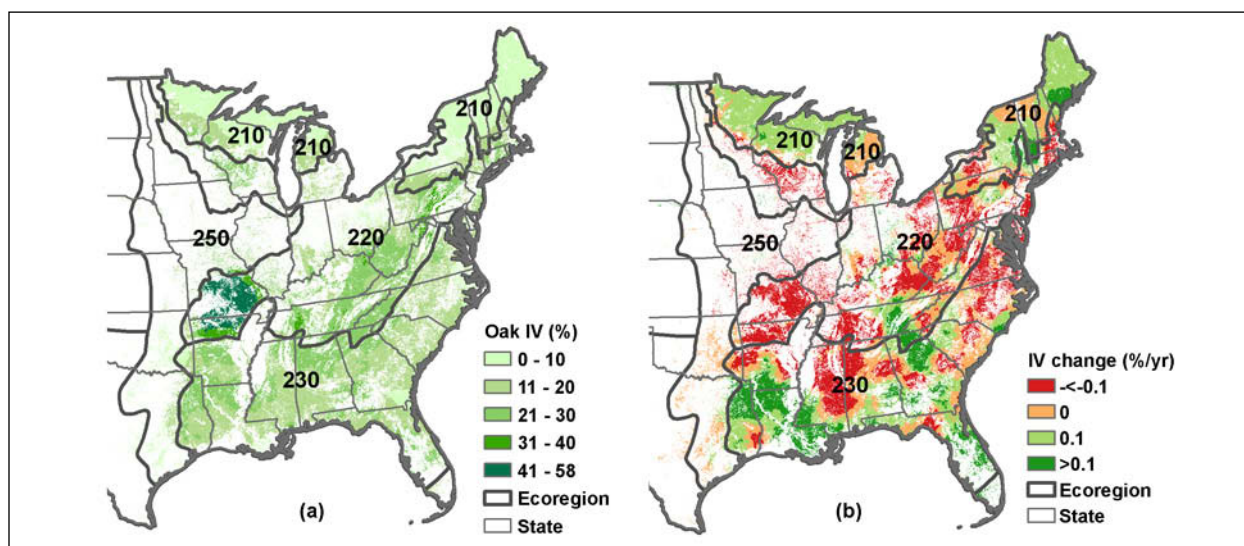


Figure 3.—(a) Current abundance for all oak species in the eastern U.S. forested areas; and (b) mean annual change of importance value for all *Quercus* species during the last two decades.

DISCUSSION

In general, maple species have registered an increase in most of the eastern forest, while oaks and hickories have registered a decrease in relative abundance, especially the Central Hardwood Region during the last two decades. However, the change is not universal. Maple species have sharply decreased in the southern portion of the New England area, while oaks have experienced an increase in majority of the Northern Hardwood Region. The reasons for these changes are unclear and probably complex, but several exogenous factors have been implicated because they are known to affect the growth and survival of these species groups, and they have changed markedly during the twentieth century. Fire suppression may be a major factor for the species composition change (Brose and others 1999, McCune and Cottam 1985). Modern forest harvest practices may be another major factor that favors shade-tolerant species (Fei and Steiner 2009). Invasions of exotic pests and pathogens such as gypsy moth (*Lymantria dispar*) and chestnut blight (*Cryphonectria parasitica*) may also result in altered species abundance (Fajvan and Woods 1996, Vandermast and others 2002). At the same time, the dramatic increase in white-tailed deer (*Odocoileus virginianus*) populations during the 20th century may also have contributed to declines in preferred food species including oak and hickory (Kittredge and Ashton 1995). Other factors such as climate change and modern land-use may also have contributed to the change in species composition. Overall, further study of species level abundance change is needed to better understand this problem.

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

COMPOSITION, STRUCTURE, AND DYNAMICS OF THE ILLINOIS OZARK HILLS FOREST

Lisa M. Helmig and James S. Fralish¹

Abstract.—In the mature oak-hickory ecosystem of the Illinois Ozark Hills, forest community composition, dynamics, and structure were studied to examine the extent of conversion to mesophytic species and eventually predict the broad threshold time of complete conversion. Tree, sapling, and seedling data were collected from 87 plots distributed throughout the region. Data for the tree stratum and for the seedling and sapling strata were analyzed by Ecological Land Type (ELT): south slope, ridgetop, north slope, and stream terrace. To compare the vegetation among ELTs and between stand strata, importance values (relative basal area for trees; relative density for seedlings and saplings) were calculated and a composition index [$CI = \sum(\text{Species Importance Value} \times \text{Adaptation Value})$] developed for each stratum. On south slope and ridgetop ELTs, CI values near 600 were typical of an oak-hickory dominated overstory (Adaptation Values [AVs] = 4-6.5). Dominant species were white and black oak. On the more mesic north slope ELT, a CI of 774 indicated a mixed overstory dominated by northern red oak, white oak, yellow-poplar, and black oak. The Terrace ELT was dominated by yellow-poplar and a mixture of upland and bottomland species; overstory CI was 747. For all ELTs, sapling CI ranged between 852 and 958, while seedling CI varied between 772 and 921. These CI values indicated an understory moderately to strongly dominated by mesophytes, primarily sugar maple, American beech, and red maple (AVs = 9.1, 9.6, and 10, respectively). Given the universal and extensive development of the understory, the Illinois Ozark Hills seem destined to be the first major ecosystem to convert from a forest dominated by oak-hickory to one dominated by maple-beech.

INTRODUCTION

Succession is creating a major change in community composition of nearly all North American forests. Succession is defined here as the temporal replacement of a forest community composed of shade-intolerant or shade-intermediate disturbance-dependent (early successional) species by one in which more shade-tolerant mesophytic (late successional) species dominate to form a compositionally stable community (Fralish and others 1991). Within the ecological landscape, stands of early successional species typically have developed after a major period of disturbance, specifically after fire or fire and logging. If a lengthy period of protection or absence of disturbance follows this period, succession often occurs. The process is largely driven by differential shade tolerances among species, and if seed is available, generally proceeds unrestricted in the absence of disturbance and soil moisture deficits (Fralish 1988).

Parallel examples of forests composed of fire-tolerant early successional species are found throughout North America. In the Rocky Mountains, lodgepole pine (*Pinus contorta* Dougl. ex Loud.) stands are changing

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to Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) and Englemann spruce (*Picea engelmannii* Parry ex Engelm.). Large areas of quaking aspen (*Populus tremuloides* Michx.) and paper birch (*Betula papyrifera* Marsh.) are being replaced by white spruce (*Picea glauca* [(Moench)] Voss) and balsam fir (*Abies balsamea* [L.] Mill.) in Canada. In the Lake states, quaking aspen and jack pine (*Pinus banksiana* Lamb.) are being replaced by white pine (*Pinus strobus* L.) on sand soil and northern hardwoods on loam soil. In the southeastern United States, southern pine (for example, shortleaf pine [*Pinus echinata* Mill.] and loblolly pine [*Pinus taeda* L.]) stands have understories dominated by various species of scrub oak (*Quercus* spp.) and hickory (*Carya* spp.) (Baker and Langdon 1990, Lawson 1990).

On the deep, moist soil of the Central Hardwood Forest Region, the oak-hickory (*Quercus-Carya*) forest community is undergoing a parallel compositional change to mesophytic species, primarily sugar maple (*Acer saccharum* Marsh.) but also including American beech (*Fagus grandifolia* Ehrh.), red maple (*Acer rubrum* L.), and a variety of mesophytic shade-tolerant hardwoods, depending on the region. Invasion has been reported by many researchers working in the oak-hickory dominated communities of Pennsylvania, Indiana, Kentucky and Tennessee, Michigan, Wisconsin, Missouri, Iowa, and Kansas. Refer to Fralish and McArdle (2009) for citations. Most of these studies investigated only present forest condition based on a structure-for-time approach, that is, an oak-hickory overstory canopy and an understory dominated by shade-tolerant mesophytes that eventually will replace the overstory.

Recently, Fralish and McArdle (2009) examined Ozark forest change over a 300-year period characterized by three distinct disturbance regimes. During the last regime, which began about 1930 and continues to the present, the researchers reported a rapid rate of composition change. However, community structure was based on total understory density and was not fully detailed, as estimating the broad threshold time of conversion was not an objective of the research. This broad time or period is defined here as the time at which the biomass of a functional group whose seed is distributed by wind exceeds that of a functional group whose seed is distributed by gravity and animals. The research reported here closely examines community composition change and structure in more detail, and is the first phase in determining the threshold time of conversion from an ecosystem dominated by oak and hickory over the past 4,000-5,000 years (Delcourt and Delcourt 1987, Franklin 1994) to one dominated by sugar maple, American beech, red maple, and other mesophytic species in this century.

The specific objectives of the present study were to do the following for each Ecological Land Type (ELT):

- (1) determine average composition, density, and basal area for seedling, sapling, and tree strata, and
- (2) examine forest structure by interfacing data from the seedling and sapling strata with data from several tree size-class strata. The baseline data from objectives (1) and (2), along with growth and mortality data, will be used to estimate the time of conversion. For increased reliability, we doubled the number of plots used by Fralish and McArdle (2009) to more fully document the extent of the successional process in each ELT, and identify any ELT whose community composition might skew or bias the results.

STUDY AREA

The Illinois Ozark Hills is that portion (approximately 50,000 ha) of the extensive Ozark Plateaus physiographic region that extends into southwestern Illinois. Physiographic regions bordering the region include the sandstone Shawnee Hills on the east, the Mississippi River Alluvial Plain on the west and south, the Till Plain region on the north, and the Coastal Plain on the south/southeast. Schwegman (1973) identified

the area as the Southern Section of the Ozark Division. The Illinois Ozark Hills generally outline the western-most section of the Shawnee National Forest. See Fralish and McArdle (2009) for map of area.

Bedrock is Devonian-age cherty limestone uplifted during the Pennsylvanian period. Where chert was exposed on ridgetops and steep slopes, splitting resulted in a subsequent accumulation of fragments on lower slopes and in ravines. Steep slopes and narrow deep valleys characterize the mature dendritic drainage pattern.

The area has been indirectly impacted by Pleistocene glaciation. When the glaciers melted, glacial debris was carried downstream and sorted; a silty flour was deposited in the Mississippi River Alluvial Plain. During dry winter periods for the past 11,000 years, westerly winds lifted particles of this alluvium out of the flood plain and deposited it on the uplands. Near the Mississippi River, these loess deposits are up to 10 m deep on the bluffs and adjacent ridges but as thin as a meter on steep side slopes. Deposit thickness decreases approximately eastward from the Mississippi River across Illinois to near the Ohio River (Fralish and others 2002).

A variety of Alfisols and Entisols occur within the study area. Stookey (fine-silty, mixed, mesic Typic Hapludalf) and Hosmer (fine-silty, mixed, mesic Fragic Hapludalf) types were developed in deep, non-cherty loess deposits of the upper slopes and ridgetops. Hosmer soil has a fragipan and a bisequm profile. Goss soil (clayey-skeletal, mixed, mesic Typic Paleudalf) and Baxter soil (fine, mixed, mesic Typic Paleudalf) are found in thin layers of loess, clay, chert, and limestone on steep mid- to lower slopes. Goss soil contains more chert and has a higher base saturation than Baxter. Elsay (loamy-skeletal mixed, nonacid, mesic Typic Udifluvents) and Wakeland soils (coarse-silty mixed, nonacid, mesic Typic Udifluvents) occur along stream terraces. Elsay soil contains coarse chert fragments in the control section while Wakeland soil is more poorly drained (Natural Resources Conservation Service 2005, 2007).

The oak-hickory forests of the area generally were established in the mid- to late 1800s and early 1900s, under a regime of relatively heavy disturbance (Fralish and McArdle 2009). The major disturbance that initiated the period was the 1810-11 earthquakes, measuring between 8.4 and 8.7 on the Richter scale (Atkinson 1989). It was the most significant seismic event to occur in the central United States in historic times (Street 1982). These earthquakes severely damaged the forests of the region (Fralish and McArdle 2009) just as settlers began moving into the region. With settlement came extensive cutting of healthy and damaged forests; building of roads, towns, and larger cities; grazing of forests, and development of industries. Moreover, these activities and clearing of land for farms and orchards resulted in many large, frequent, and severe wildfires (van de Gevel and Ruffner 2006) that continued into the early decades of the 1900s (Miller 1920, 1923).

METHODS

FIELD PROCEDURES

Data from two sets of plots were used in the study. One set of 42 plots was located within the relatively undisturbed 144-ha Ozark Hills Nature Preserve within Trail of Tears State Forest. Plots were randomly located on sites that had a uniform soil on a ground surface of uniform aspect and steepness. The forest community on an acceptable site was required to have a homogeneous mixture of species, be at least 80 years in age, and lack evidence of disturbance such as fire, cutting, or grazing. These stands appeared to be typical for the Ozark Hills forest ecosystem.

Plots were distributed to obtain data from within four distinct topographic units (south slope, broad ridgetop, north slope, and stream terrace). Following the productivity levels and site index differentials for various aspects (azimuths) reported by Beers and others (1966) and Lloyd and Lemmon (1970), we separated “north” and “south” slopes along a diagonal from southeast (135°) to northwest (315°). The broad ridgetop ELT was differentiated on the basis of the thick loess cap (5-10 m) compared to the thinner deposit of side slopes (1-3 m). It also is differentiated from the “knife edge” ridgetops that are common within the Ozark Hills. The terrace is an area of deposition while the other topographic units are erosional in nature. Terrace environmental conditions and vegetation also interface with bottomland areas.

In keeping with the National Ecological Classification System (U. S. Department of Agriculture 1993, Keys and others 1995) the four topographic units were considered ELTs as more than one site type (i.e., Ecological Land Type Phases [ELTPs]) were found in several units (Fralish and McArdle 2009).

A conduit pipe (permanent marker) was used as the center of three nested circular plots. On a 0.04-ha plot (radius = 11.28 m), trees (stems 9.0 cm and larger) were recorded by genus, species, and diameter at breast height (d.b.h.) to the nearest 0.1 cm. Nomenclature follows Fralish and Franklin (2002). Saplings (stems 1.0 - 8.99 cm d.b.h.) on a 0.012-ha plot (radius = 6.18 m) and seedlings (stems equal <1.0 cm d.b.h. but taller than 10 cm) on the 0.003-ha plot (radius = 3.09 m) were recorded by genus, species, and size class. Aspect in degrees azimuth, slope position, and percent slope were recorded for all plots.

Data also were collected from a set of 46 plots located at randomly selected section and quarter-section corners distributed throughout the region. Prior to field sampling and for each ELT, a pool of 15 potential sampling points (corners) was randomly identified from U.S. Geologic Survey 7.5-minute quadrangle maps. Out of this pool, 10-12 points were sampled. As each corner was visited, stand and site conditions were evaluated for sampling. If the conditions met the criteria established for the first set of plots sampled in the Nature Preserve, data were immediately collected using the same sampling procedures but with one variation. To avoid the possible influence of past disturbance directly at the corner, the center point for the nested sapling and seedling plots was moved in a randomly selected cardinal direction to the midpoint (5.64 m) between the tree plot center and perimeter.

DATA SUMMARY AND ANALYSIS

Tree diameter was converted to tree basal area (BA) and BA summed by species for all plots within each ELT. These BA values were converted to BA/ha by multiplying by a conversion factor which varied with the number of plots: $\text{Basal Area} \times [10,000 \text{ m}^2 / (400 \text{ m}^2 \times \text{number of plots})]$. Tree densities were calculated in a similar manner. For each species, relative basal area (henceforth referred to as “Importance Value,” IV_{100}) was calculated by dividing species BA for an ELT by the BA for all species in the ELT and multiplying by 100. These procedures follow Fralish and others (1991; 1993) and Fralish and McArdle (2009). Basal area alone was used as an IV because tree biomass, live sapwood, and leaf area are directly related to basal area (Rogers and Hinckley 1979, Fralish 1994).

Seedling and sapling stem counts were summed by species for all plots within an ELT. Species density (number/ha) was determined for each of the four ELTs by multiplying total stem count by a conversion factor which varied with the number and size of plots: $\text{stem count} \times [10,000 \text{ m}^2 / (\text{plot size in m}^2 \times \text{number of plots})]$. Seedling and sapling relative densities (henceforth referred to as “Importance Value,” IV_{100}) were calculated by dividing the density for a species in an ELT by the density for all species in the ELT and multiplying by 100.

Compositional stability of stands in different ELTs was evaluated by comparing a composition index (composition index [CI]; *sensu* Curtis 1959) value for each size class: $CI = \sum (\text{Species Adaptation Value} \times \text{Species Importance Value})$. Adaptation Values were taken from Fralish (1988) and Fralish and others (1993). Shade-intolerant (early successional) species (for example, chestnut oak [*Q. prinus* L.], post oak [*Q. stellata* Wang.]) have low values (1-3), shade-intermediate species (for example, white oak [*Q. alba* L.], black oak [*Q. velutina* Lam.]) have values between 4 and 7, and shade-tolerant (late successional) species (for example, American beech and sugar maple) have high values (8-10). CI values ranged from 100 to 1000 and provided a means to compare strata within stands (trees vs. saplings vs. seedlings) and between stands (for example, trees vs. trees).

As an additional test of compositional stability, percent similarity (PERSIM) between overstory and understory strata was calculated: $PERSIM = 2W/a+b$ where “a” is the sum of the values in the overstory, “b” is the sum of the values in the understory, and “W” is the sum of the lesser of the two values for species common to both strata. However, when relativized data (IV_{100}) are used, as in the present study, the equation reduces to “W”. Similarity values of 55-90 percent indicate relatively similar to strongly similar strata; 45-55 percent, marginally similar strata; and <45 percent, dissimilar strata (Fralish and others 1991, Fralish and McArdle 2009).

RESULTS

SOUTH SLOPE ELT

The tree stratum of the south slope ELT community was dominated by white oak (IV = 39.3) and black oak (IV = 23.5), with northern red oak (*Q. rubra* L.) and pignut/red hickory (*Carya glabra* [Mill.] Sweet-Carya *ovalis* [Wang.] Sargent) complex of secondary importance (IV range = 5.1-7.4) (Table 1). Two-thirds of the 20 overstory species on this ELT were oak and hickory species with a combined importance of 88.6 (oak IV = 75.5; hickory IV = 13.1). Collectively, the importance of early successional species (IV = 89.8) was substantially greater than the importance of late successional species (8.6). Only seven late successional mesophytic species appeared in the sample. The CI value of 593 is indicative of a forest overstory dominated by a variety of oak and hickory species whose AVs range from 3.2 to 6.5. Community mean basal area was 24.9 m²/ha.

In contrast to the overstory, the dominant species in the sapling stratum were sugar maple (IV = 26.7), American beech (IV = 20.2), and red maple (IV = 17.2) (Table 2). Collective importance of late successional species was 71.1 and only 27.3 for early successional species. The combined importance of hickory species (total IV = 13.8) was greater than that of oak (9.5). The composition index of 889 indicated the high level of dominance by late successional species, which have AVs of 8-10. Sapling density was 980 stems/ha.

Similar to the sapling stratum, the seedling stratum also was strongly dominated by late successional species (collective IV = 61.8) but had a larger component of early successional species (Table 3). The most important species were red maple (IV = 17.4), American beech (IV = 14.6), black oak (IV = 16.6), sugar maple (IV = 8.8), white oak (IV = 7.6), and white ash (IV = 5.8). Twenty-five species were recorded. Density was 9,064 stems/ha and CI was 789.

Table 1.—Tree species' adaptation and importance values in four Ecological Land Types of the Illinois Ozark Hills region. Species with importance values >5 are in bold format. Adaptation values are from Fralish (1988) and Fralish and others (1993). The letter W indicates the adaptation value for the mesic to hydric section of the moisture gradient. Species number in the north slope ELT based on 20 plots.

Species	Adaptation Value	Ecological Land Type			
		South Slope	Broad Ridgetop	North Slope	Stream Terrace
Early Successional Xeric and Xeric-Mesic Species					
Chestnut Oak	1.0	0.1	3.3		
Winged Elm	2.1		0.1		
Post Oak	3.2	1.4	2.2		
Black Hickory	3.5	0.4	0.8		
Scarlet Oak	4.7	2.3	2.3	1.2	
Southern Red Oak	4.8	3.8			
Black Oak	5.0	23.5	27.3	10.8	
Mockernut Hickory	5.0	1.8	0.4	0.6	1.6
White Oak	5.4	39.3	30.5	15.8	2.9
Chinkapin Oak	6.5			0.5	1.5
Shagbark Hickory	6.5	3.1	1.3	2.2	0.1
Pignut/Red Hickory	6.5	7.4	11.1	7.2	1.0
Northern Red Oak	8.0	5.1	7.0	17.0	5.3
Yellow-Poplar	9.5	1.6	7.5	11.9	22.0
SUBTOTAL		89.8	93.8	67.2	34.4
Late Successional Mesophytic Species					
Blackgum	7.2	1.9	0.2	1.1	
White Ash	8.7	1.6	2.2	2.3	0.9
Hackberry	8.8			0.3	1.8
Black Cherry	8.8			0.4	2.7
Red Maple	9.1	0.2	0.3	1.0	
Cucumber-Tree	9.5	0.2	0.6	2.7	0.4
Bitternut Hickory	9.5	0.4		0.3	9.4
Slippery Elm	9.5	1.8		2.3	8.2
American Beech	9.6	2.5		14.7	0.4
Sugar Maple	10.0		2.7	5.6	11.0
Blue-Beech	10.0				0.7
KY Coffee-Tree	10.0				1.0
Black Walnut	10.0			1.2	3.1
SUBTOTAL		8.6	6.0	31.8	39.6
Early Successional Mesic-Hydric and Hydric Species					
American Elm	9.5W			0.1	0.3
Boxelder	7.0W				3.9
Swamp Chestnut Oak	6.0W				2.0
Sweetgum	4.0W	1.6	0.2	0.8	6.9
Eastern Sycamore	5.0W				12.9
SUBTOTAL		1.6	0.2	0.9	26.0
TOTAL		100.0	100.0	100.0	100.0
Number of Plots					
No. of Species		20	20	28	20
Basal Area (m²/ha)		18	19	20	22
Density (Stems/ha)		24.9	21.7	27.1	27.2
Composition Index		403	380	239	392
		593	593	774	747

Table 2.—Species' importance values for saplings in the communities of four Ecological Land Types in the Illinois Ozark Hills region. Species with importance values >5 are in bold format. Adaptation values are from Fralish (1988) and Fralish and others (1993). The letter W indicates the adaptation value for the mesic to hydric section of the moisture gradient. Number of species in the north slope ELT based on 20 plots.

Species	Adaptation Value	Ecological Land Type			
		South Slope	Broad Ridgetop	North Slope	Stream Terrace
Early Successional Xeric and Xeric-Mesic Species					
Chestnut Oak	1.0	2.0	0.5		
Persimmon	1.5	0.7	0.5		
Winger Elm	2.1	3.3	2.5		
Post Oak	3.2	2.3			
Black Oak	5.0	2.1	1.0		
Chinkapin Oak	6.0				1.8
White Oak	5.4	3.1	3.5	1.2	0.9
Pignut/Red Hickory	6.5	8.0	10.9	2.3	
Shagbark Hickory	6.5	5.8	1.0	0.8	
SUBTOTAL		27.3	19.9	4.3	2.7
Late Successional Mesophytic Species					
Blackgum	7.2	1.6	1.0	1.6	8.1
White Ash	8.7	0.8	2.0	1.2	0.9
Black Cherry	8.8		1.0		
Red Maple	9.1	17.2	6.4	9.0	
Cucumber-Tree	9.5				0.9
Slippery Elm	9.5	0.9		0.8	9.0
Bitternut Hickory	9.5	2.9	1.5	1.2	6.3
American Beech	9.6	20.2	22.2	29.9	11.7
Sugar Maple	10.0	26.7	46.0	46.5	28.9
Ohio Buckeye	10.0				1.8
Blue-Beech	10.0	0.8		0.4	28.8
SUBTOTAL		71.1	80.1	90.6	96.4
Early Successional Mesic-Hydric and Hydric Species					
American Elm	9.5W	1.6		4.7	
Sweetgum	4.0W			0.4	0.9
SUBTOTAL		1.6	0.0	5.1	0.9
TOTAL		100.0	100.0	100.0	100.0
Number of Plots		20	20	27	20
No. Species		16	15	11	12
Density (Stems/ha)		980	779	973	580
Composition Index		889	852	958	948

Table 3.—Species' adaptation and importance values for seedlings in the communities of four Ecological Land Types of the Illinois Ozark Hills region. Species in each group are listed from lowest to highest adaptation value (AV). Adaptation values are from Fralish (1988) for the Shawnee Hills, IL, and Fralish and others (1993) for Land Between the Lakes, KY and TN. Importance values greater than 5 percent in bold. The letter W indicates the adaptation value for the mesic to hydric section of the moisture gradient. Species number for north slope ELT based on 20 plots.

Species	Adaptation Value	Ecological Land Type			
		South Slope	Broad Ridgetop	North Slope	Stream Terrace
Early Successional Xeric and Xeric-Mesic Species					
Chestnut Oak	1.0		1.7		
Persimmon	1.5	1.6	2.2	0.7	0.6
Winged Elm	2.1	1.2	0.7		0.4
Post Oak	3.2	0.6			
Scarlet Oak	4.7	0.2	0.2		
Black Oak	5.0	16.6	13.9	1.5	0.6
Mockernut Hickory	5.0		0.5		
White Oak	5.4	7.6	6.2	5.7	0.6
Chinkapin Oak	6.5	0.4	0.2	0.4	2.7
Shagbark Hickory	6.5	3.5	6.0	1.3	
Pignut/Red Hickory	6.5	1.8	3.3	1.9	
Northern Red Oak	8.0	2.9	1.9	5.0	1.2
Yellow-Poplar	9.5	0.6	1.4	7.0	0.6
SUBTOTAL		37.0	38.2	23.5	6.7
Late Successional Mesophytic Species					
Blackgum	7.2	1.6	2.9	2.2	0.4
White Ash	8.7	5.8	9.8	9.6	11.2
Hackberry	8.8	0.2			0.2
Black Cherry	8.8	1.2	5.0	0.6	5.6
Red Maple	9.1	17.7	17.0	12.0	
Cucumber-Tree	9.5	0.2		1.3	0.2
Bitternut Hickory	9.5	2.9	2.4	4.8	11.1
Slippery Elm	9.5	7.3	3.6	4.3	40.1
American Beech	9.6	14.6	14.6	21.7	6.6
Sugar Maple	10.0	8.6	6.5	16.3	5.8
Black Walnut	10.0			0.2	0.2
Butternut				0.2	1.9
Blue-Beech	10.0	1.9		0.7	1.3
American Basswood	10.0			0.4	
SUBTOTAL		62.0	61.8	74.3	84.6
Early Successional Mesic-Hydric and Hydric Species					
American Elm	9.5W			2.0	5.8
Boxelder	7.0W				1.9
Swamp Chestnut Oak	6.0W				0.4
Shellbark Hickory	5.5W	1.0			0.2
Sweetgum	3.0W			0.2	0.4
SUBTOTAL		1.0		2.2	8.7
TOTAL		100.0	100.0	100.0	100.0
Number of Plots		20	20	28	20
No. of Species		18	19	20	22
Basal Area (m²/ha)		24.9	21.7	27.1	27.2
Density (Stems/ha)		403	380	239	392
Composition Index		593	593	774	747

An average community structure graph shows the extent of mesophyte invasion into oak-hickory dominated forest (Fig. 1A). The cohort invasion appears more as a wave pattern rather than continuous input. The graph shows that in general there is little similarity between the sapling and tree strata, and particularly with the 60+ cm tree size class.

BROAD RIDGETOP ELT

The ridgetop ELT overstory was strongly dominated by early successional species (combined IV = 93.8). The most important species were white oak (IV = 30.5) and black oak (IV = 27.3), followed by pignut/red hickory (IV = 11.1), yellow-poplar (*Liriodendron tulipifera* L.) (IV = 7.5), and northern red oak (IV = 7.0) (Table 1). Oak and hickory species had a combined importance of 86.2 (oak IV = 72.6; hickory IV = 13.6). The five mesophytic species had a combined importance of only 6.0. The CI value of 593 was identical to that of the south slope ELT. Community mean basal area was 21.7 m²/ha, somewhat higher than the basal area for the south slope ELT.

The sapling stratum was dominated by mesophytic species. Sugar maple was the major species (IV = 46.0), along with American beech (IV = 22.2) (Table 2). The strong dominance by sugar maple and beech is reflected in the CI value of 852. Oak species had a combined IV of only 5.0 and hickory an IV of 11.9. Early successional species formed only 21.4 percent of the community. Only 14 species were present.

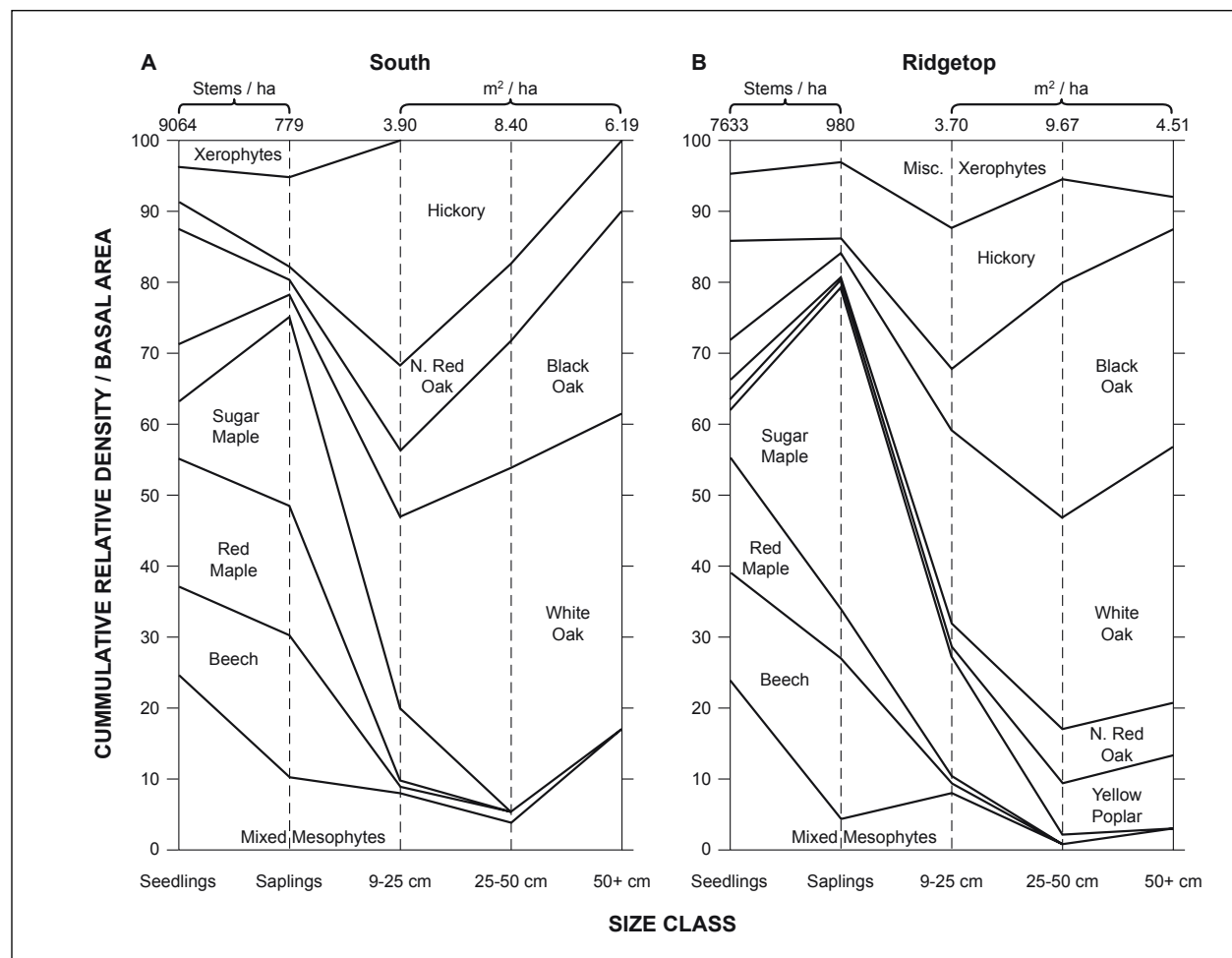


Figure 1.—Forest community structure diagrams for the south slope and ridgetop Ecological Land Types.

As in the south slope ELT, red maple was the leading dominant (IV = 17.0) in the seedling stratum, followed closely by American beech (IV = 14.6), black oak (IV = 13.9), and white ash (*Fraxinus americana* L.) (IV = 9.8) (Table 3). Importance values for shagbark hickory (*Carya ovata* [Mill.] K. Koch), sugar maple, and black cherry (*Prunus serotina* Ehrh.) ranged between 5.0 and 6.5.

Compared to the sapling stratum, the stronger importance of oak and hickory species and a reduced importance of sugar maple resulted in a lower CI (722). The number of species (21) was greater than in the tree and sapling strata primarily because of the increased number of oak and hickory species. Density was 7,633 stems/ha. The general structure of the ridgetop ELT community (Fig. 1B) is similar to that of the south slope ELT community.

NORTH SLOPE ELT

The tree stratum of the north slope community was dominated by a mixture of species, including northern red oak (IV = 17.0), white oak (IV = 15.8), American beech (IV = 14.7), yellow-poplar (IV = 11.9), black oak (IV = 10.8), and sugar maple (IV = 5.6) (Table 1). Compared with the community on south and ridgetop ELTs, dominance shifted to a more equitable balance among species. However, oak dominated the community with a combined IV of 45.3. Three hickory species had a total IV of 10.3 for a total oak and hickory importance of 55.6. Thirteen mesophytic species were present. The CI (774) indicates the presence of a combined component of early and late successional species (IVs = 7.1-10).

The sapling stratum had a larger mesophytic component and less equitability than the mixed overstory canopy. Sugar maple (IV = 46.5), beech (IV = 29.9), and red maple (IV = 9.0) had a combined importance of 90.6 (Table 2). Oak and hickory had a combined IV of only 4.3, the total importance for early successional species. The CI of 958 (maximum 1,000) reflects the composition of nearly pure late successional species.

Compared to the sapling stratum, the seedling layer was somewhat less strongly dominated by mesophytic species (CI = 892) but had a greater equitability among four species: American beech (IV = 20.8), sugar maple (IV = 17.3), red maple (IVN=12.0), and white ash (IV = 9.7) (Table 3). The combined IV of early successional species is only 23.5 while that of late successional species is 74.3.

Compared to the south slope or ridgetop ELT communities, the general structure of the north slope ELT community (Fig. 2A) reflected a more rapid invasion by mesophytes. Both sugar maple and beech have a strong wave-form invasion pattern although the two patterns are not synchronized.

STREAM TERRACE ELT

A mixture of upland early and late successional species and bottomland species dominated the stream terrace ELT community in approximately equal numbers (combined IV = 34.4, 39.9, and 25.7, respectively) (Table 1). Oak and hickory had a collective IV of 18.5. Yellow-poplar was the only single major early successional species (IV = 22.0), and it is considered more of a mesophyte than a species of dry sites (Beck 1990). Dominance by the late successional species was nearly equal among sugar maple (IV = 11.0), bitternut hickory (*C. cordiformis* [Wangenh.] K. Koch) (IV = 9.4), and slippery elm (*Ulmus rubra* Muhl.) (IV = 8.2). Sycamore (*Platanus occidentalis* L.) (IV = 12.9) and sweetgum (*Liquidambar styraciflua* L.) (IV = 6.9) are bottomland species that commonly invade mesic sites after disturbance. Compared to upland sites, this community had the highest basal area (27.2 m/ha), with a CI value of 747, reflecting the mixture of early

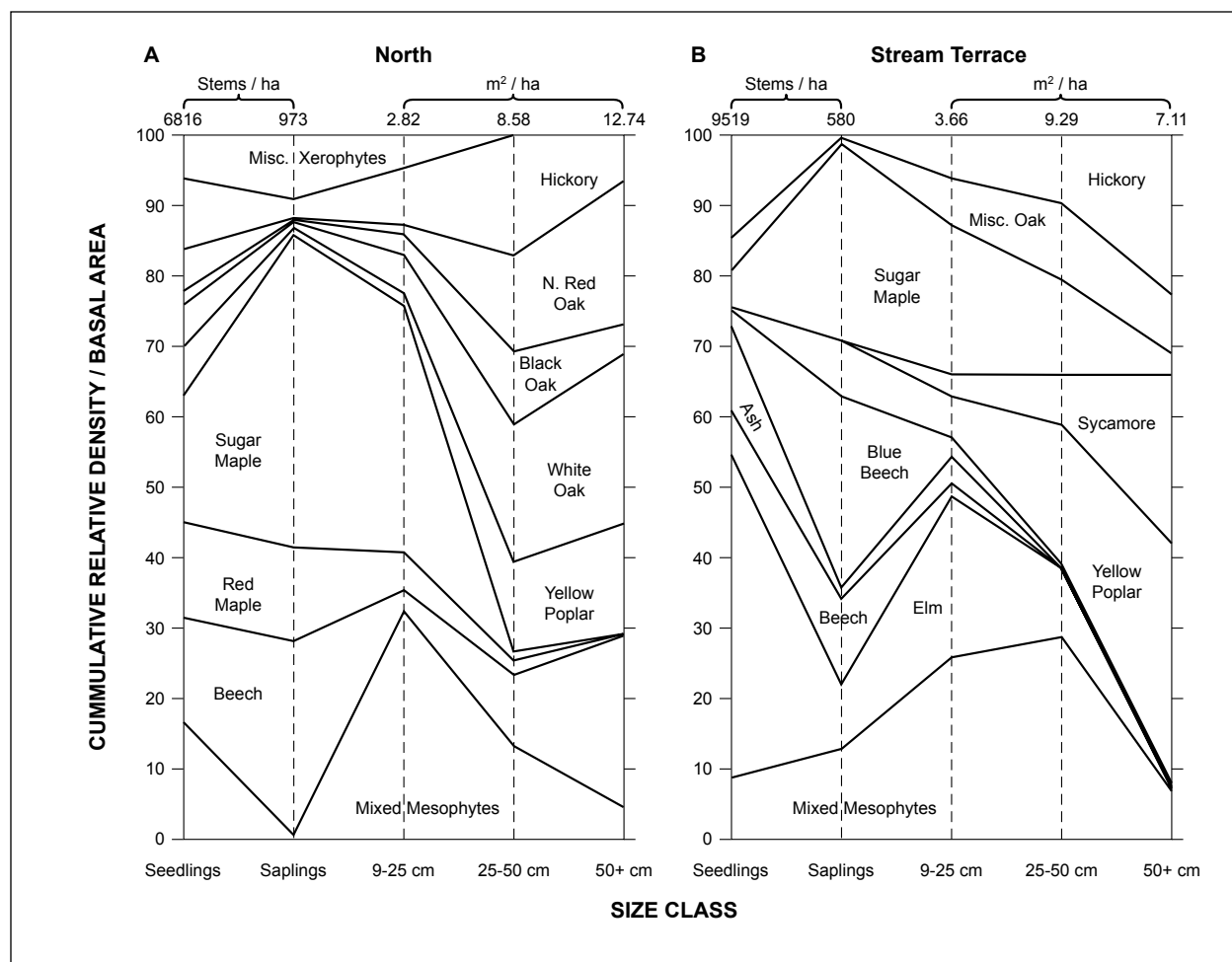


Figure 2.—Forest community structure diagrams for the north slope and stream terrace Ecological Land Types.

and late successional species and bottomland species. Species richness was 22, of which about 75 percent was mesophytic species and 25 percent was oak and hickory species.

The sapling stratum was dominated equally by two mesophytes—sugar maple (IV = 28.9) and blue-beech (*Carpinus caroliniana* Walt.) (IV = 28.8)—but American beech (IV = 11.7) and slippery elm (IV = 9.0) also were important (Table 2). However, blue-beech does not have the genetic potential to grow into the large-size tree classes (30+ cm d.b.h.). Excluding blue-beech from the potential overstory canopy, the IVs of the major mesophytes were sugar maple, 40.4; American beech, 16.2; blackgum, 11.1; slippery elm, 9.9; and yellow-poplar, 8.7. Collectively, the combined importance of mesophytes in the sapling stratum was more than 90.5 percent. When yellow-poplar is included as a cross-over mesophyte (it is also early successional), the IV is 96.4.

Species composition of the seedling stratum was distinctly different from that of the tree and sapling strata. Seedlings were dominated by slippery elm (IV = 40.1), with white ash (IV = 11.2) and bitternut hickory (IV = 11.1) as secondary species (Table 3). The importance of maple and American beech was relatively low (IV = 5.8 and 6.6, respectively). Composition index was high (846) although sugar maple and beech did not have high importance values.

Structurally, sugar maple and other mesophytes have already progressed well into the tree canopy, with diameters of a few trees exceeding 50 cm (Fig. 2B). Sugar maple, slippery elm, and other mesophytes are also invading in wave-form patterns that are not synchronized.

COMMUNITY AND STRATA SIMILARITY

The three upland ELTs (south slope, ridgetop, and north slope) are relatively similar in composition and structure. The tree strata of the south slope and ridgetop communities were most similar in composition (PERSIM = 76.5 percent) although there also was high similarity between the ridgetop and north slope (PERSIM = 69.3 percent). The south slope and north slope tree strata were marginally similar (PERSIM = 53.0 percent). For these upland ELTs, the sapling strata also are similar to each other (PERSIM = 66.6-75.9 percent), as are the seedling strata (PERSIM = 64.1-79.4 percent).

In contrast, there is little similarity between the tree overstory and the understory for the upland ELTs. Percent similarity between the tree and sapling strata for these three ELTs ranged from 25.1 to 30.4 and between the tree and seedling strata from 35.4 to 44.9. Similarity between the tree and seedling strata was greater than between the tree and sapling strata because of seedlings (established and ephemeral) of early successional species. Here “ephemeral” refers to seedlings produced from recently germinated seed; most of these seedlings are not likely to live more than a few months or a year.

Similarity values between the seedling and sapling strata were not consistent. The composition of these strata was similar for the south slope ELT (PERSIM = 72.2) and the north slope ELT (PERSIM = 58.4), but was dissimilar for the ridgetop (PERSIM = 42.4). The dissimilarity between the strata for the ridgetop is primarily attributed to the high importance of sugar maple in the sapling stratum compared to its low importance in the seedling stratum.

The composition of the stream terrace tree canopy was extremely dissimilar (PERSIM = 11.6-25.2 percent) from the canopies of the other ELTs. Similarly, the stream terrace sapling stratum and seedling stratum are relatively dissimilar from their counterparts on the upland ELTs (PERSIM = 44.9-46.4 percent and 37.2-38.8 percent, respectively). The sapling and seedling strata of the stream terrace also are dissimilar (PERSIM = 27.6 percent). This low similarity is attributed to the wide difference in IVs for slippery elm and blue-beech (Table 3).

DISCUSSION

In the early 1930s, large areas of land in the Ozark Hills were abandoned and subsequently purchased by the U.S. Department of Agriculture, Forest Service (Shawnee National Forest) and the State of Illinois (Trail of Tears State Forest). Since that time and with the establishment of homes in the region, fire is no longer a factor influencing forest community composition. Timber harvesting also has been minimal and generally absent on public land for the past 30+ years. The result is that in the absence of disturbances, major composition changes are occurring within this regional forest (Osier and others 2006, van de Gevel and Ruffner 2006, Fralish and McArdle 2009).

Data from plots concentrated within the 144-ha Ozark Hills Nature Preserve and from plots systematically located on section and quarter-section corners distributed throughout the region leave little doubt that succession—that is, the replacement of overstory oak and hickory by mesophytic species—is rapidly occurring

(Helmig 1997). In related studies in the Ozark Hills, Groninger and others (2002) and Osier and others (2006) reported a substantial change from an oak-dominated forest toward a maple-dominated forest based on data from plots established in 1980 and remeasured in 2000. Nearby, Weaver and Ashby (1971) and Shotola and others (1992) found a similar pattern over a 36-year period in Weaver's Woods.

The analysis of forest community data by size class strata (Figs. 1 and 2) outlines the present similar encroachment pattern in the upland ELT communities, which would be expected to continue over time (structure-for-time analysis) and in the absence of active forest management or a major disturbance. The forest overstory on south, ridgetop, and north slopes is presently dominated by oak and hickory (CI = 593-774) while dominance is about equally shared with mesophytic species on the stream terrace. Composition index values from near 800 to more than 950 in both the sapling and seedling strata indicate the strong dominance by mesophytic species that ultimately will be the next forest as the overstory trees die or are cut.

MOISTURE-ELT COMMUNITY RELATIONSHIPS

An implied soil moisture-evapotranspiration gradient is expressed by the sequence of ELTs from the mesic south slope and ridgetop ELTs to the relatively more mesic north slope and strongly mesic stream terrace ELTs. Fralish and McArdle (2009) split out a southwest ELT, but there was no observable difference in composition between it and the south ELT. However, the basal area of the southwest slope community was the lowest for all ELT communities, suggesting the southwest slope may be somewhat less productive.

In the Ozark Hills, the south slope ELT often is relatively steep (30-60 percent), with only 100-200 cm of soil depth although the soil often is deeper. The south slope and ridgetop ELTs receive the full impact of the growing season sun and rapidly lose soil moisture. The ridgetop ELT may have up to a 10-m loess deposit for buffering this loss, but frequently a fragipan at a depth of 60-70 cm limits tree root penetration. Because of the angle of incoming solar radiation, the north ELT has a reduced solar radiation load compared to the south slope, and often experiences topographic shading during early and late daylight hours. Stream terraces often have deep soil; receive runoff from adjacent slopes; and are shaded for various parts of the day, making them moist and relatively cool.

The importance of early successional species in the tree stratum generally follows the moisture-ELT gradient. The importance of early successional species decreases from an IV of 89.8 and 93.8 on the south slope and ridgetop, respectively, to 67.2 on the north slope and 34.4 on the terrace. The heavy disturbance by wildfire and timber harvesting during the mid- to late 1800s and early 1900s generally opened sites for establishment of the oak-hickory community across the gradient (Fralish and McArdle 2009).

The pattern for late successional species' importance is the reverse of that for early successional species. On the drier south and ridgetop ELTs, late successional tree species have been slow to invade or survive and thus have low importance (IV = 8.6 and 6.0, respectively) (Table 2). The heavy disturbance, probably combined with high soil surface temperatures, apparently had a substantial impact on mesophytic species. On the cool, moist north and alluvial ELTs, IVs for mesophytes increase to 31.9 and 39.9, respectively. On mesic sites, survival rates apparently have been higher and invasion more rapid. The result of these two reverse patterns for early and late successional species is that overstory CI increases from 593 on the south slope to 747 on the terrace ELT.

Community basal area also is related to ELT environment, with moderate values on the more xeric ELTs (24.9 and 21.7 m²/ha) and higher values on the moist ELTs (27.1 and 27.2 m²/ha). Because of the deep, moist soil and/or low evapotranspiration on cool slopes, the basal area is likely to increase to as high as 35 m²/ha, a level similar to that reported for several stands on mesic sites in the Shawnee Hills (Fralish 1976, 1988, 1994; Fralish and others 1991) and for the central states in general (Held and Winstead 1975). For permanent plots on a mesic site in the Kaskaskia Experimental Forest of southern Illinois, basal area increased from 19.8 m²/ha in 1935 to 31.1 m²/ha in 1973 (Schlesinger 1976) to 32.0 m²/ha in 1978; by 1983, the basal area decreased slightly to 30.2 m²/ha as several large oak trees died (Fralish 1988). In 1997, basal area was 34.3 m²/ha (Zaczek and others 2002). However, basal area is likely to continue to increase somewhat as the stand is becoming progressively dominated by sugar maple.

In the tree stratum, species richness generally increases across the sequence of sites from south slopes and ridgetops to mesic north slopes and terraces. This pattern can be attributed to various combinations of early and late successional and bottomland species. No similar patterns exist for either the sapling or seedling strata. Vertically, however, there is a distinct pattern of species richness. Lowest species richness is found in the sapling stratum (11-16 species), with the highest richness in the seedling stratum (21-25 species) and intermediate values in the tree stratum (18-23 species). A number of species in the tree stratum (scarlet oak [*Q. coccinea* Muenchh.], southern red oak [*Q. falcata* Michx.], northern red oak, black hickory [*Carya texana* Buck.], and mockernut hickory [*Carya tomentosa* (Poir.) Nutt.]) are not found in the sapling layer but reappear here in small numbers in the seedling stratum along with stems of nearly all mesophytic species. The large species number in the seedling stratum results from a combination of older established seedlings and a continuous influx of recently established seedlings of shade-intolerant and -intermediate tree species that are likely to live only a short while (ephemerals). The tree stratum probably was initiated under relatively open (moderately disturbed) conditions, which allowed a large number of species to survive and grow to become trees. The saplings developed under the present forest overstory and with sufficient stress to reduce the number of species.

DISTURBANCE EFFECTS

Oak and hickory have dominated the Central States forest for about the past five centuries (Delcourt and Delcourt 1987, Franklin 1994). Given the deep soil, high water availability, lack of disturbance, and a long time period, the entire Ozark Hills forest will eventually be dominated by mesophytes to the near-exclusion of early successional species. Ecologists believe that fire and fire interacting with microclimate and topography were the factors that allowed forest stands to be dominated by oak and hickory and prevented domination of the overstory canopy by sugar maple, red maple, and beech. Thus, we agree with Van Lear (1988), Abrams (1992), Olson (1996), Ruffner and Groninger (2004), and Nowacki and Abrams (2008) that fire was a major factor in the development of early and present oak-hickory communities.

Forests protected from disturbances such as fire, grazing, and logging will change over time as a result of small-scale disturbances, such as the death of dominant or codominant trees, which create canopy gaps. Sugar maple is an extremely competitive species in the gap-phase replacement process, as demonstrated in various studies (Parker and others 1985, Parker and Sherwood 1986, Ward and Parker 1987, Parker and Ward 1987, McCarthy and others 1987, McCarthy and Wistendal 1988, Spetich and Parker 1998). Edgington (1991) came to a similar conclusion and Lin and Augsburg (2008) found a population explosion of sugar maple in Brownfield Woods (Illinois), where permanent plots were established in 1925 (Telford 1926). Shade-tolerant

species such as sugar maple and beech can withstand long periods of suppression in the understory (Burns and Honkala 1990, Cho and Boerner 1991). Abrams and Downs (1990) studied the successional change of an old-growth white oak forest in Pennsylvania and reported that logging operations during the 1930s and 1940s released understory mesophytic species.

CONCLUSIONS AND RECOMMENDATIONS

In the absence of disturbance, the forest communities of the Illinois Ozark Hills ultimately will be dominated by mesophytic species. However, with the well developed midcanopy of mesophytic species, any disturbance that creates gaps in the overstory is likely to accelerate the successional process. The recent severe windstorm (derecho of 8 May 2009) with maximum wind velocity in excess of 100 mph is an example of such a disturbance. Many large overstory trees were blown over in the northern section of the Ozark Hills. The development of the forest in these patches should be monitored.

Conversion is likely to be a long, continuous process as the mesophytic component of the oak-hickory forest is still developing. However, it should be of interest to resource managers and scientists to have an estimate of the threshold time of conversion. In a preliminary analysis, Helmig (1997) used stand table projection and basal area values for early and late successional species. The analysis indicated that the time of conversion may be between 2040 and 2050, but additional analysis is necessary.

From now until the basal area (biomass) of mesophytic species equals and exceeds that of oak and hickory, there will be important changes in the forest ecosystem. More than 45 years ago, Voigt and Mohlenbrock (1964) strongly suggested that the tree and floristic diversity of the Illinois Ozark Hills was comparable to that of the Great Smoky Mountains. An important question is what effect succession and conversion will have on maintaining an ecosystem of exceptional biodiversity. Fralish (1997, 2004) and Fralish and McArdle (2009) have expressed major concern regarding the loss of herbaceous species resulting from a multi-layered overstory, increased leaf area, low light levels at the forest floor, and a deeper litter layer. It is our opinion that the successional process for the entire region has advanced beyond control; moderate fire and physical removal of larger trees of mesophytic species would be required to regenerate oak. In view of the situation, changes in tree, shrub, and herbaceous species and density should be carefully monitored in the Illinois Ozark Hills, which we maintain probably will be the first large-scale oak-hickory ecosystem to convert to one dominated by maple and beech.

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EFFICACY OF HERBICIDE TREATMENTS FOR CONTROLLING RESIDUAL SPROUTING OF TREE-OF-HEAVEN

Daniel Bowker and Jeff Stringer¹

Abstract.—Tree-of-heaven (*Ailanthus altissima* [Mill.] Swingle) is one of the most widely distributed exotic invasive tree species in the United States and has become naturalized throughout the central hardwood forest. It opportunistically establishes in areas associated with silvicultural regeneration treatments and extirpation often requires the targeting of individual trees due to the close proximity of native co-occurring stems. Further, tree-of-heaven is a prolific root and stump sprouter, and extirpating this exotic invasive tree species requires not only killing the aboveground stem but also ensuring that lateral roots are killed. The study was designed to test the efficacy of five individual tree treatments in killing stems and preventing sprouting: EZ-Ject® (granulated glyphosate), full basal bark (triclopyr ester), hack and squirt (picloram/2,4-D), hack and squirt (glyphosate), and tree injection (picloram/2,4-D). The mortality and sprouting of sapling-sized tree-of-heaven developing in two naturally regenerating hardwood stands were investigated. We treated 410 stems. Treatments provided effective top kill compared to the untreated control group with treatment top dieback figures ranging from 91 to 100 percent compared to less than 15 percent for the untreated control. Treatment areas were examined for the occurrence and type of sprouts (basal stem, root collar, and lateral root). The EZ-Ject® glyphosate treatment exhibited 33-percent total sprouting in 2006 and 5 percent in 2007 compared to a range of 3 to 12 percent for other treatments in 2006 and 0 to 2 percent for all other treatments in 2007. This study indicates that all forms of sprouting should be considered when control options are tested and that EZ-Ject® applications with glyphosate may not provide thorough control of sprouting.

INTRODUCTION

Tree-of-heaven (*Ailanthus altissima* [Mill.] Swingle) is an exotic invasive tree species introduced into the United States in the 18th century; it is now naturalized throughout much of North America (Burch and Zedaker 2003). It is an invasive pioneer species, showing prolific root and stump sprouting, producing large amounts of seed, and undergoing rapid juvenile growth, with the potential to create pure stands where it invades (Burch and Zedaker 2003). Tree-of-heaven is phenotypically plastic, adjusting to various temperature, humidity, light, and moisture levels (Meloche and Murphy 2006), and is not limited by soil stoniness or pH, or urban pollution levels (Burch and Zedaker 2003). The species typically invades along forest edges, infiltrating the interior forest by way of roads, trails, and disturbance gaps (Meloche and Murphy 2006). Though the species is intermediate to intolerant of shade, root sprouts from its shallow, spreading root system grow slowly and can survive for years under an intact canopy, enabling this species to respond to disturbance (Burch and Zedaker 2003, DiTomaso and Kyser 2007). These ecological factors contribute heavily to tree-of-heaven's ability to invade disturbed areas in central hardwood forests, and spread into undisturbed areas.

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While management for tree-of-heaven is not yet well established (Meloche and Murphy 2006), several studies over the past few years have begun to explore control methods (Johnson and others 2001, Burch and Zedaker 2003, Lewis and McCarthy 2006, Meloche and Murphy 2006, DiTomaso and Kyser 2007, Lewis 2007, Lewis and McCarthy 2008). Studies have found that control methods without herbicide application result in worsening the infestation through the abundance of basal and lateral root sprouting (Pannill 2009, Burch and Zedaker 2003, Meloche and Murphy 2006, DiTomaso and Kyser 2007). Among studies that did use herbicide control, however, results have been mixed. Burch and Zedaker (2003) found that various full basal bark treatment combinations of triclopyr, imazapyr, and picloram all completely controlled stump sprouting. The best top kill resulted from a combination of 20 percent triclopyr and 5 percent picloram (Burch and Zedaker 2003). DiTomaso and Kyser (2007) looked at triclopyr, imazapyr, and glyphosate use in four different treatment types: cut stump, full basal bark spray, stem injection, and stump injection. They found that imazapyr provided good control of stems 1.6 to 6.4 inches in diameter at knee height, and a 75-percent reduction in sprouting percent and sprout size for all types of sprouts, compared to treatments without herbicides. The cut stump treatments with imazapyr and triclopyr both provided excellent control of all sprouts compared to the glyphosate treatment; however, stump injection provided unacceptable control regardless of herbicide, and stem injection provided the best control with imazapyr and glyphosate, but not with triclopyr (DiTomaso and Kyser 2007). Finally, imazapyr and triclopyr both provided excellent control when used as a full basal bark spray (DiTomaso and Kyser 2007). Johnson and others (2001) found that triclopyr alone or in combination with imazapyr in a full basal bark spray provided effective canopy mortality and sprout control. Lewis (2007) found that stem injection using the EZ-Ject® (distributed by ArborSystems, Omaha, NE) herbicide system resulted in effective stem mortality with use of imazapyr and glyphosate, but not with triclopyr, though all three herbicides completely controlled sprouting. Meloche and Murphy (2006) found that a cut stump treatment with glyphosate was most effective at managing juvenile stems (less than 23.6 inches in height and less than 2 inches in diameter at breast height [d.b.h.]) that were too small on which to effectively use EZ-Ject®. EZ-Ject® with glyphosate effectively controlled more mature stems (greater than 23.6 inches in height and greater than 2 inches d.b.h.) but did not effectively control sprouts.

Our study reports the results of treating tree-of-heaven stems ranging from 0.1 to 3.9 inches d.b.h. in two naturally regenerating hardwood stands with five methods: tree injection using the EZ-Ject® lance with glyphosate capsules, full basal bark application using 25-percent triclopyr (Garlon 4® [Dow AgroSciences LLC, Indianapolis, IN]), hack and squirt using 100-percent picloram/2,4-D (Pathway [Dow AgroSciences]), liquid tree injection with 100-percent picloram/2,4-D (Pathway®) using the Jim-Gem® tree injector (Forestry Suppliers, Jackson, MS), and hack and squirt with 100-percent glyphosate (Accord® [Dow AgroSciences]). The objective of our study was to increase the amount of available tree response data on herbicide options for controlling existing stems as well as basal stem, root collar, and lateral root sprouting of tree-of-heaven.

STUDY AREAS

This study took place during the growing seasons of 2006 and 2007 on the University of Kentucky's Robinson Forest, a 15,000-acre experimental forest located in Breathitt, Knott, and Perry Counties in eastern Kentucky. The forest is in the southern section of the central hardwood forest region (longitude -83.14°W, latitude 37.47°N), and is composed of mixed mesophytic and oak-hickory forest types. Two stands were used for this study. Stand 1 is located on the upper third of a southwest-facing slope. Stand 2 occurs on the upper third of a north-facing slope. Both stands received a deferment harvest, retaining 10-15 ft² of basal area per acre of dominant and co-dominant trees. The stands were allowed to regenerate naturally and tree-of-heaven invaded

the stands during this time. Individual tree-of-heaven stems were scattered throughout the native regenerating age class. Stand 1 regenerating age class averaged 2 to 3 inches d.b.h. and was 8 years old. Stand 2 regenerating age class averaged 1 to 2 inches d.b.h. and was 5 years old. We treated 213 tree-of-heaven stems (0.2 to 3.9 inches d.b.h.) in June 2006 and 197 tree-of-heaven stems (0.1 to 2.3 inches d.b.h.) in June 2007.

MATERIALS AND METHODS

Each of five two-person crews was randomly assigned one of the treatments. Crews were then randomly assigned transects approximately 20 feet in width through each of the stands. Individual tree-of-heaven stems were treated and marked for future measurements. No other species were treated.

Table 1 shows the five treatments, number of stems treated by treatment type, and year treated. Treatments were as follows: (1) EZ-Ject® herbicide system with glyphosate capsules, at the manufacturer's recommended rate of one capsule per stem for trees less than 2.5 inches d.b.h. and one capsule for every 4 inches of circumference for trees larger than 2.5 inches d.b.h.; (2) full basal bark spray with 25 percent triclopyr (Garlon 4® and 75 percent nonpolar carrier, wetting the entire circumference of the lower 16 inches of the stem; (3) hack and squirt, using 0.03 fl. oz. of undiluted Pathway® per 2 to 3 inches of stem circumference; (4) tree injector with 0.03 fl. oz. of undiluted (Pathway® per 2 to 3 inches of stem circumference; and (5) hack and squirt with 0.03 fl. oz. of undiluted Accord® per 2 to 3 inches of stem circumference. In 2006, 50 tree-of-heaven stems scattered throughout the treated stand were missed during treatment implementation and were used as untreated controls for the study. No stems were missed during 2007, so there is no control for that year.

One year after herbicide treatment, all treated stems were placed into percent top dieback classes, using a visual estimate of the canopy area that had died since the treatment. Each treated stem was surveyed for associated sprouts. Each sprout was classified as to its provenance, falling into one of three categories: lateral root (lateral root sprouts that were difficult to visually assign to individual stems were excavated and traced to the parent tree), root collar, or basal stem (those sprouts occurring between the root collar and approximately 1 foot above the ground).

The GLM procedure of the SAS statistical software package (SAS Institute, Cary, NC) was used to determine whether there were differences in the percent of treated stems with sprouts ($100 \times [\# \text{ of stems with sprouts} / \text{total stems treated per treatment}]$), at the $\alpha=0.05$ level. Years 2006 and 2007 were analyzed separately.

Table 1.—Five treatments used for tree-of-heaven control in this study, including delivery method, herbicide used, and total number of stems treated by year.

Treatment	Delivery method	Herbicide	No. of stems treated 2006	No. of stems treated 2007
0 (control)	None	None	50	0
1	EZ-Ject® lance	Glyphosate	18	43
2	Basal bark spray	Triclopyr	38	48
3	Hack and squirt	Picloram/2,4-D	39	26
4	Jim-Gem® tree injector	Picloram/2,4-D	34	39
5	Hack and squirt	Glyphosate	84	41

RESULTS

Overall, average top dieback was greater than 91 percent for all treatments in both years (Table 2), compared to 14 percent for the untreated control group in 2006.

Table 3 shows the percent of treated tree-of-heaven stems with living sprouts 1 year after treatment. In 2006, 12 percent of the untreated control group exhibited living sprouts, which was not statistically different from all other treatments, with the exception of the EZ-Ject glyphosate treatment. The EZ-Ject® with glyphosate treatment resulted in 33 percent of the trees sprouting, significantly higher than the other treatments, which ranged from 3 to 12 percent (Table 3). Sprouting for the EZ-Ject® glyphosate treatment was composed of 28 percent lateral root sprouts, 66 percent root collar sprouts, and 6 percent basal stem sprouts (Table 4).

Table 2.—Average percent top dieback by treatment and year.

Treatment	Delivery method	Herbicide	Average percent top dieback 2006	Average percent top dieback 2007
0 (control)	None	None	14	
1	EZ-Ject® lance	Glyphosate	100	100
2	Basal bark spray	Triclopyr	91	100
3	Hack and squirt	Picloram/2,4-D	100	96
4	Jim-Gem® tree injector	Picloram/2,4-D	100	100
5	Hack and squirt	Glyphosate	96	96

Table 3.—Percent treated stems with sprouts by treatment and year. Letters show differences among population means within that year.

Treatment	Delivery method	Herbicide	Percent treated stems with sprouts 2006	Percent treated stems with sprouts 2007
0 (control)	None	None	12 a	
1	EZ-Ject® lance	Glyphosate	33 b	5 a
2	Basal bark spray	Triclopyr	3 a	0 a
3	Hack and squirt	Picloram/2,4-D	3 a	0 a
4	Jim-Gem® tree injector	Picloram/2,4-D	12 a	0 a
5	Hack and squirt	Glyphosate	11 a	2 a

Table 4.—Location of sprouts by treatment and year. Figures are percent of total sprouts in each location.

Treatment	Delivery method	Herbicide	Percent of total sprouts from lateral root		Percent of total sprouts from root collar		Percent of total sprouts from basal stem	
			'06	'07	'06	'07	'06	'07
0 (control)	None	None	37		0		64	
1	EZ-Ject® lance	Glyphosate	28	64	66	0	6	36
2	Basal bark spray	Triclopyr	100	0	0	0	0	0
3	Hack and squirt	Picloram/2,4-D	100	0	0	0	0	0
4	Jim-Gem® tree injector	Picloram/2,4-D	75	0	25	0	0	0
5	Hack and squirt	Glyphosate	53	0	47	0	0	100

For the 2007 treatments, the EZ-Ject® with glyphosate exhibited 5 percent stems with living sprouts, compared to hack and squirt with glyphosate at 2 percent (Table 3). The basal bark treatment with triclopyr, hack and squirt with picloram/2,4-D, and tree injector with picloram/2,4-D all showed no indications of sprouting. None of the 2007 treatments were statistically different. Sprout location percentages are shown in Table 4.

DISCUSSION

The project provided information relative to sprouting of tree-of-heaven subjected to methods and herbicides commonly used for eradication and control. Sprouting, particularly from lateral roots, is important to understand and control, especially for this species, which has a high propensity for this type of response to top kill. All of the treatments incorporating glyphosate, triclopyr, and picloram/2,4-D resulted in levels of top kill that would be viewed as an operational success (> 91 percent).

All of the treatments with the exception of the EZ-Ject® with glyphosate were found to produce no more sprouting than was associated with trees that were not treated. In 2006, 33 percent of EZ-Ject® glyphosate-treated stems showed live sprouts 1 year after treatment. In 2007, no method was statistically different from any other, but we noted a trend associated with a higher average sprouting percentage for the glyphosate treatments compared to the others. Meloche and Murphy (2006) also found that the EZ-Ject® with glyphosate did not effectively control sprouting. It is possible that the EZ-Ject® does not sufficiently distribute the glyphosate around the stem, leaving strips of cambium alive that contribute to the higher incidence of sprouting, or that this method does not provide for a sufficient amount of active ingredient at the recommended application rate. Without testing the EZ-Ject® applicator with the other herbicides as well as with glyphosate, however, it is not possible to determine whether the glyphosate or the application technique resulted in reduced sprouting control.

Similarly to Burch and Zedaker (2003) and DiTomaso and Kyser (2007), our study found that triclopyr in a basal bark spray provided effective sprout control. Picloram/2,4-D also provided effective sprouting control in our study, when used both with the hack and squirt method and with the tree injector.

Several combinations of herbicide and delivery method will work to control existing stems of tree-of-heaven. As shown by the results of this study and others in the references above, glyphosate, triclopyr, and picloram alone or in combination with 2,4-D, and imazapyr, all provide good control of aboveground stems with various delivery methods. However, some considerations with using picloram or imazapyr should be taken into account before their use in controlling tree-of-heaven. Picloram is not registered for use in California or near water in any state, calling for another choice of herbicide in those situations (DiTomaso and Kyser 2007). Imazapyr, a commonly used chemical for hardwood control, has been found to have nontarget activity both when used as a basal bark spray and when directly injected into the target stem. One study found that using imazapyr in a basal bark spray left a dead zone of 3.3 ft. around the base of the target stem, and the authors mention that it is a broader-spectrum herbicide with longer residual activity and may not be a good choice in situations with nearby desirable stems (DiTomaso and Kyser 2007). Another study found that imazapyr, when directly injected into the target tree-of-heaven stem using the EZ-Ject® lance at the manufacturer's recommended rate, translocated and killed an average of 2.5 nontarget nearby stems, of both tree-of-heaven and other species, with damage seen up to 10 ft. away from the injected stem (Lewis and McCarthy 2008). Given the other choices, imazapyr should be used only in situations where the risk to other nearby stems is minimal, as with a pure stand of undesirable species (Lewis and McCarthy 2008).

CONCLUSIONS

The larger issue with tree-of-heaven is not control of existing stems, but sufficient suppression of sprouting so that the infestation is not made worse by an abundance of sprouts. Based on our study and the others reviewed here, picloram and imazapyr have excellent sprout control, and triclopyr provided excellent sprout control except when applied by the hack and squirt method by DiTomaso and Kyser (2007). However, there is a proliferation of evidence that glyphosate does not provide sufficient sprout control. In the first year of our study, the EZ-Ject® lance with glyphosate was the only treatment that was statistically worse than all other methods, though hack and squirt with glyphosate was not different from any other method. Meloche and Murphy (2006) also did not have success with control of sprouts using glyphosate in the EZ-Ject®, and DiTomaso and Kyser (2007) report that glyphosate did not sufficiently control sprouting in a cut stump treatment. Lewis (2007) did have success with the EZ-Ject® lance and glyphosate in controlling sprouts, though that study is somewhat of an anomaly, as no sprouting was reported after any treatment option (Lewis and McCarthy 2006, Lewis 2007).

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MAXIMIZING CARBON STORAGE IN THE APPALACHIANS: A METHOD FOR CONSIDERING THE RISK OF DISTURBANCE EVENTS

Michael R. Vanderberg, Kevin Boston, and John Bailey¹

Abstract.—Accounting for the probability of loss due to disturbance events can influence the prediction of carbon flux over a planning horizon, and can affect the determination of optimal silvicultural regimes to maximize terrestrial carbon storage. A preliminary model that includes forest disturbance-related carbon loss was developed to maximize expected values of carbon stocks. Stand-level optimization was used to develop silvicultural regimes that present a range of forest management benefits, while maximizing expected carbon stocks in the process. Potential benefits include: (1) a reduction in the risk of carbon loss due to disturbance events; (2) increased carbon storage; and (3) the offset of treatment costs through the utilization of treatment-generated woody material. This methodology may also provide a baseline for a full accounting of forestry carbon offset projects. The ability to increase terrestrial carbon density can be limited with the occurrence of disturbance events, as exhibited by some Appalachian forest types. However, silvicultural regimes maximized the expected value of in-forest carbon and carbon in wood products over the life of the analysis period in other Appalachian forest types. The results of this analysis suggest that treatments that effectively manipulate structure, age, and composition have the greatest potential for maximizing terrestrial carbon stocks by simultaneously considering both the risk of emission and storage potential.

INTRODUCTION

Forests are typically a sink for terrestrial carbon (Goodale and others 2002), and forest management has been suggested as a possible means of both reducing carbon dioxide emissions and enhancing carbon sinks (Birdsey and others 2000, Perez-Garcia and others 2005). Increasing the terrestrial density of carbon stored in both forests and wood products could offset up to 20 percent of U.S. emissions by the year 2025 (Chameides and Oppenheimer 2007), and entities such as the Chicago Climate Exchange and the California Climate Action Registry may act as carbon markets for the efficient reduction of atmospheric carbon inputs from forests (Ruddell and others 2007). Forest management objectives include reducing emissions from wildfire and increasing both the proportion and retention of carbon stored in wood products (Birdsey and others 2000).

Under no-management, a forest exhibits the potential for continued growth and subsequent carbon sequestration (Harmon and others 1990, Harmon 2001). The forest will continue accumulating carbon generated through photosynthesis until it is released back to the atmosphere through respiration or the decay of organic matter (Karjalainen and others 1999). Without a disturbance event, this scenario results in a forest that may act as a net carbon sink up to stand ages of 800 years (Luyssaert and others 2008).

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A no-management scenario in disturbance-prone forests of the Appalachian forest region includes the risk of fire, windthrow, ice damage, and pest outbreaks, as well as the carbon emissions associated with such disturbances. Biogenic emissions do not directly follow non-fire-related disturbances, although forests will eventually emit carbon from decay and decomposition. However, pyrogenic carbon emissions are immediate, depend on fire severity (Campbell and others 2007), and can also cause large biogenic emissions in years following the actual fire (Turner and others 2007). Sampson and Clark (1996) suggest that biogenic processes stimulated by tree mortality emit an average of five times the original pyrogenic carbon release over the next 50 years following a fire event. Stand-replacing fire with higher mortality would contribute even greater emissions.

Greenhouse gases, including carbon dioxide, are now considered to be a public health hazard. Mitigating climate change through the reduction of carbon dioxide is a national, as well as global, challenge. Forestry is poised to become a frontrunner in the pursuit of increasing carbon stocks at the terrestrial level. However, to maximize additional carbon storage, methods must be employed to account for uncertainty in terms of carbon losses to the atmosphere. The disturbance-prone forests of the Appalachian region present many opportunities for carbon loss, yet storage can be increased if optimal decision-making processes are employed.

This paper uses simplified examples to present a conceptual method for maximizing the terrestrial carbon storage potential associated with Appalachian forests. The focus is on the proof of concept (i.e., using strategic management regimes to capture potential carbon loss due to disturbance events), not the expected values associated with the example results.

METHODS

Eleven forest types were analyzed for their potential to store aboveground carbon, given the probability of fire and other disturbance events. Forest types were those used to classify the Southern Appalachian Potential Natural Vegetation Groups (PNVG) contained within the LANDFIRE project (Schmidt and others 2002). A summary of the Southern Appalachian PNVG can be found in Table 1.

To demonstrate the development of an optimal forest treatment regime to maximize carbon storage, dynamic programming was applied to solve problems containing two treatment alternatives (control and 15-percent basal area reduction) at each stage for hypothetical stands of each Southern Appalachian PNVG. A simple

Table 1.—Disturbance return intervals and mortality for southern Appalachian forests

Description	PNVG Code	mFRI	Mortality	mDRI	Mortality
Bottomland hardwood forest	R8FPFOpi	108	0.361	90	0.611
Hemlock-white pine-hardwood	R8HEWP	174	0.253	129	0.423
Mixed mesophytic hardwood	R8MMHW	78	0.214	59	0.334
Appalachian dry mesic oak forest	R8OACOm	13	0.177	11	0.237
Eastern dry-xeric oak	R8OAKxe	8	0.178	8	0.238
Appalachian oak hickory pine	R8OHPI	5	0.139	4	0.169
Oak-ash-woodland	R8OKAW	27	0.349	22	0.579
Appalachian shortleaf pine	R8PIECap	6	0.142	5	0.182
Appalachian Virginia pine	R8PIVlap	22	0.302	18	0.502
Southern Appalachian high-elevation forest	R8SAHE	312	0.690	231	0.950
Table mountain/pitch pine	R8TMPP	5	0.150	4	0.200

growth model was applied, where basal area was increased at a constant rate over three 30-year periods, regardless of prior treatment intensity or disturbance event. To simulate the effect of a silvicultural treatment on tree mortality, given the occurrence of fire or other disturbance, standing basal area reduction factors based on LANDFIRE Rapid Assessment Reference Condition Models were used for each PNVG (Schmidt and others 2002). A disturbance event within a treated stand incurred no reduction in the standing basal area, while a disturbance event within an untreated stand incurred mortality set to the corresponding reduction in standing basal area.

To track additional carbon storage over the decision pathway network (Figs. 1 and 2), standing tree basal area was converted to carbon at a conservative rate of 2.85 Mg per m² (Matthews 1993, Luyssaert and others 2008). A utilization rate was applied to treatment-generated woody material, and material was allocated to end-use classes with corresponding decay rates. Additional carbon values were calculated as the amount of expected carbon at the end of the regime on a per-Mg of carbon basis. Additional carbon was measured as any carbon storage gained by employing a silvicultural treatment regime over time when compared to the baseline. The baseline carbon value for each PNVG was measured from the decision pathway that underwent no treatment during the 90-year analysis period. Therefore, only decision pathways that included at least one treatment over the analysis period could store additional carbon.

PROBLEM FORMULATION

Decisions during the 90-year analysis period were made every 30 years over three major stages corresponding to growth periods, and two probabilistic stages corresponding to post-disturbance conditions, starting at year 0. At each major stage, two alternatives for each state were possible (no treatment, treatment). The objective function used in the problem formulation maximized the expected value of aboveground carbon stored in standing trees, woody debris, and wood products due to each decision. Major stage computations followed traditional dynamic programming recursive form (Bellman 1957), where maximum carbon storage for stage t is dependent on the maximum carbon storage for stage $t + 1$.

The element of disturbance within the analysis framework was implemented following the first and third stages. At these stages, the basal area stand descriptor variable was subjected to the decision of no disturbance or disturbance. If there was no disturbance, no disturbance-related mortality was modeled. Mean fire return interval and mean disturbance return interval (mDRI) were used as proxies in the determination of fire risk and total disturbance risk probabilities, respectively. Risk probabilities followed a binomial distribution, where the risk was equal to the probability of a single disturbance event occurrence within a stand over a 30-year period.

The values obtained during the stochastic stages represent the expected amount of terrestrial carbon after accounting for the probability of a disturbance event occurring under specified conditions and assumptions. Fire behavior and disturbance damage were expressed as a mortality rate in the alternative generation procedure. Data and assumptions used to determine the optimal treatment regimes are summarized in Table 2.

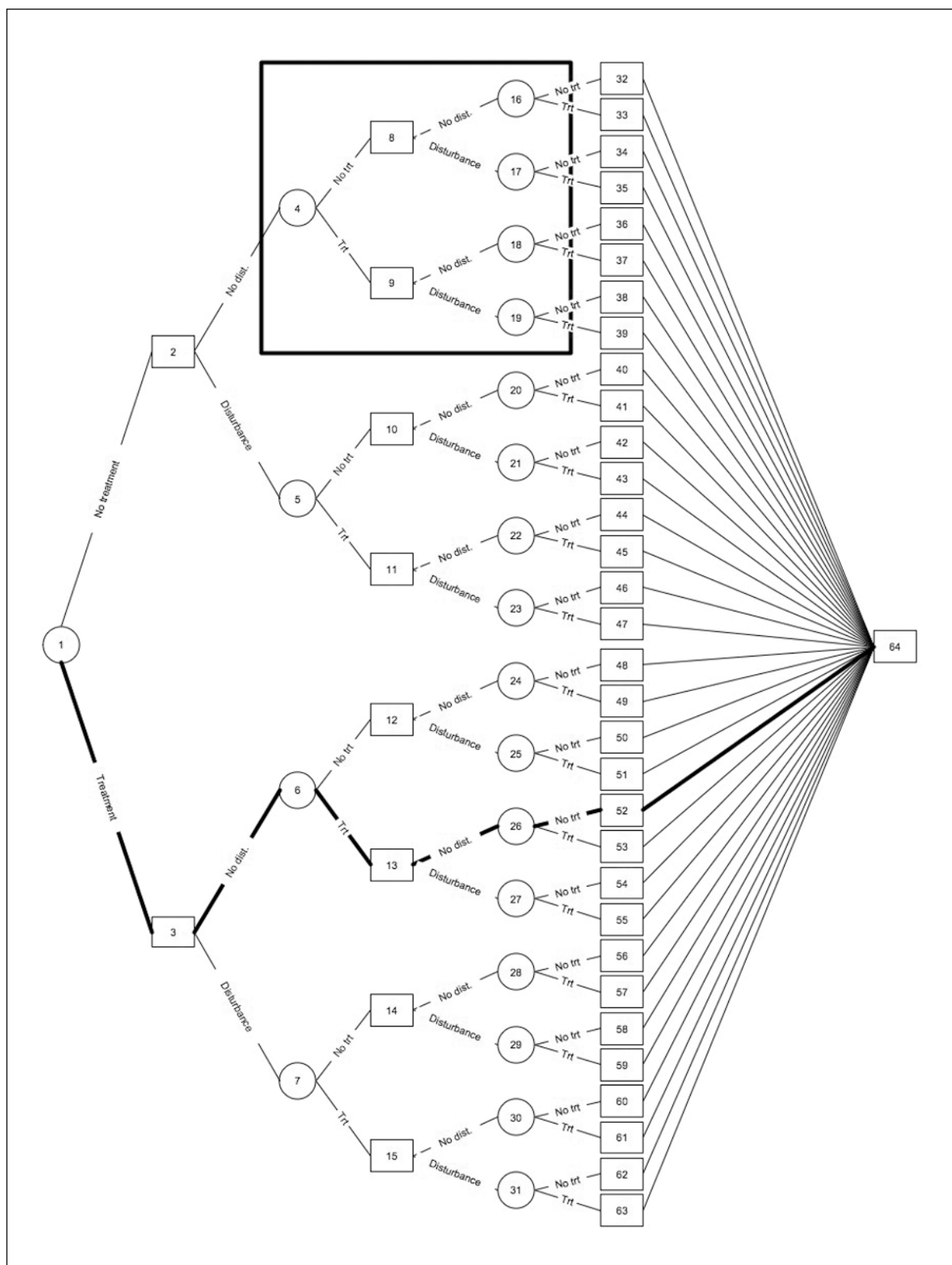


Figure 1.—Example of dynamic programming decision pathway network and optimal regime for PNVG R8OKAW: oak-ash-woodland (boxed area found in Fig. 2).

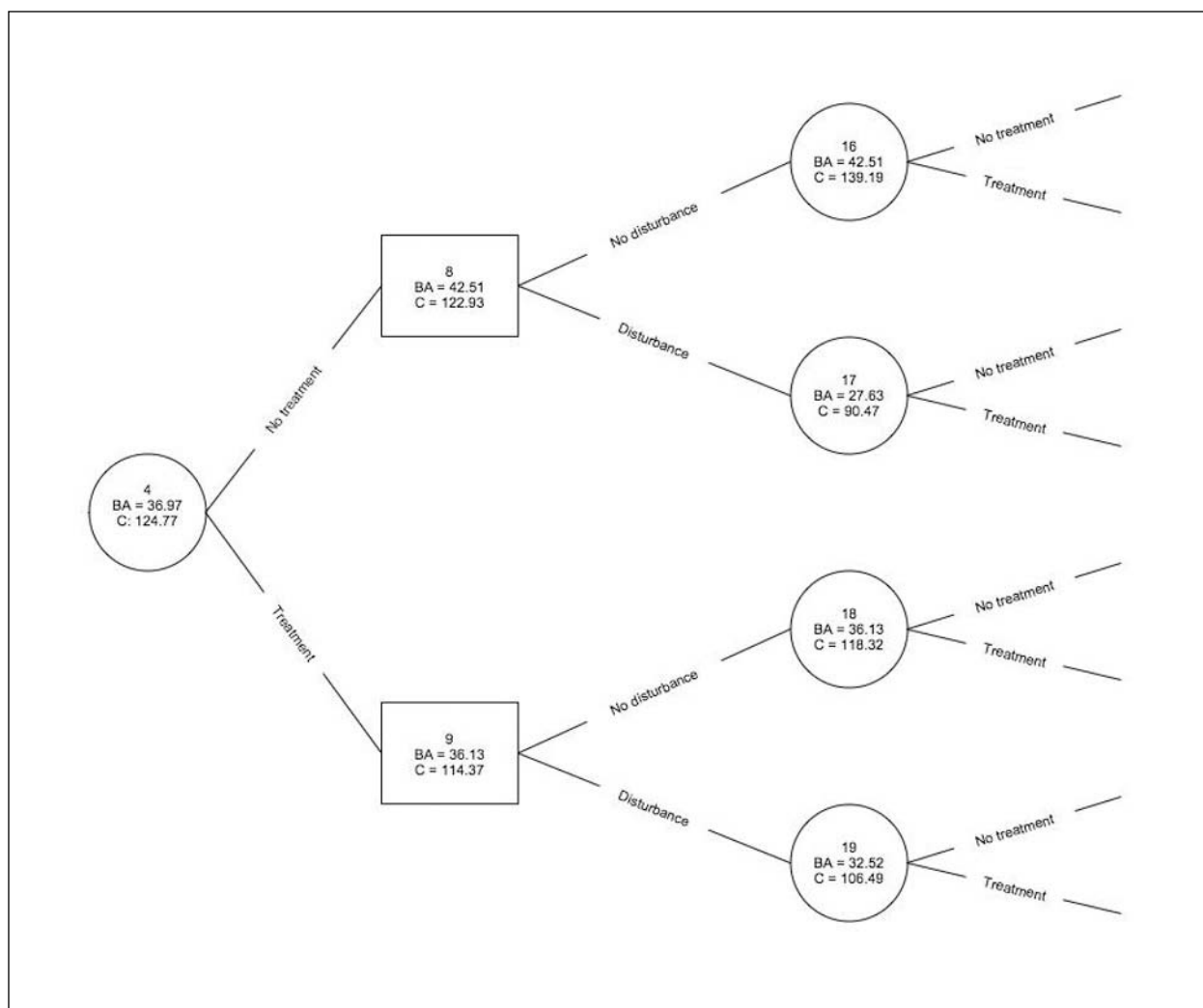


Figure 2.—Detail of decision network incorporating the probability of a disturbance event.

Table 2.—Summary of maximum carbon storage model parameters and assumptions

Parameter Description	Value
30-year ba/ha growth rate (Brooks et al. 2008)	1.25
Fraction of treatment ba/ha removal (Stephens and Moghaddas 2005)	0.15
Initial stand ba/ha (Brooks et al. 2008)	22.06 m ²
Woody material utilization rate (James et al. 2007)	0.85
Utilized woody material annual decay rate (Winjum et al. 1998)	0.01
Non-utilized woody material annual decay rate (Spies et al. 1988)	0.029
Fraction of utilized material subject to decay (James et al. 2007)	0.75
Fraction of utilized material permanently stored (Ximenes et al. 2008)	0.25
ba/ha to carbon volume conversion (Luyssaert et al. 2008, Matthews 1993)	2.85 Mg/m ²

RESULTS

The analysis and optimization procedure yielded silvicultural regimes to maximize carbon storage for each Southern Appalachian PNVG. Incorporating only the risk of fire, we determined that 9 of the 11 forest types showed the maximum carbon storage under the no-treatment scenario (Fig. 2). The two forest types that showed maximum carbon under treatment scenarios were R8OKAW (oak-ash-woodland) and R8PIVlap (Appalachian Virginia pine). Adding the risk of other disturbance to the risk of fire, we determined that an additional three forest types showed the maximum carbon storage under treatment scenarios. The three forest types were R8FPFOpi (bottomland hardwood forest), R8MMHW (mixed mesophytic hardwood), and R8SAHE (Southern Appalachian high-elevation forest).

DISCUSSION

Maximizing terrestrial carbon stocks from forests is a complex problem well suited for the use of dynamic programming for decisionmaking. The results of this analysis suggest that in some cases, treating the forest over time may be beneficial in reducing the risk of carbon emissions due to disturbance events. In turn, accounting for disturbance events can lead toward increased expected values of forest-based carbon stocks over the life of a planning horizon. Figure 3 shows the percent difference in maximum carbon storage over time between the best treatment-decision pathway and the control for each PNVG. A simple way to interpret

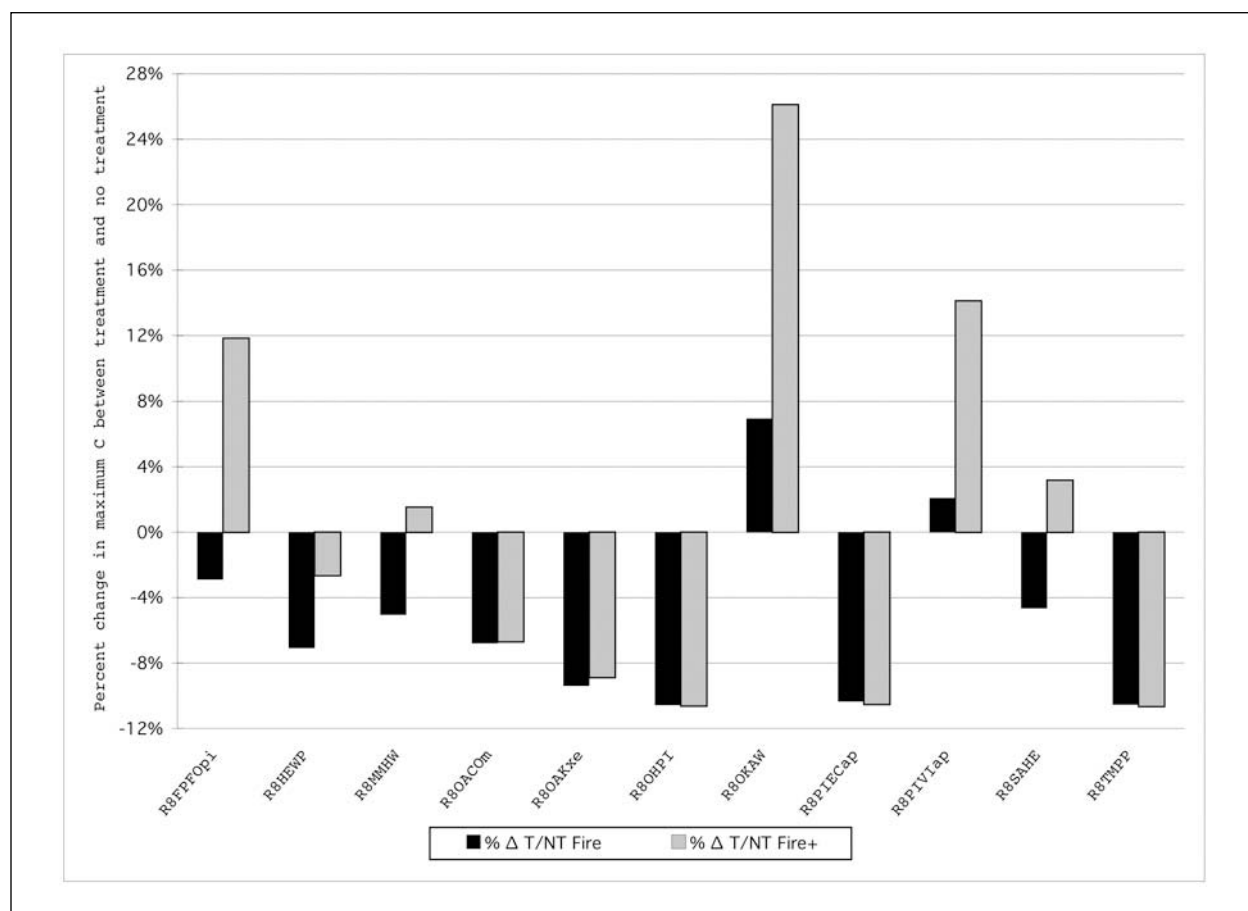


Figure 3.—Percent change in maximum carbon between treatment and control for fire and disturbance return intervals (refer to Results and Discussion for description).

Figure 3 is as follows: If the percent change is positive, expected carbon storage could be maximized by way of a treatment regime within the example provided in this paper; otherwise expected carbon stocks could be maximized without human interference.

Treatment regimes for the fire-risk only analysis maximized 2 of 11 forest types. Treating the oak-ash-woodland showed the potential for increasing carbon stocks by nearly 7 percent over the control, while treating the Appalachian Virginia pine increased stocks by just over 2 percent. Treatment regimes that incorporated the total disturbance risk maximized carbon storage for three more forest types while increasing the expected stocks by an additional 12 to 19 percent for the oak-ash-woodland and Appalachian Virginia pine stands. Of the three additional forest types, the bottomland hardwood forest increased carbon stocks by nearly 12 percent over the control. The Southern Appalachian high-elevation forest and mixed mesophytic hardwood forest types increased stocks by 1.5 to 3.0 percent over the control.

The best method for maximizing terrestrial-based forest carbon stocks depends on the type of forest being analyzed. Appalachian forests with a low mDRI (i.e., less than 20 years) were shown to store more carbon by way of no treatment over the analysis period. A logical explanation could be that the mortality rates associated with frequent disturbances are lower than or close to the treatment intensities. Similarly, Appalachian forests with a higher mDRI (i.e., greater than 20 years) were shown to store more expected carbon by way of a treatment regime, with the exception of R8HEWP (hemlock-white pine-hardwood). This result may be explained by the hemlock-white pine-hardwood forest type exhibiting the second highest mDRI, but also having the second lowest mortality rate of those forests with a mDRI greater than 20 years.

The greatest potential for increasing carbon storage by way of a silvicultural regime exists in bottomland hardwood forests, oak-ash-woodlands, and Appalachian Virginia pine stands. Oak-ash-woodlands have the most potential, showing an increase of more than 26 percent over the control. When the no-management regimes stored more carbon compared to treated stands, there was no difference greater than 11 percent.

There is also the opportunity to use increased carbon storage from forests as an investment or cost-offsetting measure. Assuming a value of \$5.83 per Mg carbon (Hamilton and others 2008), monetary returns ranging from approximately \$4 to upwards of \$57 per hectare may be possible, depending on initial forest and market conditions. Furthermore, if a silvicultural regime is expected to maximize forest-based carbon stocks, then there is also potential for revenue from the sale of treatment-based woody material if the markets exist.

Overall, terrestrial carbon stocks potentially can be increased by strategically managing the disturbance-prone Appalachian forests. As shown in this analysis, not all forest types stand to benefit from silvicultural treatment regimes in terms of carbon storage over time. However, some forest types may benefit. These forest types are prime candidates for further theoretical and empirical research to determine their potential for contribution to the low-carbon economy. Furthermore, added complexity, such as increasing the number of decision stages, expanding the silvicultural treatment options, and modeling the response of forest stands to both disturbance events and treatments, can lead towards increased carbon storage in other forest types.

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STATUS AND POTENTIAL OF TERRESTRIAL CARBON SEQUESTRATION IN WEST VIRGINIA

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Abstract.—Terrestrial ecosystem management offers cost-effective ways to enhance carbon (C) sequestration. This study utilized C stock and C sequestration in forest and agricultural lands, abandoned mine lands, and harvested wood products to estimate the net current annual C sequestration in West Virginia. Several management options within these components were simulated using a systems modeling approach to estimate C sequestration potentials. With such activities as substituting unused biomass resources for bioenergy production, converting marginal lands to forest covers, and extending the life of harvested wood products, atmospheric C sequestration by terrestrial ecosystems in West Virginia can be increased to 8.40 million t C/year from the current level of 5.50 million t C/year. These activities have the potential to provide additional benefits in terms of energy savings and CO₂ emission reduction at the source compared to other available carbon sequestration alternatives.

INTRODUCTION

Almost all of West Virginia consists of terrestrial ecosystems, and forest ecosystems alone cover more than 80 percent of terrestrial components (Fig. 1). Interest in atmospheric carbon (C) reduction through terrestrial ecosystem management is strong in West Virginia (as elsewhere) not only because of the prevalence of these ecosystems but also because of the cost-effectiveness of these components as a means of reducing atmospheric C compared to available alternatives. Managing these components requires knowledge about the limits of these ecosystem components' capacity to sequester atmospheric C and about the limiting factors of other components that directly affect such potentials. This paper describes the status and potential of C sequestration in terrestrial ecosystem components of forest lands, agricultural lands, mine lands, and harvested wood products (HWP) through different enhancement strategies along with limitations. Additionally, the paper compares the terrestrial ecosystem-based C sequestration options with other options on the basis of cost, feasibility, and efficiency of emission reduction in West Virginia.

METHODS

Estimates of forest C stock were obtained using Carbon On Line Estimator (COLE) available at <http://ncasi.uml.edu/COLE/>. Information on available mine lands, boundaries, and status was obtained from the West Virginia Department of Environmental Protection (2008). Information on C in harvested wood and applicable end uses was obtained from Timber Products Output data, available at <http://nrs.fs.fed.us/fia/topics/tpo/>. Energy production and C emission reduction through replacement of coal fuel with biomass for energy production were obtained from available conversion factors applicable for West Virginia (USDA, Forest Products Laboratory 2004, Wang and others 2007, Plant Fossils of West Virginia 2009).

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Carbon content in wood biomass was estimated as half of the total biomass (Eq. 1). The biomass content removed from the forest was obtained by using a volume-density relationship (USDA, Forest Products Laboratory 1953, Grantham and Ellis 1974, Smith 1991) for the different species (Eq. 2). Between 2000 and 2006, the ratio of net growth to removal (Eq. 3) in West Virginia forests was approximately 1.5 in terms of volume of growing stock (Sharma and others 2009, USDA, Forest Service 2009). This ratio implies that 40 percent of current growth is harvested and the remaining 60 percent remains standing. This ratio was even higher in previous years: 2.1 in 1994 and 2.0 in 1979 (Widmann and others 1998). This growth-to-removal ratio means that there will be two distinct pools of forest. One pool has a higher mean age and a

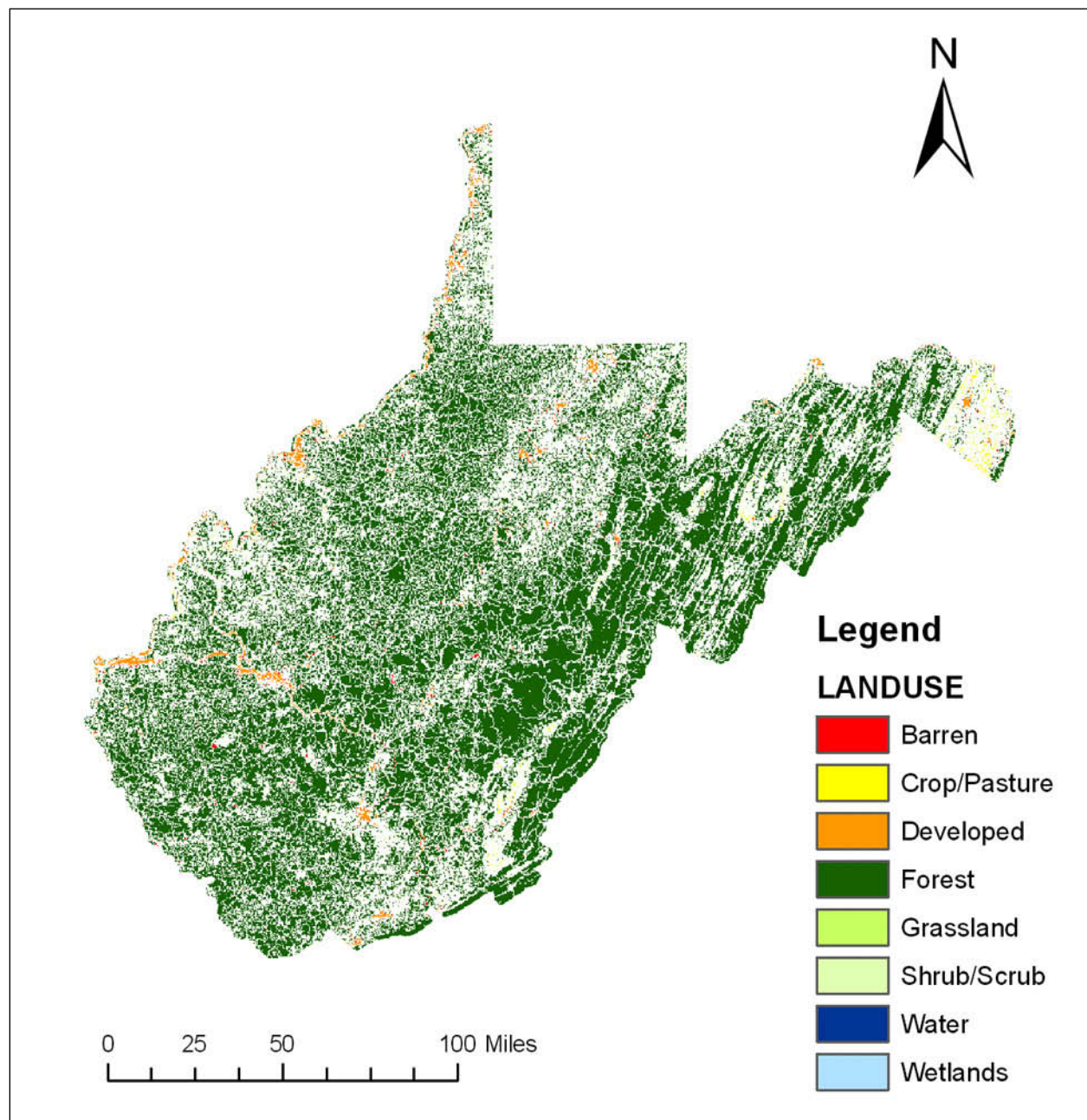


Figure 1.—Major land use in West Virginia (Data source: USGS NLCD 2001).

lower sequestration rate than the other pool. The mean C sequestration rate for the entire forest was obtained by proportionally applying two growth rates using Eq. (4) (Sharma 2010). These two growth rates were applied to capture the variations in growth rate inherent in the nonlinear growth characteristics of forests. A forest sequesters C at different rates along its life; initially, the rate is low, followed by rapid growth in the middle and finally a slower growth rate in old stands (Sharma 2010). Removal of C occurs when forests are harvested and this stock is estimated using Eq. (5) (derived from Eq. [3]). The harvested products emit C to the atmosphere through decomposition and decay and often go through several production stages during their lives. Depending on product use, these products have variable product life and a variable amount of C stored at different times. Amount of C remaining in a product was estimated from an exponential decay function with the half-life of C in the particular product type (Table 1) using Eq. (6). When harvesting occurs each year, the resulting annual value of C in harvested products can be estimated by using Eq. (7). Eq. (5) remains the parent equation for Eq. (6) and (7). C sequestered in agricultural lands and mine lands was obtained by using area and C sequestration rate. Total terrestrial sequestration was obtained by combining C sequestered in forests, agricultural lands, mine lands, and harvested wood product pools (Eq. 8).

$$B_c = 0.5 B \quad (1)$$

$$B_s = V_{B_i} D_{B_i} \text{ for } i = 1 \dots \text{species group} \quad (2)$$

$$GRR = \frac{ACA}{ACR} \quad (3)$$

$$ACA = A \frac{1}{GRR + 1} (GRR * r_1 + r_2) \quad (4)$$

$$AGR = \frac{ACA}{GRR} \quad (5)$$

$$Q_y = \sum_{n=1}^N Q_n * [1 - \frac{1:n(2)}{h_n}]^{(y-1)} \quad (6)$$

$$Q_y = \sum_{n=1}^N Q_n * [1 - \frac{1:n(2)}{h_n}]^{(y-1)} + c \sum_{i=2}^y \Delta Q_n [1 - \frac{1:n(2)}{h_n}]^{(t-2)} \quad (7)$$

$$Tc = ACA + ACR + (A_{ag} - A_{mag}) * r_{ag} + A_{mag} * r_{agf} + r_{mlf} * AML \quad (8)$$

where B_c is carbon in removed forest biomass, B_s is biomass content in tons for species group s , V_{B_i} is volume of biomass removed in cubic feet from species group i , D_{B_i} is density of biomass removed from species group i , GRR is growth-to-removal ratio (> 0), ACA is annual C accrual (in million tons) in forest ecosystems, A is

Table 1.—Wood product half-life (years).

SN	Category	Half-life, years*	Applicable products
1	Long-term product	30	Saw logs; composite products
2	Medium-term product	15	Posts, poles, and pilings; veneer logs; fiber
3	Short-term product	1	Fuelwood; misc. products; pulpwood
4	Landfills	145	Discarded residues
5	Dump and let decay	5	Unused products

*Schelhaas and others 2004

total area of forests (in million acres), r_1 is C sequestration rate from forest growth (t C/acre) in old and over-mature stands, r_2 is C sequestration rate (t C/acre) in young to mature stands, ACR is annual carbon removed (million tons) from the forest ecosystem, Q_{t+1} is quantity of carbon at time y , Q_n is the initial quantity of carbon in the n^{th} product, Y is number of years, y is any future time period between 1 and Y , N is number of different products from forest harvest, n is any product between 1 and ∞ , h_n is expected half-life of product n , t is time period (year), ΔQ_n is the quantity of addition to previous stock in the n^{th} category, $i = 1 \dots y$ where $y \geq 2$, $c = 0$ if $y < 2$ and 1 otherwise, Tc is total terrestrial carbon, A_{ag} is area of agricultural lands, A_{mag} is area of marginal agricultural lands, r_{ag} is C sequestration rate in agricultural lands, r_{agf} is C sequestration rate in afforested marginal agricultural lands, A_{mi} is area of mine lands, and r_{ami} is C sequestration rate by afforestation in mine lands. Parameters used in the model were obtained from several sources and are listed in Table 2.

RESULTS

CARBON STOCK AND ANNUAL SEQUESTRATION

Twelve million acres of forest land in West Virginia store approximately 1,035 million tons of carbon in different ecosystem components; more than half of the total C is stored in standing live trees (Fig. 2). The state's hardwood forests store a majority of the total C (Table 3). These forests sequester C at the rate of 0.32 t C/acre (range 0.18-0.60 t C/acre) in harvested stands and at the rate of 0.21 t C/acre (range 0.05-0.29 t C/acre) in mature stands. The net annual C sequestration by these forests thus is about 3.9 million tons. These estimates represent the net accumulation of C after deduction due to mortality and emissions and are based on 200-year projections.

Table 2.—Definition of model parameters.

Parameter	Definition	Source
A	Area of forest land (million acres)	USGS NLCD (Homer and others 2004)
r_1	Forest C sequestration rate in old stands (t C/acre/year)	Sharma and others 2009, FVS model
r_2	Forest C sequestration rate in young stand (t C/acre/year)	Sharma and others 2009, FVS model
GRR	Growth-to-removal ratio	USDA-FS 2009
A_{ag}	Area of agricultural land (million acres)	USGS NLCD (Homer and others 2004), Niu and Duiker 2006
A_{mag}	Area of marginal agricultural lands (million acres)	Niu and Duiker 2006
r_{ag}	C sequestration rate in agricultural lands (t C/acre/year)	Sperow and others 2003
r_{agf}	Carbon sequestration rate in agricultural lands with afforestation (t C/acre/year)	Sperow and others 2003, Niu and Duiker 2006
AML	Area of abandoned mine lands (million acre)	WVDEP 2008
r_{am}	C sequestration in abandoned mine lands (t C/acre/year)	Sperow 2006
r_{mif}	C sequestration in abandoned mine lands with afforestation (t C/acre/year)	Sperow 2006
V_{Bi}	Volume of biomass (m^3) in species i	TPO data,
D_{Bi}	Density of biomass (t/m^3) in species i	USDA-FPL 1953, Grantham and Eillis 1974, Smith 1991

Source: Sharma 2010

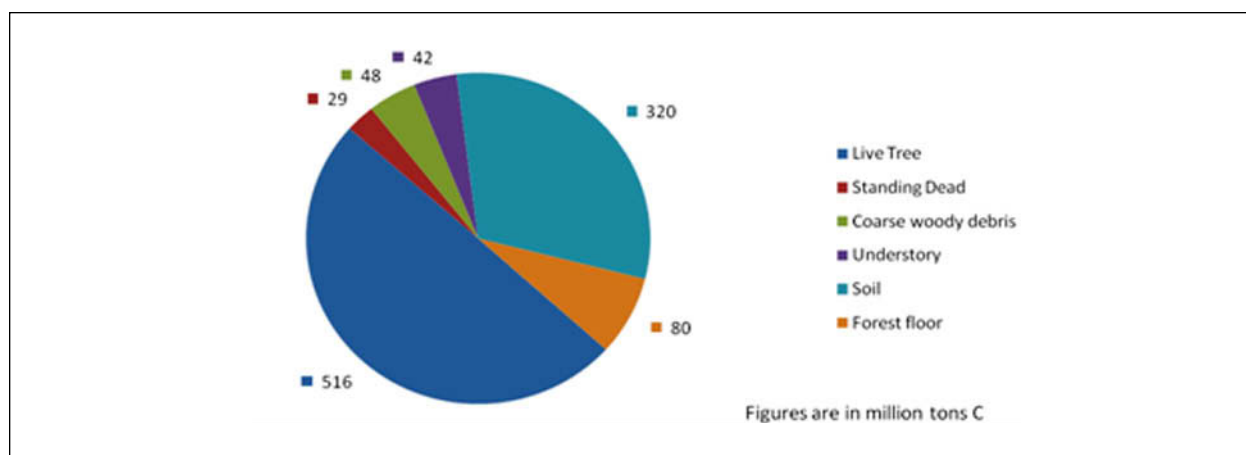


Figure 2.—Carbon storage in West Virginia forest ecosystem components (in million tons).

Table 3.—Amount of carbon (million t) stored in forests in West Virginia.

Forest type/group	t C/acre	Area (million acres)	Total (million t C)
Aspen/Birch Group	41.7	0.04	2
Elm/Ash/Cottonwood Group	108.36	0.04	4
Loblolly/Shortleaf Pine Group	72.88	0.04	3
Maple/Beech/Birch Group	95.91	3.91	375
Oak/Hickory Group	78.69	7.63	601
Oak/Pine Group	89.29	0.35	31
Spruce/Fir Group	88.31	0.02	2
White/Red/Jack Pine Group	89.05	0.08	7
All	85.23	12.12	1,035

Source: Carbon On Line Estimator (COLE)

Forest harvest in the state has remained constant at the rate of about 0.25 billion cubic feet per year. These removals are transformed into different wood products and do not include non-growing stock removals from thinning and silvicultural operations. HWP contain C sequestered in woody biomass from the atmosphere through the process of photosynthesis. Atmospheric C can be lowered by increasing C stock in HWP (Dixon and others 1994). The net reduction, however, depends on the final end uses and life of the wood products. Of the total biomass removed from the forests, approximately one-third remains in the forest as logging residue, almost one-fifth ends up as mill residue, and slightly more than half of the total growing stock ends up in final wood products (Table 4).

The harvested wood products including logging residues contain 2.5 million t C. These removals are not considered “loss” if they can be put to uses other than burning. The net carbon storage over a 20-year period (assuming an equal amount of harvest every year, i.e., annual removal of 1.37 million t C from the forest) when put to use as in Figure 3d, would make a total of 50.12 million t C stocked in wood products. Decay and several short-term uses, however, diminishes this stock to 24.94 million t C over the next 20 years. If the forest can sustainably grow this amount and the use pattern remains the same for a defined time period, almost half of the carbon stored in the harvested volume is emitted back to the atmosphere, and the remaining half (i.e., 1.25 million tons annually) of the harvested volume remains stored in harvested wood products.

Table 4.—Annual estimate of volume of wood products.

	Round wood (cft)	Mill residue (cft)	Logging residue (cft)	Total (cft)	Percent
Composite products	11,771,990			11,771,990	3.52
Fuelwood	4,131,773	20,562,169		24,693,942	7.39
Posts, Poles, and Pilings	3,776,820			3,776,820	1.13
Pulpwood	21,734,742			21,734,742	6.5
Sawlogs	117,518,971			117,518,971	35.17
Veneer logs	7,849,775			7,849,775	2.35
Fiber		19,899,719		19,899,719	5.96
Misc. products	468,992	12,511,635		12,980,628	3.88
Unused		8,684,602	105,250,164	113,934,766	34.1
Total	167,253,064	61,658,125	105,250,164	334,161,353	100
Percent	50.05	18.45	31.5		

Data source: USDA Timber Products Output data (1997, 2002, 2007)

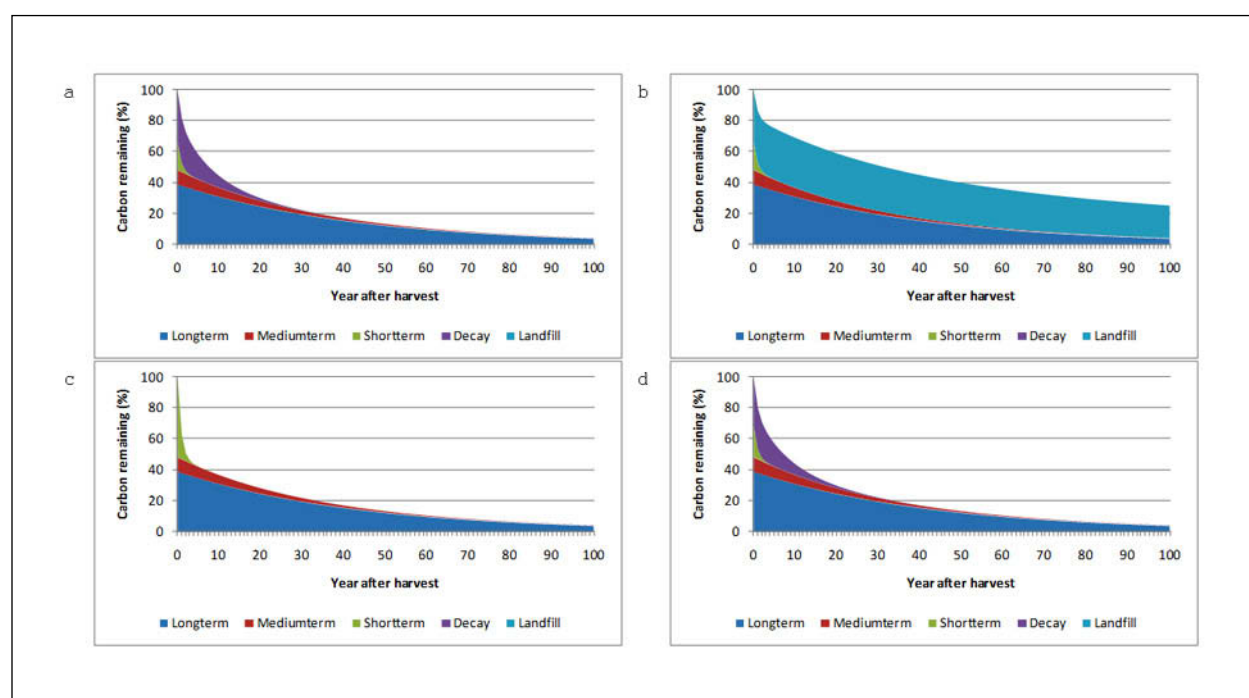


Figure 3.—Effect of different wood product use on carbon storage. (a) Unused residues are left to decay; (b) Unused residues are disposed in landfills; (c) Unused residues are used as biofuel; (d) Unused mill residues are used as biofuel and logging residues are left to decay.

Two million acres of crop lands in West Virginia sequester about 0.05-0.3 t C per acre per year in soil components, resulting in a total C sequestration of 0.1-0.6 (mean 0.35) million tons annually (Sperow and others 2003). In these lands, the growth in the aboveground biomass portion is consumed and/or decomposed within a year and only C in the soil layer lasts for more than a year. More than 452,000 acres of surface mine lands are devoid of biomass cover. As of 2008, about 29 percent of the total active mine lands are being reclaimed and their sequestration is negligible.

CARBON SEQUESTRATION ENHANCEMENT POTENTIAL

Approximately 5.5 million tons of C is annually sequestered in terrestrial components in West Virginia. Forest growth alone provides more than 70 percent of the total annual terrestrial sequestration, followed by sequestration in harvested wood products. Approximately 40 percent of agricultural lands in West Virginia are reported to be marginal land, i.e., agricultural land with severe limitation due to soil, slope, erodibility, and other environmental factors that make the primary agricultural activity less profitable. Afforestation of marginal agricultural lands has the potential for sequestration of 1.37-1.75 t C per acre per year or 1.6-2.06 million t C per year (Niu and Duiker 2006). If this activity is carried out, mean annual C sequestration in agricultural land becomes 1.74 million tons.

The forest C stock can be increased by bringing back the harvested stands to forest use and managing them sustainably. When the growth-to-removal ratio is brought closer to 1 by increasing harvests and re-growing forests in harvested regions, it increases the available growing space and the proportion of young stands. These younger trees sequester more C per unit of time and per unit area, resulting in additional sequestration of 0.94 million t C in the forest ecosystem. Increased harvesting will also contribute 83.4 million cft of wood to the existing estimate of HWP (assuming a constant annual removal rate). With these additional removals, approximately 28.35 million cft of unused wood will be available for potential biomass fuel. The HWP will then sequester an additional 0.29 million t C annually for the entire state.

Reclamation of abandoned mine lands through afforestation would result in an increase in forest area and annual accrual of 0.56 million t C. This estimate uses a C growth rate of 2.3 t C per acre in bare soil, which was obtained from COLE model. Thus, West Virginia could increase the current C sequestration rate by up to 52 percent to achieve a net annual sequestration of 8.40 million tons. These potentials can be achieved by changing land use activities in the marginal agricultural and abandoned mine sites, enhancing forest growth, and increasing the output of sustainable wood products (Table 5).

ALTERNATIVES TO TERRESTRIAL C SEQUESTRATION

Carbon dioxide capture and storage (CCS) is the use of technology to collect and concentrate the CO₂ produced by industrial and energy-related sources, transport it to a suitable storage location, and then store it away from the atmosphere for an extended period of time. Though considered a technologically viable option for reducing atmospheric C significantly, CCS is still an expensive measure. The cost of managing C through CCS ranges from \$60 to \$300 per ton (International Panel on Climate Change 2005, Friedmann and others

Table 5.—Status and potential of terrestrial carbon sequestration in West Virginia.

Component	Annual Sequestration (million t C)	
	Current	Potential
Forest land	3.90	5.00
Agricultural land	0.35	0.14
Harvested wood products	1.25	1.55
Abandoned mine land afforestation	-	0.11
Marginal agricultural land afforestation	-	1.60
Total Sequestration	5.50	8.40
Percent change	100.00	152.72

2006). On the other hand, the cost of carbon sequestration from forest activities ranges from \$10 to \$150 per ton; the median range lies between \$14 and \$41/t C (Richards and Stokes 2004). Although cheap, forestry based C sequestration has an upper limit on total sequestration potential. At present, up to 27 percent more C can be sequestered by growing forests trees of suitable species in marginal lands. The avoided C emissions associated with forest-based fossil fuel alternatives, i.e., fuel wood systems, represent a significant amount of C emission savings. Current uses of wood products account for 28,458 tons of C emission annually and supply 455 million British thermal units (BTUs). These wood resources would eventually decay and emit C to the atmosphere but when used as fuel, can help reduce emissions. The energy produced can be considered as savings as this amount of energy would have had to be obtained by burning fossil fuels. BTU savings can be increased further by using unused logging residues to produce energy, which could provide an additional 1.3 billion BTUs and save emissions of 85,000 t C. The economic savings of woody biomass use in power generation can be further increased when we take into account the expense of using geologic sequestration to dispose of the carbon produced during power generation.

DISCUSSION AND CONCLUSIONS

Sustainability is key to West Virginia forests' ability to function continuously as C sinks, which can be achieved by removal without exceeding net growth in a particular year and use of the removed products for long-term purposes or as replacements for fossil fuels. Sustainable harvests have the potential to increase supply by 83.4 million cft of wood, which will contribute to 28.35 million cft of residues for possible biomass-based energy and can prevent approximately 84,451 tons of carbon from being emitted from fossil fuels. Increasing biomass use in power plants can boost demand for timber products, creating an incentive to grow more timber, which will result in more sequestration along with C emission reduction from fossil fuels. Over the long run, the carbon neutrality of biomass use will help achieve the objective of reducing C emissions. When such opportunities exist, the unused woody biomass would start gaining some value, which is likely to motivate land owners to convert surface mine lands and marginal agricultural lands into forestland use. This option is likely to be lucrative if benefits of emerging carbon markets can be made available to land owners who wish to convert the land use to forestry.

Preventing decay of these wood products can help in offsetting C emissions. Short-term wood products, such as pulp and paper, would be considered as lasting for 1 year or less. On the other hand, long-lasting wood products, such as structural timber and furniture, retain stored carbon for an extended period of time and go through multiple instances of recycling. Recycling, avoiding combustion to dispose of wood, and utilizing available residues to replace fossil fuel for power generation would enhance C sequestration. Similarly, proper landfills and wood burials can be helpful in enhancing C sequestration. When available, unused woody residues are used to substitute for some of the coal in power plants, the corresponding amount of C can be saved. Using biomass for energy production combined with CCS at the source will make C emissions negative: C in the atmosphere will be permanently reduced by substituting carbon-neutral, sustainably grown biomass for fossil fuels. The future direction of the current study is towards studying the limits and costs of implementing the strategies identified for C sequestration enhancements in West Virginia.

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ARE NITRATE EXPORTS IN STREAM WATER LINKED TO NITROGEN FLUXES IN DECOMPOSING FOLIAR LITTER?

Kathryn B. Piatek and Mary Beth Adams¹

Abstract.—The central hardwood forest receives some of the highest rates of atmospheric nitrogen (N) deposition, which results in nitrate leaching to surface waters. Immobilization of N in foliar litter during litter decomposition represents a potential mechanism for temporal retention of atmospherically deposited N in forest ecosystems. When litter N dynamics switch to the N-release phase, litter N may contribute to N exports. We tested the hypothesis that nitrate exports in stream water are positively related to the N dynamics in foliar litter, with generally low exports during N immobilization and generally high exports during N mineralization in litter. We performed a simple regression analysis to test this hypothesis. We used litter N flux and nitrate export data for 2002-2006 from two watersheds at the Fernow Experimental Forest in West Virginia, one receiving ambient and one receiving experimentally increased rates of atmospheric deposition. Linear regression of monthly stream nitrate export values with litter N flux revealed no relationship. We conclude that while foliar litter immobilizes N, and the amount of immobilized N is of similar magnitude as that arriving with deposition, this N-retention mechanism does not translate into lowered nitrate exports in stream. Further, nitrate exports do not increase during the litter N mineralization phase.

INTRODUCTION

Many of the hardwood forests in the eastern United States leach nitrate to surface waters (Stoddard 1994), with potentially far-reaching ecological and economic consequences. During leaching, nitrate combines with base cations such as calcium and magnesium. A prolonged export of both of these elements may lead to soil acidification, a decrease in forest productivity, decline in forest health, and a reduction in the capacity of forest ecosystems to sequester carbon (C) (Adams and others 2000). Additionally, non-point source pollution by nitrate leads to water acidification, and represents a potential health hazard to human and aquatic life.

Nitrate exports from forests have been strongly linked to atmospheric N deposition, land-use history, and the rate and duration of N loading (Aber and others 1989, 1998; Stoddard 1994; Goodale and others 2003; Pregitzer and others 2004; Castro and others 2007). The Central Hardwood Forest receives some of the highest rates of atmospheric N deposition in the eastern United States, reaching 11 kg N ha⁻¹ yr⁻¹ in parts of West Virginia (Adams and others 2000). Mass balance estimates show that most of the atmospheric N is retained in forests (Aber and others 1998, Goodale and others 2009). Studies utilizing isotopic techniques have demonstrated that nitrate in drainage waters is primarily produced by microbial mineralization and nitrification of organic N, which strongly suggests that atmospherically deposited N is cycled (McHale and others 2002, Piatek and others 2005, Campbell and others 2006, Hales and others 2007).

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Mechanisms of atmospheric N retention in impacted forests have been a focus of much research in the last couple of decades (Aber and others 1998). N retention processes remain elusive, however (Magill and others 2000). The hypothesis that abiotic complexation in soils occurred via a “ferrous wheel” mechanism had some merit (Davidson and others 2003) but was unsupported in a large study of different forest types (Coleman and others 2007). Immobilization in microbial biomass, increases in net primary productivity, and mycorrhizal uptake remain among the strongest candidates for likely mechanisms of N retention (Aber and others 1998).

Immobilization of N in the biomass of microbes colonizing fresh foliar litter during litter decomposition represents a potential mechanism for short-term N retention of atmospherically deposited N in forest ecosystems. Immobilization is a microbial process of sequestration of elements limiting microbial reproduction and growth when energy sources in the form of easily decomposable C are abundant (Gallardo and Schlesinger 1994). However, assessments of the capacity of microbial biomass to retain atmospheric N are conflicting, the conflict likely arising from the timing of N-label application and different ways of treating the forest floor (Zogg and others 2000). Namely, N dynamics in fresh hardwood foliar litter exhibit distinct phases. The highest rates of immobilization start shortly after leaf fall in October and last 4-6 months; mineralization begins shortly after peak immobilization (Adams and Angradi 1996, Piatek and others 2009a). Therefore, a summer application of N label when mineralization in fresh foliar litter begins will in effect nullify the microbial N sink activity (Fig. 1). Inclusion of the partly-decomposed forest floor together with fresh litter lowers the effective N sink because older litter exhibits net mineralization rather than immobilization. However, treating fresh litter apart from the old litter layers as well as applying N tracer in the fall is expected to yield a much stronger N immobilization, or N sink activity.

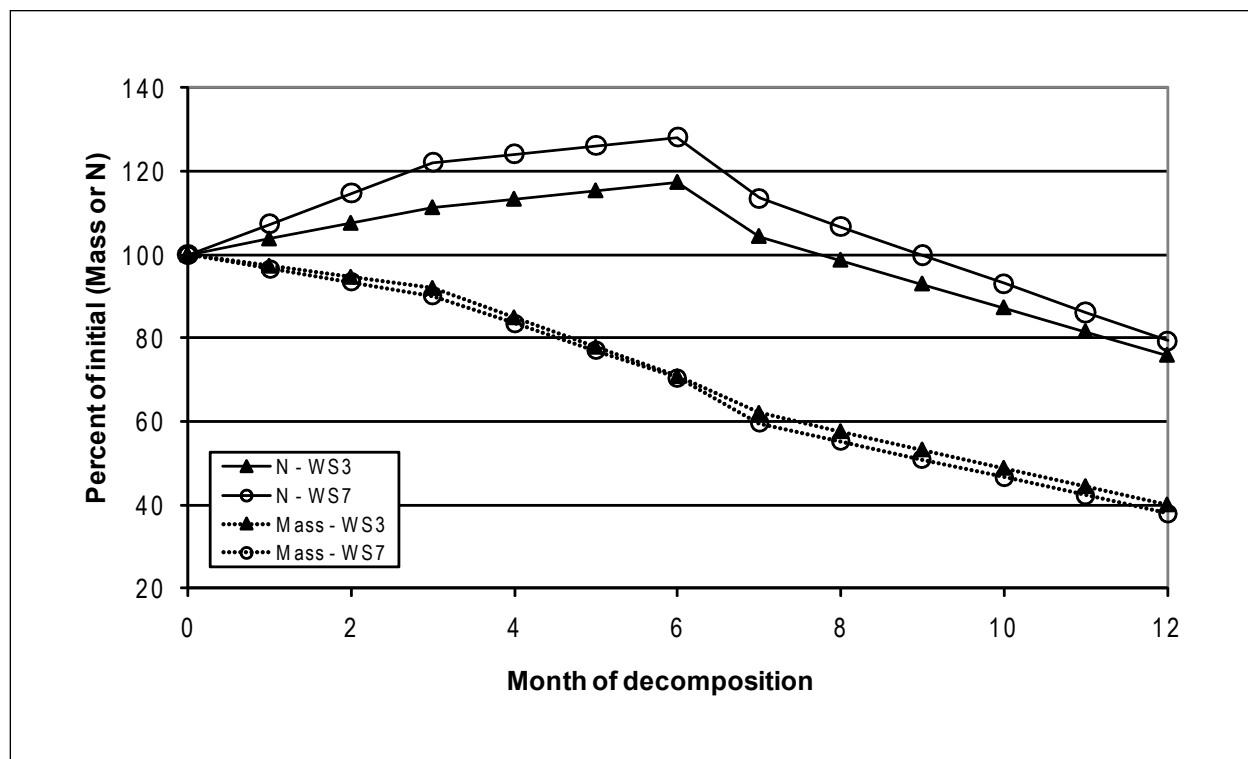


Figure 1.—Foliar litter N and mass loss dynamics (in percent of initial amounts) during 12 months of decomposition starting with litterfall in October (month 0) for two watersheds – reference (WS7) and high deposition (WS3). Data adapted from Piatek and others (2009a). “0” is October and “12” is October of the following year.

Several other lines of reasoning support the role of microbial N immobilization in foliar litter in N retention. First, rates of N immobilization in deciduous foliar litter are often of the same magnitude as rates of N deposition (Magill and Aber 1998, Piatek and others 2009a). Second, fresh foliar litter is the first point of interception of throughfall N with the soil. Third, immobilization in litter, and subsequent litter N mineralization and nitrification, yields a microbial isotopic signature of nitrate, such as is commonly found in drainage waters (McHale and others 2002, Piatek and others 2005, Campbell and others 2006, Hales and others 2007).

Recent work at the Fernow Experimental Forest in West Virginia showed that foliar litter composed of about 50 percent oak, 20 percent yellow-poplar, 15 percent maple, and smaller quantities of magnolia, pin and black cherry, and birch immobilizes at least 30 percent more N than initially present in fresh litterfall (Piatek and others 2009a). Depending on the total mass of litter, these litter N dynamics result in immobilization of 6 to 11 kg N ha⁻¹ yr⁻¹. For N deposition levels of 11 kg ha⁻¹ yr⁻¹, the above rates of N immobilization represent a potential retention of 55 to 100 percent of the amount of N in throughfall (Adams and others 2000). Lower rates of immobilization are common under conditions of increased rates of N deposition, while higher rates of immobilization are possible in stands with a higher oak component (Templer and others 2005; Piatek and others 2009a, 2010). When leaf litter begins to mineralize around April, plant N uptake can assume the primary role in N retention. The strength of these retention mechanisms is likely to decrease and then possibly subside when plant uptake diminishes and a new litter layer (Oi) is yet to form at the end of the summer into early fall.

We hypothesized that a positive relationship exists between N flux in decomposing foliar litter and stream nitrate dynamics. According to this hypothesis, lower nitrate exports should generally coincide with the N immobilization phase and higher nitrate exports should generally coincide with the mineralization phase. The objective of this study then was to test this hypothesis and to investigate a potential link between N dynamics in foliar litter and nitrate exports from forests. For this investigation we conducted a simple regression analysis of monthly litter N dynamics on stream nitrate exports for two watersheds at the Fernow between 2002 and 2006.

METHODS

LEAF LITTER NITROGEN DYNAMICS

Nitrogen dynamics (rates of immobilization and mineralization) in fresh litter were adapted from a study by Piatek and others (2009a) in a central hardwood forest at the Fernow Experimental Forest in Parsons, WV. Briefly, known amounts of fresh litter collected in the long-term soil productivity plots at the Fernow were allowed to decompose in nylon mesh bags for 12 months in 2006-2007. Litter was collected from, and later allowed to decompose in, reference (untreated) plots and plots treated with annual additions of 36 kg N and 40 kg sulfur (S) ha⁻¹ as ammonium sulfate (increased rates of deposition). Species composition and specific weight proportions reflecting the actual litter mix were (in percent of dry weight): oak (48), yellow poplar (19), maple (14), magnolia (12), cherry (4), and sweet birch (3). Mass loss and N contents were quantified at 0, 3, 6, 7, and 12 months after the start of the experiment. Monthly data were obtained by linear interpolation between data points (Fig. 1).

Mass loss and litter N dynamics from the above plot-level study (Fig. 1) were applied to litterfall mass for two watersheds at the Fernow for 2002-2006 (Table 1). Watershed WS3 receives annual additions of 36 kg N ha⁻¹ yr⁻¹ and 40 kg S ha⁻¹ yr⁻¹ in an effort to accelerate soil acidification and N leaching (Adams and others 2006). Watershed WS7 is a reference watershed. Both watersheds are approximately 40-year-old central hardwood forests. Nitrogen dynamics in litter differ between the reference and the treated stands in that immobilization levels are significantly higher in litter decomposing in the reference than in the treated stands (Fig. 1; Piatek and others 2009a). Therefore, treatment-specific N dynamics were applied to the correspondingly treated watersheds.

To calculate litter N dynamics, first, annual litterfall mass data for WS3 and WS7 were obtained for the watersheds from annual collections in litter traps. Detailed collection methods are provided in Adams (2008). Mass loss of this litter during a 12-month decomposition period was calculated as the product of initial litterfall mass and percent mass remaining from Figure 1 (Table 1).

Second, monthly N content in these litters was calculated separately for each watershed as the product of initial litter mass (watershed-level data) and initial litter N concentration and percent of initial N content remaining (plot-level data). Nitrogen immobilization was indicated by increases over initial litter N content (concentration times mass). Nitrogen mineralization started when litter N content decreased from maximum immobilization; net N mineralization was the decrease in N content below initial litter N (initial equals 100 percent) (Fig. 1, Table 2).

Third, N fluxes in litter were calculated as the difference in N contents between months (Table 3). Nitrogen fluxes were used in the regression in Figures 2 and 3. For the regression analysis, N flux during immobilization was treated as a loss (negative) because immobilization in effect temporarily removes N from the system. Nitrogen flux during mineralization was treated as an addition (positive) because mineralization contributes N to the system.

STREAM NITRATE EXPORTS

Volume-weighted monthly nitrate-N exports (kg N ha⁻¹ mo⁻¹) and nitrate-N concentrations (mg L⁻¹ mo⁻¹) in stream water were obtained for 2002 through 2006 for both watersheds. Stream chemistry has been monitored at the Fernow in gauged watersheds, and long-term data are available.

Table 1.—Litterfall mass (kg ha⁻¹) and examples of mass decrease during 12 months of decomposition in two watersheds, one treated with increased rates of N deposition (WS3) and a reference (WS7). Initial litter mass from Adams (2008); decomposition mass from Piatek and others (2009a) as shown in Figure 1. Litter mass at 3, 6, and 12 months is calculated from percent of initial mass remaining (Fig. 1). “0” denotes October and “12” denotes October of the following year.

Year	WS3				WS7			
	0	3	6	12	0	3	6	12
2002	2,920	2,686.4	2,067.4	1,168.0	3,100	2,790.0	2,185.5	1,844.5
2003	3,350	3,082.0	2,371.8	1,340.0	3,440	3,096.0	2,425.2	2,046.8
2004	2,910	2,677.2	2,060.3	1,164.0	3,250	2,925.0	2,291.3	1,933.8
2005	3,630	3,339.6	2,570.0	1,452.0	3,490	3,141.0	2,460.5	2,076.6
2006	2,890	2,658.8	2,046.1	1,156.0	3,360	3,024.0	2,368.8	1,999.2

Table 2.—Example litter N contents^a (kg ha⁻¹) during 12 months of litter decomposition^b in two watersheds, one treated with increased rates of N deposition (WS3) and a reference (WS7). Nitrogen immobilization is indicated by increases over initial litter N, gross mineralization is a decrease in litter N since peak immobilization, and net mineralization is a decrease in litter N below initial content. Nitrogen dynamics adapted from Piatek and others (2009a). “0” denotes October and “12” denotes October of the following year.

Year	WS3				WS7			
	0	3	6	12	0	3	6	12
2002	35.3	39.32	41.4	26.8	33.5	40.9	42.9	26.6
2003	40.5	45.1	47.5	30.7	37.2	45.4	47.7	29.5
2004	35.2	39.2	41.3	26.7	35.1	42.9	45.0	27.8
2005	43.9	48.8	51.5	33.3	37.7	46.0	48.3	29.9
2006	35.0	38.9	41.1	26.6	36.3	44.3	46.5	28.8

^aLitter N content is calculated as litter N concentration * litter mass; Initial N concentration for WS3 was 1.21 percent and for WS, 1.1 percent (Piatek and others, in press).

^bLitter N content during decomposition is calculated as initial litter N concentration * percent of initial N remaining (from Piatek and others, in press; Fig. 1).

Table 3.—Monthly litter N fluxes (differences between months in kg ha⁻¹ mo⁻¹) during 12 months of decomposition in two watersheds, one treated with increased rates of N deposition (WS3) and a reference (WS7). Negative fluxes denote immobilization and reflect N removal from the system. Positive fluxes denote mineralization from litter, or N input.

Month	WS3					WS7				
	2002	2003	2004	2005	2006	2002	2003	2004	2005	2006
Oct	0	0	0	0	0	0	0	0	0	0
Nov	-1.3	-1.5	-1.3	-1.6	-1.3	-1.3	-1.4	-1.3	-1.4	-1.4
Dec	-1.3	-1.5	-1.3	-1.6	-1.3	-1.3	-1.4	-1.3	-1.4	-1.4
Jan	-1.5	-1.3	-1.5	-1.3	-1.7	-0.7	-4.9	-5.4	-5.1	-5.5
Feb	-0.8	-0.8	-0.8	-0.7	-0.9	-0.7	-0.7	-0.8	-0.7	-0.8
Mar	-0.8	-0.8	-0.8	-0.7	-0.9	-0.7	-0.7	-0.8	-0.7	-0.8
Apr	-0.8	-0.8	-0.8	-0.7	-0.9	4.9	-0.7	-0.8	-0.7	-0.8
May	5.3	4.6	5.3	4.6	5.7	2.3	5.0	5.5	5.2	5.6
Jun	2.0	2.0	2.3	2.0	2.5	2.3	1.9	2.1	2.0	2.2
Jul	2.0	2.0	2.3	2.0	2.5	2.3	1.9	2.1	2.0	2.2
Aug	2.0	2.0	2.3	2.0	2.5	2.3	1.9	2.1	2.0	2.2
Sep	2.0	2.0	2.3	2.0	2.5	2.3	1.9	2.1	2.0	2.2

REGRESSION ANALYSIS

Simple regression analyses of monthly litter N contents with monthly stream nitrate-N exports and concentrations were performed using SAS[®] statistical analysis package (SAS Institute, Inc., Cary, NC). Litter N contents during the immobilization phase are considered as “-”, or removal phase, and as “+”, or addition phase, during mineralization. P-values for the regression models were considered significant at $\alpha \leq 0.05$.

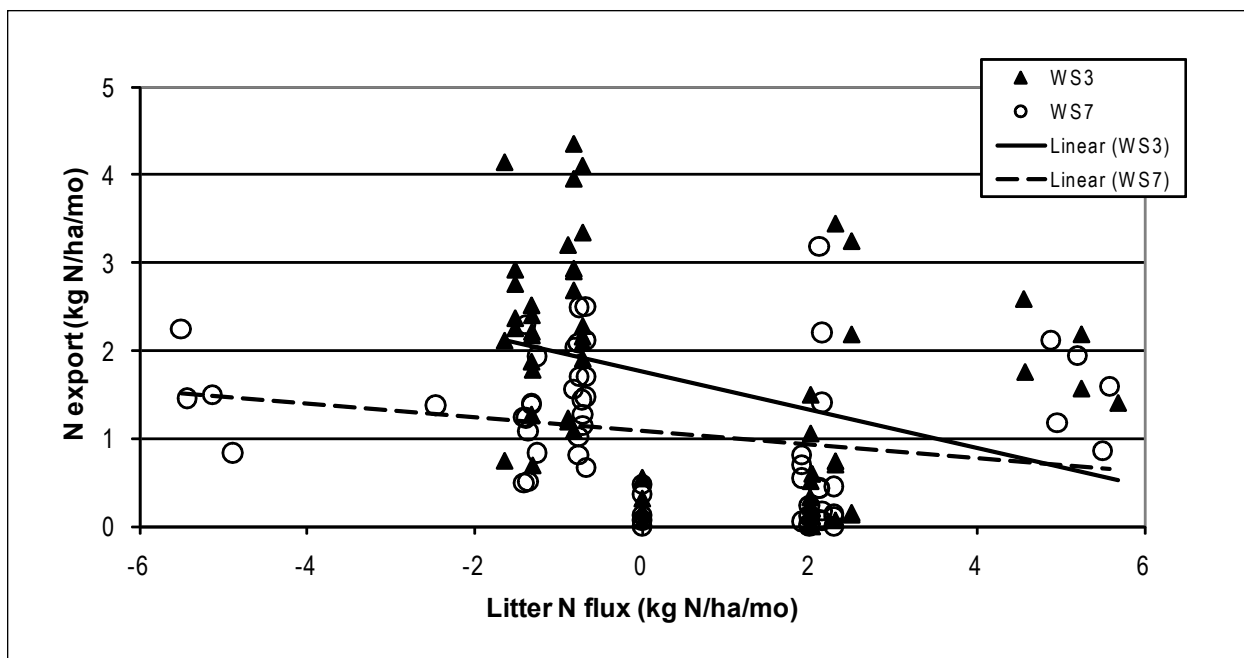


Figure 2.—The relationship of stream N exports to litter N flux. Negative litter N flux denotes microbial N immobilization as a form of N removal, while positive flux denotes N mineralization as a form of N addition. Litter N flux is the difference in N content between that in a corresponding month of decomposition and that in initial litter (from Table 1). Included are 2002-2006 monthly data for a reference (WS7) and a high-deposition watershed (WS3).

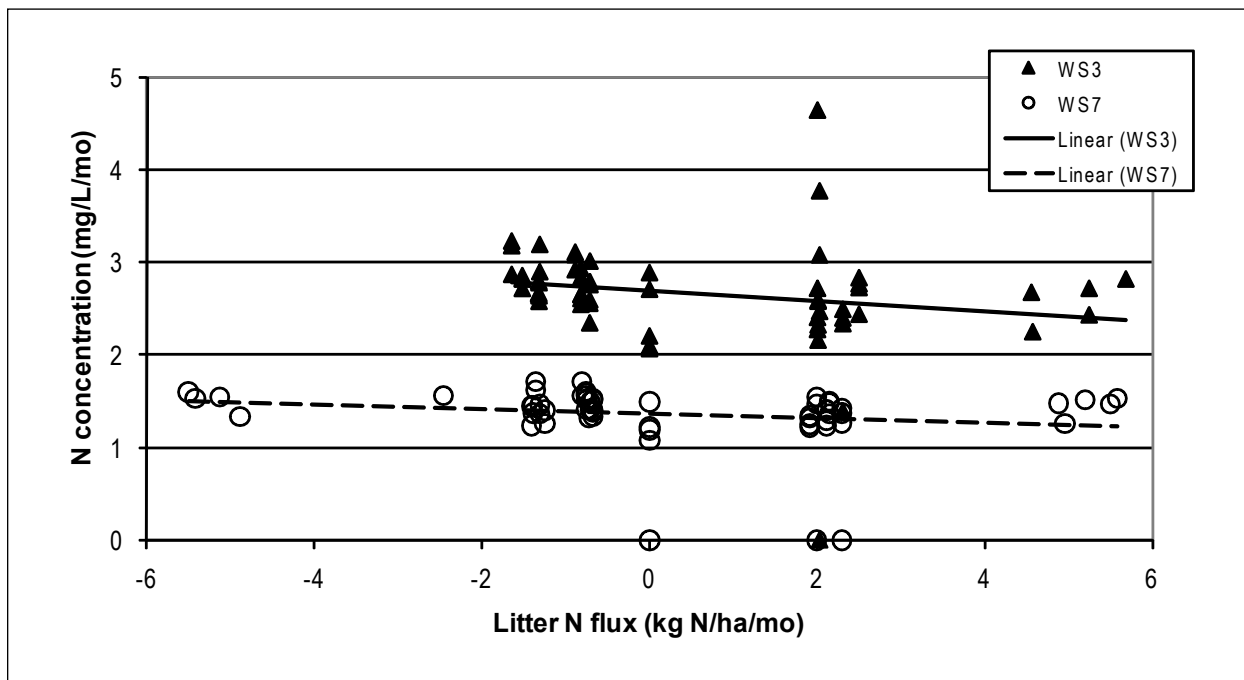


Figure 3.—The relationship of stream N concentrations to litter N flux. Negative litter N flux denotes microbial N immobilization as a form of N removal, while positive flux denotes N mineralization as a form of N addition. Litter N flux is the difference in N content between that in a corresponding month of decomposition and that in initial litter (from Table 1). Included are 2002-2006 monthly data for a reference (WS7) and a high-deposition watershed (WS3).

RESULTS

The reference watershed WS7 exhibited somewhat higher masses of litterfall than did the treated watershed WS3 for all years included except 2005. Decomposition decreased the mass of litter in both watersheds by about 40 percent in 12 months (Table 1). Nitrogen immobilization peaked in April, 6 months after decomposition started. Net mineralization started 2 months later in June (Fig. 1, Table 2). The highest monthly immobilization rates occurred during the first 3 months of decomposition. The monthly mineralization fluxes were almost two times greater than immobilization fluxes (Table 3).

We found no relationship between nitrate-N exports in stream water and litter N flux in the reference watershed (Fig. 2). Regression coefficients for the model evaluating the effects of litter N flux on stream nitrate-N exports were 0.12 ($p = 0.007$) for WS3, and 0.05 ($p = 0.079$) for WS7. There was also no relationship between stream nitrate-N concentrations and litter N flux (Fig. 3). Regression coefficients for the model evaluating the effects of litter N flux on stream nitrate-N concentrations were 0.03 ($p = 0.162$) for WS3, and 0.05 ($p = 0.106$) for WS7.

DISCUSSION

Under ambient N deposition in the reference watershed WS7, N immobilization in foliar litter did not coincide with low stream nitrate-N, and N mineralization did not coincide with high stream nitrate-N. In WS3, however, that relationship was significant, and litter N dynamics appeared to explain a small amount of variation in stream nitrate-N levels (r^2 for the regression model = 0.12). However, the relationship in the treated watershed was inverse, with higher nitrate exports during periods of N immobilization into litter, opposite of what we hypothesized (Fig. 2). We conclude that our hypothesis about the effects of decomposing foliar litter on stream nitrate levels was not supported in either watershed.

The significance of the relationship between nitrate exports and litter N in the treated watershed appears to be statistical rather than biological. The treated watershed exported more N, on average $0.5 \text{ kg N ha}^{-1} \text{ mo}^{-1}$, reaching up to $2 \text{ kg N ha}^{-1} \text{ mo}^{-1}$ more in any one month and year than the reference. At the same time, rates of litter N immobilization were lower in the litter in the treated watershed than in the reference (Fig. 2). Increased nitrate exports and decreased levels of N immobilization are typical forest-ecosystem responses to elevated levels of atmospheric N deposition (Aber and others 1989, 1998). Further, in this study, nitrate exports in summer were similar for both watersheds, probably due to comparable levels of N uptake by the vegetation; N mineralization levels from litter at that time were also similar. Therefore, the differences between the watersheds appear to consist of a combination of higher nitrate exports and lower immobilization in the treated watershed than in the reference.

Litter in the reference watershed has a greater capacity for immobilization than litter in the treated watershed (Fig. 1; Piatek and others 2009a). Even with the higher capacity for potential N retention, however, litter in the reference apparently neither removed sufficient N from the system to affect stream nitrate levels, nor did it contribute enough N to result in a significant increase in stream nitrate. In view of some new evidence that N mineralization from litter may not yield soluble forms of N as is typically assumed (Piatek 2011), it is possible that decomposing litter does not contribute soluble N. Such a functional detachment of N processes in fresh litter from N processes in partly decomposed forest floor (Oa+Oe) has been observed in hardwood forests (Currie and Nadelhoffer 1999, Fisk and Fahey 2001), and may apply to stream N exports.

Nitrate expression in stream is a function of both nitrate production and hydrologic control. Even if litter N entered soil horizons, it would become part of the soil-available N pool. Soil N mineralization rates under hardwood forests are high at 10.5 kg N ha⁻¹ mo⁻¹ for WS7 and 11.2 kg N ha⁻¹ mo⁻¹ for WS3, with very high rates of nitrification, at 91 and 105 percent of mineralization (Gilliam and others 2001). By comparison, monthly litter N fluxes averaged 2 kg ha⁻¹. Soil N cycling rates are sufficiently high to dilute the flux of N from litter, as well as from atmospheric sources. According to the flushing-draining hypothesis of nitrate movement, once nitrate is produced in soils, it may be flushed to the stream if the nitrate-source area is hydrologically connected to the stream during high levels of soil moisture. Alternatively, nitrate may drain to groundwater, when hydrologic connectivity with stream is lost at low soil-moisture levels (Creed and others 1996). This mechanism has been confirmed for some forested watersheds in the northeastern United States, and may operate at the Fernow (Piatek and others 2009b). The flushing-draining mechanism can disconnect soil N dynamics spatially and temporally from stream nitrate exports. Therefore, the next step in an effort to explain the dynamics of monthly nitrate exports in these watersheds will be to address the effects of hydrologic fluxes.

We now discuss the validity of our approach of applying litter N dynamics from one area, albeit nearby, to watersheds which may have a different species assemblage. Foliar litter invariably immobilizes N (e.g., Adams and Angradi 1996, Piatek and Allen 2001) as a result of N-limitation in forested ecosystems, including soil microbial biomass. Therefore, starting after litter fall in October, N contents in litter steadily increase over the initial level, until N mineralization begins, at which point N contents in litter decrease. In mixed-species litters, the magnitude of the increase varies depending on the relative amounts of easily decomposable and recalcitrant litters (Piatek and others, in press), but the overall immobilization/mineralization dynamics are preserved. Therefore, while total N numbers in litter may be affected under different species assemblages, immobilization will temporarily retain N and mineralization will contribute N to the system, preserving the relationship as analyzed here. As such, the validity of using litter N dynamics across similar but not the same watersheds is fully supported.

CONCLUSIONS

Nitrogen dynamics in foliar litter were not related to monthly nitrate levels in stream water under ambient or experimentally elevated rates of N deposition for the 5 years evaluated. It appears that litter N dynamics may be disconnected from stream nitrate dynamics.

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THE FATE OF NITROGEN MINERALIZED FROM LEAF LITTER – INITIAL EVIDENCE FROM ¹⁵N-LABELED LITTER

Kathryn B. Piatek¹

Abstract.—Decomposition of leaf litter includes microbial immobilization of nitrogen (N), followed by N mineralization. The fate of N mineralized from leaf litter is unknown. I hypothesized that N mineralized from leaf litter will be re-immobilized into other forms of organic matter, including downed wood. This mechanism may retain N in some forests. To test this hypothesis, oak leaves were enriched with a naturally occurring, stable isotope of N (¹⁵N), collected as litter, and allowed to decompose with and without wood (10 cm long, in 3 diameter classes of < 2 cm, 2-4 cm, and 4-6 cm) in litter bags in the forest floor of a central hardwood forest near Morgantown, WV. As oak litter immobilized exogenous N, ¹⁵N in litter decreased due to dilution, and then did not change significantly. Nitrogen mineralization in leaf litter did not start for at least 24 months. By the 11th month of decomposition, wood N reached between 140 and 274 percent of the initial N mass, depending on wood diameter. By the 24th month, ¹⁵N was detected in wood, supporting the hypothesis. Nitrogen-label was not detected in fresh litter above litter bags, or below litter bags in partly decomposed forest floor or soil. It appears that N mineralized from leaf litter may be transferred to and immobilized in small-diameter downed wood. If these results are confirmed in a larger study, we may need to change the way we view N cycling in the leaf litter layer; N mineralized from foliar litter may not be immediately plant-available as we now assume.

INTRODUCTION

Decomposition of foliar litter is a microbial process in which microbes use litter carbon (C) as a source of energy (Gallardo and Schlesinger 1994). While feeding on litter C, soil microbes sequester nitrogen (N) from the environment for the formation of new cells, resulting in a net increase in litter N (Berg and Staff, 1981). As decomposition progresses and labile C in litter is used up, N in the litter-microbe complex starts to decline in the process of N mineralization (Gosz and others 1973, Edmonds 1980, Berg and Staff 1981, Moore and others 2006, Piatek and others 2009).

The fate of the N that becomes mineralized from fresh foliar litter is unknown. It is typically said that leaf litter plays an important role in forest nutrition and productivity, the assumption being that mineralization of leaf litter yields plant-available forms of N. If litter N is not taken up by plants, as the case may be after forest harvest, mineralized N is nitrified and leached, potentially impacting surface water quality. However, decomposition of leaf litter proceeds within a matrix of other C pools, which constitute sources of energy for microbial growth and opportunities for further N immobilization. Therefore, partially decomposed old leaf litter layers, downed wood, roots, root exudates, and annual litter fall are possible alternative transfer pathways for leaf-mineralized N. Since forests are N-limited (Vitousek and others 1997, Oren and others 2001), and the presence of labile C is expected to result in immobilization by microbial biomass of any available N in

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N-limited systems, it seems unlikely that N mineralized from foliar litter becomes plant-available. Rather, N released from decomposing foliar litter may be rapidly re-immobilized into microbial biomass, which utilizes the next most labile source of C.

Theoretically, this re-immobilization process could continue until all labile C is exhausted. Since a new leaf litter layer is formed each fall in standing deciduous forests, in theory this closed N cycling can continue in perpetuity, or until the forest is cut and C in annual litterfall is no longer available. Downed wood may also be attractive for microbes because woody biomass contains up to 50 percent C. However, N dynamics in wood remains poorly understood due to lack of methods for accounting for loss of wood volume during decomposition (Krankina and others 1999, Creed and others 2004). Addition of N tracers showed negligible immobilization of N into fine woody debris several years later (Currie and others 2002). However, increases in wood N of up to 180 percent of initial N mass were observed in Douglas-fir early in decomposition (Edmonds and Eglitis 1989). Wood chips are apparently also capable of immobilizing N (Homyak and others 2008). A greater understanding of the role of wood in N retention is of considerable interest in intensively managed forests where sites are cleared of woody residues in preparation for tree planting. Furthermore, with a growing interest in using woody biomass for the production of biofuels, extraction of woody residues is likely to increase with as yet unknown consequences for N retention and nitrate exports.

Decomposition studies of foliar litter, however, rarely trace the fate of *released* N, making the term *N-release from litter* potentially misleading. Possible fates of litter N include mineralization to ammonium, re-immobilization, nitrification to nitrate, leaching of dissolved organic N (DON), and a migration of N bound within fungal filaments (Hart and Firestone 1991). A few studies also suggest that foliar litter does not interact with other soil or forest floor N processes (Currie and Nadelhoffer 1999, Fisk and Fahey 2001), increasing the uncertainty of the fate of litter-released N. Fates of litter-released N may determine such ecological conundrums as controls on short- and long-term N availability in N-limiting systems, and nitrate production and export after disturbance or in N-saturated systems.

Nitrogen transfers and cycling in ecosystems can be elucidated using a ^{15}N -labeling technique. Nitrogen-15 is a naturally occurring stable isotope of N which has been used to trace the fate of atmospherically deposited inorganic N (Currie and Nadelhoffer 1999, Piatek and others 2005). It has not been widely used to produce leaf-organic N compounds. To produce labeled organic N, an individual tree is injected with a ^{15}N -enriched solution early in the growing season to allow for the formation of organic N compounds in leaves. Subsequently, leaves can be collected after natural abscission and leaf fall (Christenson and others 2002). Sufficient amounts of the enriched material need to be produced to conduct a decomposition experiment, which typically lasts 1 to 2 years with several collections. Moreover, initial material must contain sufficient levels of label for label recovery among several possible pools in the forest floor. Enriched materials are analyzed by mass spectrometry at select laboratories across the country. Given these challenges, considerable amount of time and costs of isotopic analyses render the labeling technique difficult and expensive to employ and therefore still not widely used. Initial evidence, however, can provide an important justification for a further study.

To provide initial evidence for the fate of N mineralized from leaf litter and the role of downed wood in N retention, I conducted an exploratory experiment using ^{15}N -enriched oak leaf litter to trace N movement from decomposing leaf litter to downed wood, old forest floor layers, freshly deposited leaf litter, and soil. The central hypothesis tested was that N mineralized from leaf litter is re-immobilized into other available C pools, including downed wood.

METHODS

EXPERIMENTAL MATERIAL

Enrichment methods are treated in detail in Christenson and others (2002). Briefly, the bole of a northern red oak tree (*Quercus rubra*), 10 cm in diameter, was injected with a ^{15}N -enriched $(^{15}\text{NH}_4)_2\text{SO}_4$ solution in spring and summer 2004, and leaf litter was collected in fall. Fallen litter was air-dried for several months prior to use.

Availability of air-dried branch wood at the time when labeled leaf litter was available reduced the choice to a wild-grown *Malus* species. While not widely present in the central hardwood forest, it is found in now-forested old homestead sites across the northeastern United States. *Malus* then satisfied the need for a source of wood-derived C for potential microbial use.

About 6 grams of ^{15}N enriched litter, weighed to the nearest 100th of a gram, were placed in 1-mm mesh bags made of nylon window screening into 20-cm x 20-cm pockets. Twenty-two litter bags contained ^{15}N -enriched litter alone; the contents of these bags became litter type “leaves (-),” or without wood. Twenty-two other litter bags received three air-dried woody pieces 10 cm long, one in each diameter class – small (up to 2 cm in diameter), medium (2 - 4 cm), and large (4 - 6 cm) with intact bark; the contents of these bags became litter type “leaves (+),” or with wood. Each woody piece was individually weighed. Two samples of each litter type were then oven-dried at 65-70 °C until constant weight to determine the air-to-oven-dry weight adjustment for all other samples.

The remaining 20 litter bags of each litter type were placed on the forest floor at the West Virginia University Research Forest outside of Morgantown, WV (39°41'N, 79°45'W) in early November 2004, and secured with landscape pins. Placement was not random; rather, each type of litter was placed in a 40-m-long row, with each bag 2 m away from another bag in that row, and rows at least 5 m apart, for final plot size of about 60 m x 20 m, without replication. The objective behind this placement was to prevent ^{15}N from transfer between mesh bags. Bags within rows were randomized according to their number (from 1 to 20).

The forest type where the study took place is a mature central hardwood forest, with a high red oak component, and little understory. Average tree diameter at breast height is estimated at 36 cm. The forest floor depth is less than 2 cm. Dead downed wood is amply represented, though the amounts were not quantified.

SAMPLE COLLECTION AND PROCESSING

Two bags of each type [leaves(+) and leaves (-)] were collected every 3 to 4 months, with the first collection in December 2004, and the last in June 2007, for a total of 10 collections. New litter fallen on top of the mesh bags, foliage under the mesh bags (old forest floor or Oe + Oa horizons), and soil beneath bags to 20 cm depth were also collected for analysis of ^{15}N . Additionally, new litter, old forest floor, and soil were collected in six places away from any litter bags for analysis of background ^{15}N levels. Fresh unconfined litter did not contain any fine roots. Partly decomposed “old” forest floor layers, however, are made of materials tightly pressed together, and any separation becomes impossible. Soils were sieved prior to further analysis.

Materials from mesh bags were separated into leaf and wood components. Any fine roots growing into the bags were taken out, although with increasing litter decomposition that process becomes less complete. Samples from mesh bags and nonconfined background materials were oven-dried at 65-70 °C until constant

weight. All materials were subsequently weighed and ground. Due to substantial hygroscopicity, mass in dried wood fluctuated over time; to account for this fluctuation, a mass loss function was derived with an r^2 of 0.7.

Nitrogen content in decomposed materials was calculated as N mass at collection time over N mass at the start of decomposition, in percent. Net immobilization is reflected in increasing mass of litter N as decomposition progresses, while net mineralization is reflected in decreasing N mass.

¹⁵N ANALYSES

Subsamples of the ground samples were analytically weighed into 5-mm x 9-mm tin capsules (Costech Analytical International) and analyzed by mass spectrometry at the University of California–Davis stable isotope facility. Results are obtained as the ratio of ¹⁵N to ¹⁴N in sample versus that of an international standard (Air; <http://stableisotopefacility.ucdavis.edu>, accessed Dec. 1, 2009):

$$\delta^{15}\text{N} [\text{‰}] = \left(\frac{{}^{15}\text{N}/{}^{14}\text{N}_{\text{sample}}}{{}^{15}\text{N}/{}^{14}\text{N}_{\text{standard}}} - 1 \right) * 1000$$

The magnitude and timing of the decrease in ¹⁵N in enriched litter indicates ¹⁵N dilution during N immobilization (microbial input of litter-external N). An increase in ¹⁵N in wood, freshly fallen litter, old forest floor, or soil indicates a transfer by microbes of ¹⁵N from litter. Such transfer serves as initial evidence of the potential role of wood in N retention in hardwood forests.

MICROBIAL BIOMASS DYNAMICS

To evaluate microbial biomass dynamics in the experimental tissues, the method of Aber and Melillo (1982) was used. In this method, tissue initial N concentration is assumed constant, so that the increase in total substrate N over time is due to microbial biomass growth and N. N concentration in decomposing tissue is plotted against the remaining weight, and the equation parameters with N concentrations over time divided by original litter concentration are used to generate polynomial regression plots for microbial biomass (Aber and Melillo 1982). Obtained in this manner, microbial biomass dynamics helps in interpreting the observed ¹⁵N data. Namely, when ¹⁵N increases in a new (previously unlabeled) tissue, and is accompanied by microbial biomass growth in that tissue, it serves as an indication that the microbial transfer of label has occurred.

TREATMENT OF DATA

The study's objective was to provide initial evidence for a larger inquiry into N transfer pathways in the forest floor. The limited amount of enriched materials made replication impossible. Therefore, statistical evaluation of main effects of 'row' and 'tissue type' could not be conducted, as replication was only within the time of collection. The combined use of labeling and microbial biomass dynamics techniques provides synergistic evidence, but results were interpreted with due caution.

RESULTS

MASS LOSS

Twenty-four months after start of decomposition, 60 percent of leaf litter mass was lost in litter decomposing with (+) and without wood (-) (Fig. 1). The small starting sample size and relatively long decomposition period resulted in highly fragmented tissue, which did not allow some of the analyses to go beyond 24

months. Approximately 40 percent of wood mass was lost after 24 months, increasing to 50 percent by the end of the study, at 32 months of decomposition.

NITROGEN DYNAMICS IN DECOMPOSING LEAVES AND WOOD

Nitrogen concentrations in oak leaf litter increased from the initial 0.58 percent to the highest level of 2.32 percent for leaves (+) and 2.41 percent for leaves (-) in the 24th month of decomposition. Subsequently, N concentrations in leaf litter started to decline slowly (Fig. 2). Nitrogen concentrations in wood ranged from the initial 0.19 percent to an average for all diameter classes of 1.2 percent (Fig. 2).

Leaf litter immobilized N for 24 months of decomposition, observed as an increase in litter N mass (litter mass x N concentration). Leaf litter with (+) and without (-) wood had similar net N immobilization rates over the study period, except at 11 months, when leaves decomposing without wood immobilized 36 percent more N than leaves decomposing with wood (Fig. 3). After 24 months, an average of 163 percent of original N mass was present in leaf litter. Wood N contents in the large- and medium-diameter classes were similar, but the small-diameter wood exhibited large N increases and decreases. After 24 months of decomposition, an average of 125 percent of initial N mass was present in large- and medium-diameter wood. The highest net N immobilization in the small-diameter wood was 274 percent of original N mass at 11 months. Net mineralization over the next 9 months resulted in a decrease in total N content in small-diameter wood (Fig. 3).

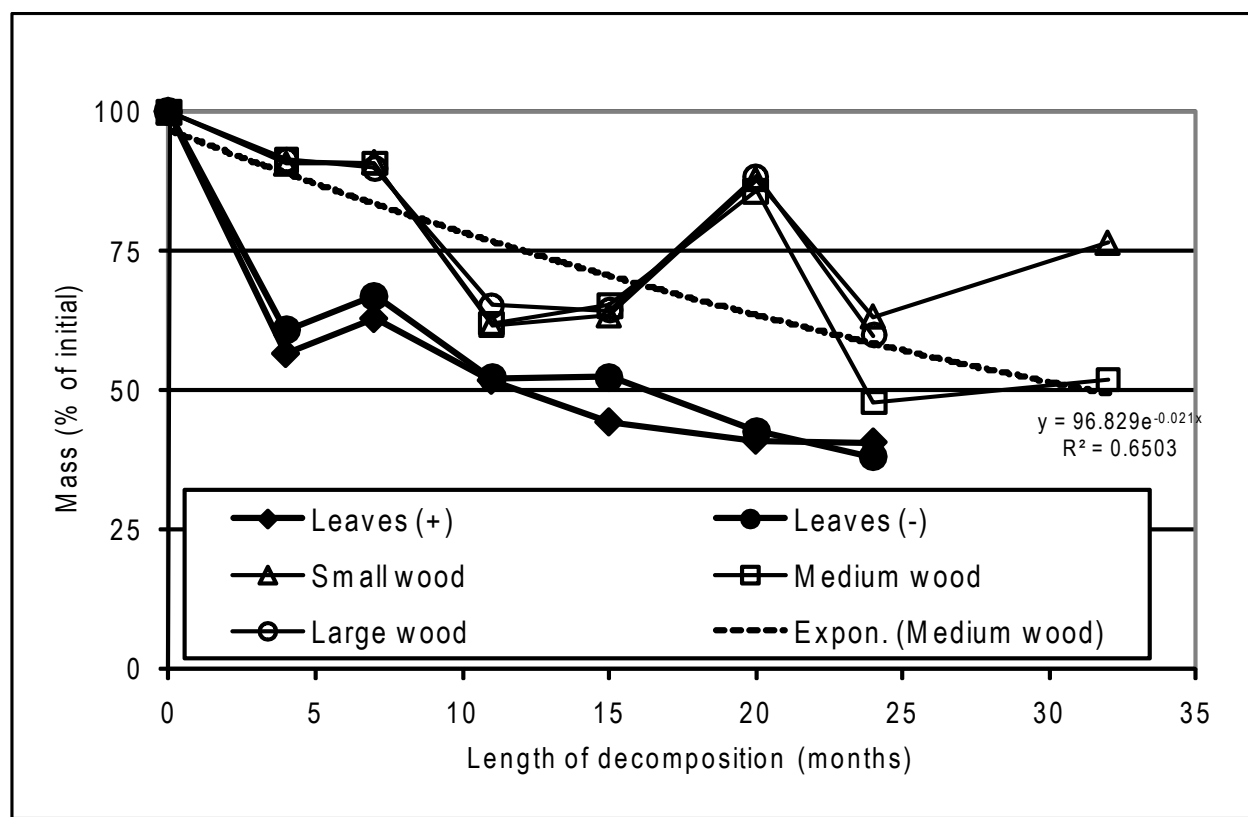


Figure 1.—Mass loss dynamics of leaves decomposing with wood (+) – dark solid diamonds, and without wood (-) – dark solid circles, and wood in three size classes (small – triangles, medium – squares, and large – circles) over time. Regression line denotes average wood decomposition.

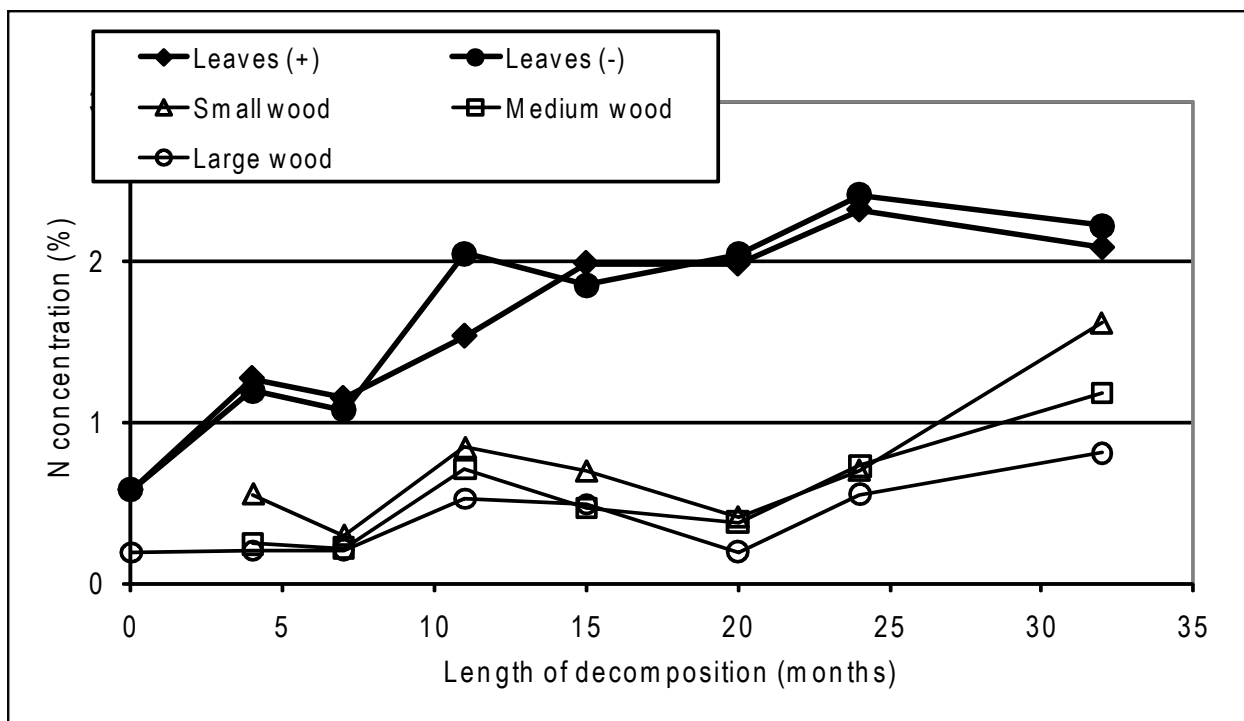


Figure 2.—Nitrogen concentration in leaves and wood during decomposition.

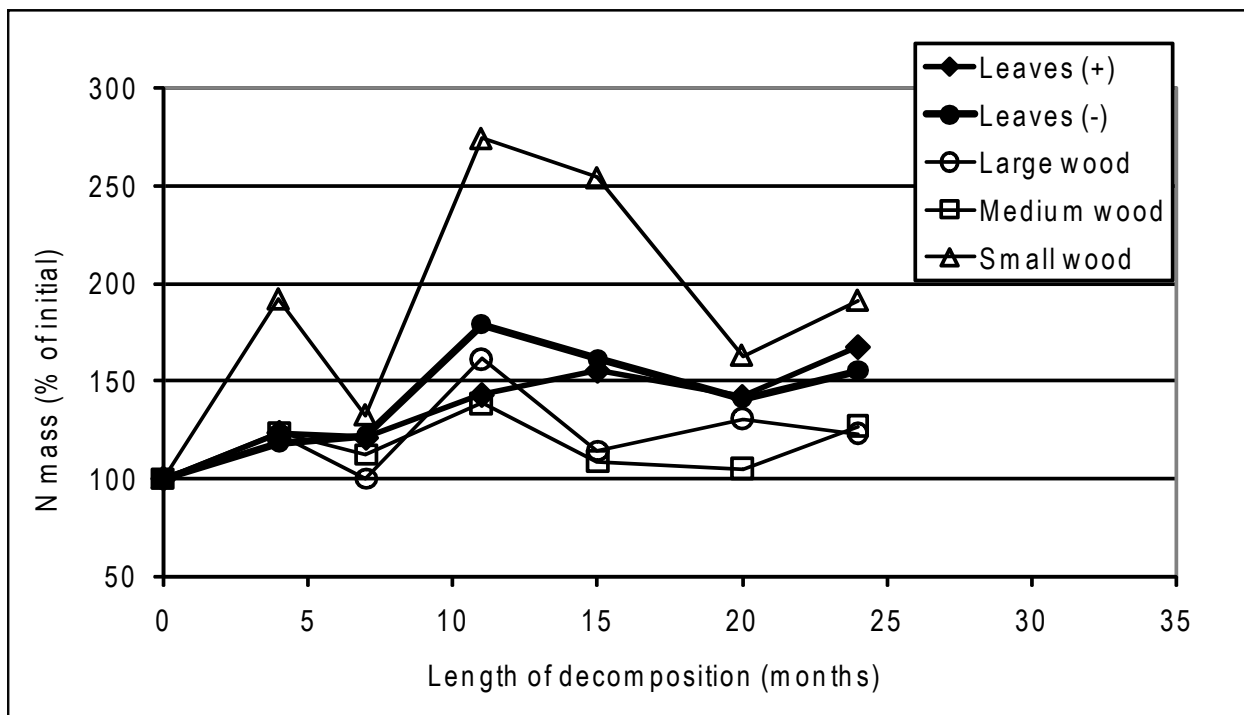


Figure 3.—Nitrogen dynamics during decomposition of leaves decomposing with wood (+) – dark solid diamonds, and without wood (-) – dark solid circles, and wood in three size classes (small – triangles, medium – squares, large – circles) over time.

CHANGES IN $\delta^{15}\text{N}$

The average starting level of ^{15}N label in experimentally enriched leaf litter was +162‰, compared to the background levels with the range of -1.4 ‰ in small diameter wood to +8.9 ‰ in the soil at the site (Fig. 4). Early $\delta^{15}\text{N}$ dynamics in leaf litter included a dilution of the label by N immobilized by leaf-colonizing microbes. Between 11 and 20 months, ^{15}N levels appeared to stabilize around 82‰ for both leaf litters (decomposing with and without wood), and then decreased to 57‰ after 24 months, and to 48‰ after 32 months of decomposition. During 0-20 months of decomposition, label was not detected in any of the other types of organic matter tested (Fig. 4). After 20 months, a slight increase was noted in wood in all size classes. This increase continued. Label was recovered in wood in all diameter classes after 24 months of decomposition, indicating a transfer of label between leaf litter and wood. After 24 months, $\delta^{15}\text{N}$ in wood ranged from 113‰ in the medium wood to 148‰ in small wood. Over the next 8 months, enrichment in wood declined to background levels in all wood sizes.

MICROBIAL BIOMASS DYNAMICS

Leaf litter with and without wood had similar microbial biomass dynamics until N concentrations reached about 1.5 percent in month 9 and 11 for leaves with and without wood, respectively (Fig. 2 and 5). Beyond tissue N concentration of 1.5 percent, microbial dynamics diverged. Leaves decomposing with wood supported less biomass per unit increase in N concentration than did leaves without wood. Wood supported much greater levels of microbial biomass at much lower N concentrations than did the leaves (Fig. 5), with small-diameter wood supporting more than large-diameter wood (Fig. 5). The trajectory for wood-supporting microbial biomass was still increasing, especially for the small-diameter wood, when the experiment ended after 32 months.

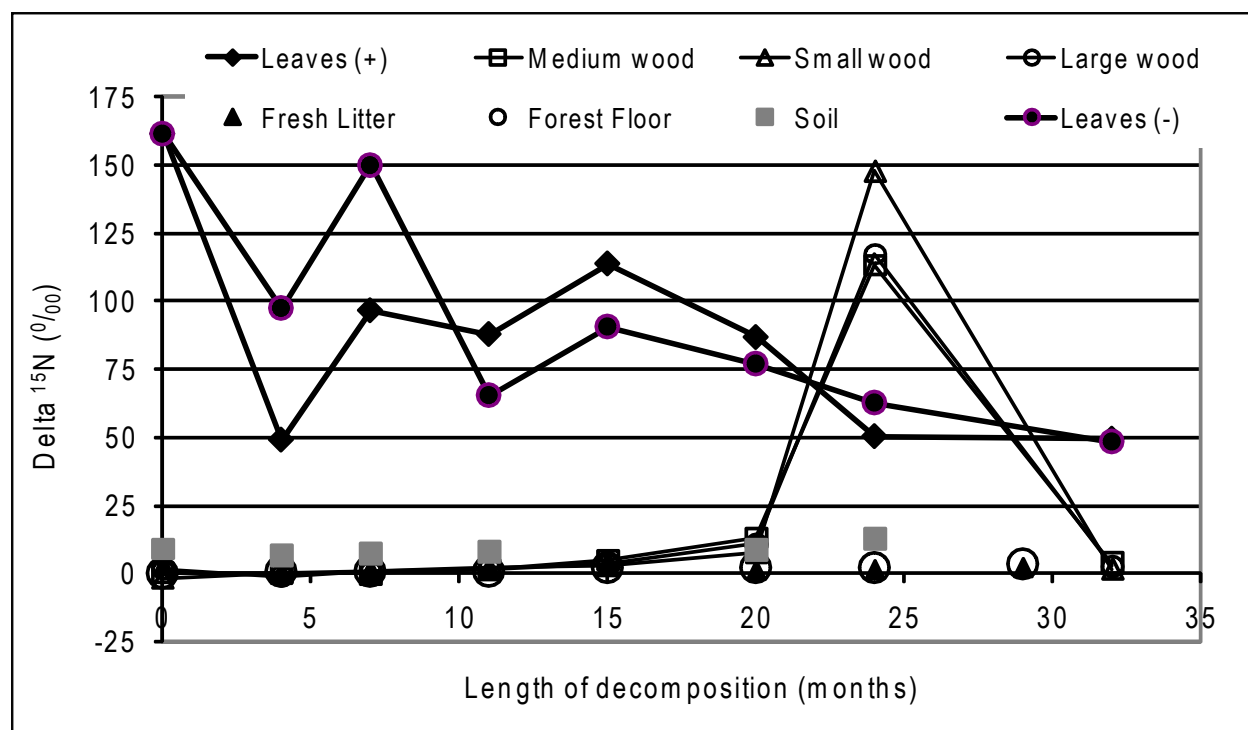


Figure 4.—Delta ^{15}N dynamics during decomposition of leaves decomposing with wood (+) – dark solid diamonds, and without wood (-) – dark empty diamonds, and wood in three size classes (small – triangles, medium – squares, and large – circles) over time, and fresh forest floor (soil – light solid squares, old forest floor – dark solid diamonds, no line, and fresh litter – triangles).

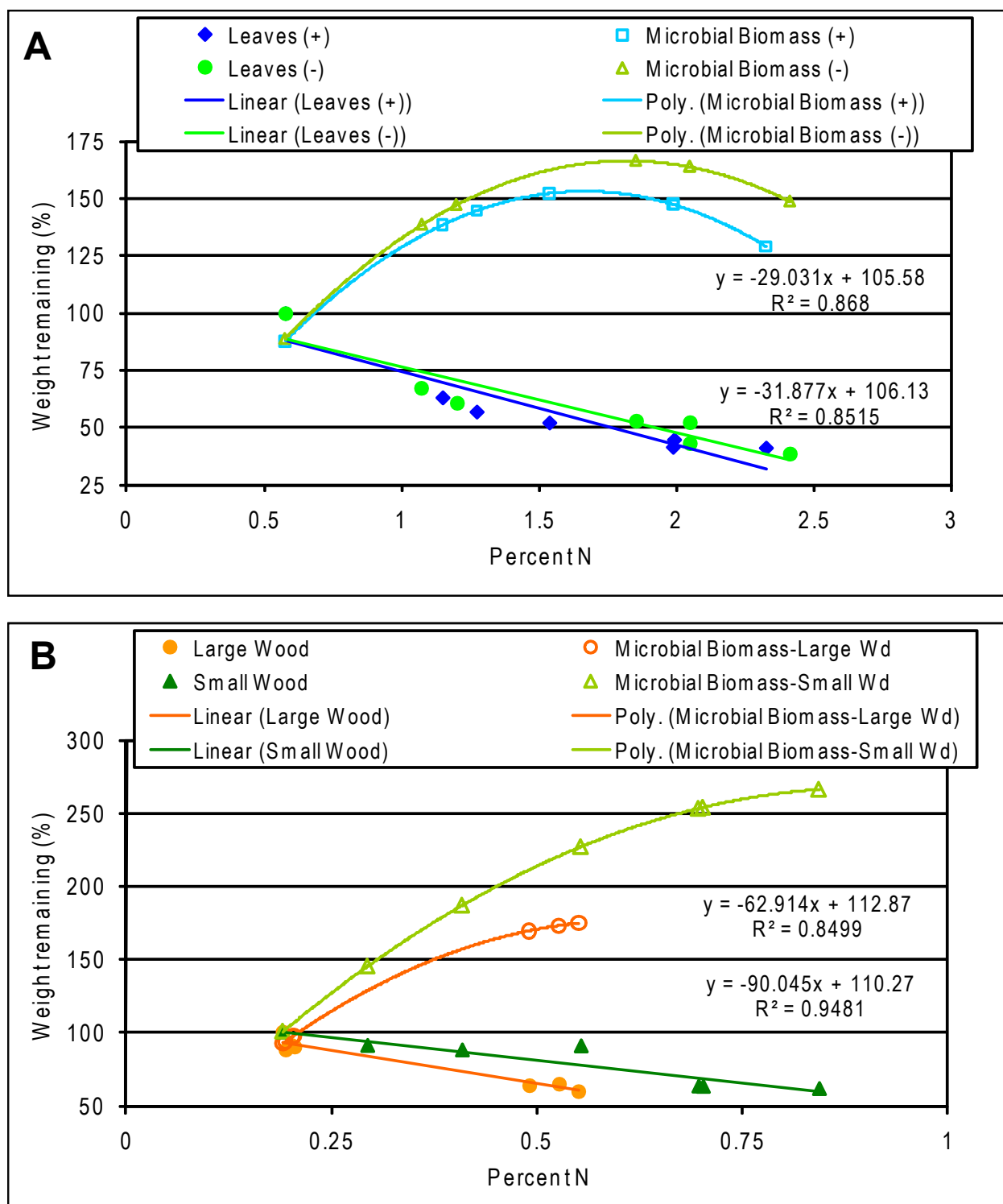


Figure 5.—Estimated microbial biomass N dynamics plotted as N concentration vs. N content (%) (after Aber and Melillo 1982). Top (A): Leaves with wood (+) – dark solid diamonds, leaves without wood (-) – solid circles, leaf microbial biomass with wood (+) – open squares, leaf microbial biomass without wood (-) – open triangles. Bottom (B): Wood pieces: Large wood – solid circles; Small wood – solid triangles. Microbial biomass: Large wood – open circles; Small wood – open triangles.

DISCUSSION

THE FATE OF N MINERALIZED FROM LEAF LITTER

The delayed appearance of N label in wood supports the study hypothesis that N mineralized from leaf litter is re-immobilized into other available C pools, in this case wood. The one-time increase and subsequent decrease in ^{15}N in wood to background levels is difficult to explain given the small overall sample size. However, six pieces of wood per collection responded in the same manner. These observations are consistent with immobilization of N into downed wood observed during decomposition of Douglas-fir and in wood chips of mixed northern hardwoods (Edmonds and Eglitis 1989, Holub and others 2001, Homyak and others 2008). Currie and others (2002) did not observe label transfer to wood, but the observations were made during advanced stages of wood decomposition.

Transfer of N label mineralized from experimental leaf litter to wood was expected to be accompanied by a corresponding decrease in $\delta^{15}\text{N}$ in leaf litter. However, the decrease in $\delta^{15}\text{N}$ in leaf litter was relatively small compared to the increase in wood. The decrease in early stages of decomposition can be explained by label dilution. Dilution was indicated by a decrease in $\delta^{15}\text{N}$ accompanied by an increase in litter total N (i.e., immobilization; Fig. 3), and a lack of label outside of the enriched leaf litter (Fig. 4). The decrease in leaf $\delta^{15}\text{N}$ together with the decline in leaf microbial biomass at that stage of decomposition (Fig. 5) suggest that some of the leaf litter-inhabiting microbes either died or migrated to other sources of C. Dead microbial biomass constitutes easily decomposable organic matter that is subject to mineralization, while the released microbial N can be re-immobilized by other microbes. Because the increase in $\delta^{15}\text{N}$ occurred in wood alone, the “other” microbes were most likely wood decomposers. The sum of ^{15}N after 24 months indicated that enrichment level in all tissues together reached 425‰, up from the starting 162‰ in leaves. Enrichment can be due to a concentrating effect of decreasing leaf and woody tissue mass on ^{15}N (and ^{14}N), as well as to microbial biomass turnover (Fig. 1 and 5).

Total N dynamics were the same in leaves decomposing in the presence or absence of wood. One exception occurred in the 11th month of decomposition (Fig. 3), when N concentration in leaf litter reached 1.5 percent and microbial biomass started to decline in leaves decomposing with wood (Figs. 2 and 5). This decline is not likely to be due to competition for available N between wood- and leaf-colonizing microbes for at least two reasons. In September, the 11th month of study, ecosystem N-sink activity is probably at its lowest, and N availability at some of the highest during the year due to slowing N uptake by plants and temperatures conducive to soil organic matter mineralization. It is then unlikely that microbial competition for available N operates under these conditions of N supply and demand. At the same time, the content of labile C in leaf litter declined after almost a full year of microbial growth and C utilization. Therefore, the decline in microbial biomass in leaf litter decomposing with wood but not in leaf litter decomposing alone may be associated with a migration of microbes from a low-C substrate to a higher-C substrate in wood.

The transfer of N label with microbes migrating from leaf litter was expected into the freshly deposited litter layer (on experimental litter bags). Fresh litter is expected to have higher contents of labile C than wood. Lack of transfer to the older forest floor below the mesh bags is consistent with earlier findings suggesting little interaction between N cycling processes in fresh litter with the underlying layers (Currie and Nadelhoffer 1999, Fisk and Fahey 2001, Christenson and others 2002). The lack of interaction may extend in the other direction as well. Lack of N transfer to soil is best confirmed with data allowing calculations of ^{15}N pools

which were not quantified in this study; nevertheless, estimates (data not shown) revealed no changes between soils away from enriched litter and soils directly under litter bags. Again, these results are not surprising if N in our initial litter (in litter bags) does not interact with the forest floor or soils below.

Types of decomposers are likely to differ among substrate types, especially when substrate types vary in their chemistry as much as do leaf litter and wood. Therefore, as microbial biomass in leaf litter declines (Fig. 5), probably as a result of a decrease in available C, the products of microbial biomass turnover in leaf litter may be immobilized by and translocated by wood-decomposing fungi, thus appearing in wood (as evidenced by changes in $\delta^{15}\text{N}$). Translocation of N within fungal filaments has been observed (Hart and Firestone 1991). Thus, neighboring wood materials may not necessarily be more “attractive” to decomposers because of their accessibility than labile C sources farther away, but wood-inhabiting fungal decomposers may be capable of accessing the products of microbial turnover in leaf litter. The described potential N transfer pathways are supported by patterns of estimated microbial biomass growth in leaves and in wood (Fig. 5); namely, when leaf microbial biomass started to decline, wood-supported biomass was expanding. However, different N transfer pathways may operate depending on the type of physical barriers and on chemical properties of decomposition microsites.

POTENTIAL IMPLICATIONS OF THE FINDINGS

A transfer of leaf-litter-mineralized N to wood and wood immobilization of N were observed in this study. The carrier of N label in this study was foliar litter of northern red oak. Northern red oak is the dominant species in the central hardwood forest type (Hicks 1998), and its relative component may reach up to 93 percent (Kromroy and others 2008). Therefore, it is a good choice for an exploratory study of N transfers in the litter layer. However, decomposition is slower and N dynamics in oak exhibits higher rates of immobilization and slower mineralization than other leaf litter types in the central hardwood forests (Piatek and others 2009, Piatek and others 2010). For example, red maple or yellow-poplar leaf litter, both common in the central hardwoods, could be expected to initially immobilize and mineralize N faster than observed here, and at smaller levels (Piatek and others 2009). As a result of faster dynamics but lesser amounts involved, label dilution in leaf litter would be less in these other species, and mineralization and re-immobilization of N to other C sources may be observed earlier in decomposition than in oak litter. Therefore, some of the results related to the timing of N transfers may be particular to the decomposition dynamics in oak litter. Overall dynamics and N transfers, however, are likely to be preserved.

The observed N immobilization into wood merits a further study into the role of wood as an N retention mechanism in forests. Mechanisms of N retention are of considerable interest on sites that either have high rates of N turnover (such as clearcut forests) or receive high rates of N deposition (Magill and Aber 1998, Piatek and others 2009); both situations may result in potentially hazardous levels of nitrate leaching into drinking water supplies or aquatic habitats. A potential N-immobilization capacity of downed wood, especially at 140-250 percent of initial mass as observed here, would mean that 45-60 kg N ha⁻¹ can be retained on a site with a hypothetical 25-kg initial N mass in fresh downed wood. (At average wood concentrations of 0.1 percent N, 2,500 kg downed wood ha⁻¹ would be needed). This amount of N retention would account for substantially more than current rates of atmospheric N deposition of about 12 kg N ha⁻¹ in the central hardwood forest region (Adams and others 2000). With subsequent mineralization of N (as indicated in this study), these rates would not translate into a long-term N sink.

Large amounts of woody residues are currently left on the ground each time forests are harvested (Grushecky and others 1997, Ares and others 2007). With the increased interests in biofuels, however, harvesting residues may be increasingly sought after for conversion to wood energy. It is therefore important to fully elucidate the role of downed wood in N cycling in our forests. This initial evidence suggests that removal of downed wood and its N immobilization capacity may have important consequences for water quality, among other ecological impacts.

CONCLUSIONS

The results of this study serve as initial evidence of potentially tight N cycling within microbial biomass between C pools in the decomposition matrix of the forest floor. If confirmed, these results may necessitate a change in the way we view N cycling after mineralization from the leaf litter layer. Contrary to current assumptions, N mineralized from foliar litter may not be immediately available to plants.

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CANOPY GAP CHARACTERISTICS OF AN OLD-GROWTH AND AN ADJACENT SECOND-GROWTH BEECH-MAPLE STAND IN NORTH-CENTRAL OHIO

David M. Hix, P. Charles Goebel, and Heather L. Whitman¹

Abstract.—The increased importance of integrating concepts of natural disturbance regimes into forest management, as well as the need to manage for complex forest structures, requires an understanding of how forest stands develop following natural disturbances. One of the primary natural disturbance types occurring in beech-maple ecosystems of the Central Hardwood Forest is canopy gaps. We characterized canopy gaps of an old-growth beech-maple stand in north-central Ohio and compared these characteristics with an adjacent mature second-growth stand. Using a line-intercept approach, we found that 9.3 percent of the forest area of the old-growth stand was in canopy gaps while 3.7 percent of the second-growth stand was in canopy gaps. Mean canopy gap size was not different between the old-growth and second-growth stands (145.6 m² and 126.8 m², respectively). Mean gap-maker size was larger in the old-growth stand than in the second-growth stand, and the species compositions of gapmakers were similar to the surrounding canopy trees. However, the modes of canopy gap formation were not different despite the differences in gapmaker species. Sugar maple and American beech are the dominant tree species regenerating in canopy gaps in both stands. Based on our results, forest managers may be able to emulate canopy gap dynamics in similar forest ecosystems using the selection method.

INTRODUCTION

Mature hardwood forests, including beech-maple (*Fagus L.-Acer L.*) ecosystems, were once extensive throughout the Central Hardwood Forest (National Council for Science and the Environment 2008). Much of the original forest has been converted to other land uses, primarily agriculture and urban development (Parker 1989). In an effort to restore and increase the complexity of altered forest ecosystems, a recent focus of forest ecologists and managers is to emulate natural disturbance regimes using innovative silvicultural practices (Franklin and others 2007). The few remaining old-growth stands are essential benchmarks, providing reference conditions for forest ecosystem restoration and management (Society for Ecological Restoration International Science and Policy Working Group 2004). Studying these stands will help us understand ecosystem processes of undisturbed forests, as well as learn how management practices may affect those processes and the ecological integrity of managed stands (National Council for Science and the Environment 2008). Using comparative studies of second- and old-growth stands is important when restoring the old-growth characteristics we may desire to develop in younger forests (Lorimer and Frelich 1994).

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One important forest ecosystem process needing more study is the canopy gap dynamics of old-growth stands. This process is important because canopy gaps directly affect the successional trajectory of a stand by influencing regeneration development, species composition, growth rates, and age structure (Lorimer 1989, Runkle 1992, Franklin and Van Pelt 2004). Specifically, the size, origin, and shape of gaps affect several stand characteristics (Runkle 1992). Individual-tree death causing canopy gaps has been found to be the primary disturbance type in beech-maple forests (Runkle 1982).

The objective of this study was to determine and contrast the canopy gap characteristics of adjacent old-growth and mature second-growth beech-maple stands in a north-central Ohio forest. This forest provides a unique opportunity to study the canopy gap characteristics of adjacent old-growth and mature second-growth beech-maple stands. Both stands have very similar environmental conditions, but they have had different disturbance histories.

STUDY AREA

This study was conducted at Crall Woods, a 37.4-ha forest in Ashland County, OH. The forest was designated a National Natural Landmark in 1974, one of 23 currently in Ohio (Ashland County Park District 2005). It is one of the few remaining old-growth forests in north-central Ohio. Two of its stands were compared in this study: a 4.0-ha mature second-growth stand and a 16.2-ha old-growth beech-maple stand.

Crall Woods is located in the Low Lime Drift Plain (Level IV Ecoregion) of the Erie/Ontario Drift Lake Plain. The Low Lime Drift Plain is characterized by gently undulating topography with scattered end moraines and kettles (Woods and others 1998). Annual precipitation in the region ranges from 900 to 1,030 mm, and annual average temperature ranges from 10 to 13 °C (McNab and Avers 1994). The predominant soil series of the area is Bennington silt loam, a somewhat poorly drained soil originating from glacial till (Aughanbaugh 1964). Embedded within Crall Woods are small vernal pools that have standing water for extended periods of the year.

The old-growth stand is dominated by sugar maple (*Acer saccharum* Marsh.), American beech (*Fagus grandifolia* Ehrh.), American basswood (*Tilia americana* L.), and yellow-poplar (*Liriodendron tulipifera* L.), and has essentially been undisturbed by humans for approximately 250 or more years (Pinheiro and others 2008). The second-growth stand contains many of the same species, as well as northern red oak (*Quercus rubra* L.) and various hickory (*Carya* Nutt.) species (Pinheiro and others 2008). Pinheiro and others (2008) determined that the second-growth stand is approximately 140 years old, and it is becoming more similar in species composition to the old-growth stand.

METHODS

DATA COLLECTION

We used a modified version of Runkle's (1992) methodology to sample the canopy gaps of both stands at Crall Woods. Four parallel line transects were permanently established approximately 50 m apart and inventoried during August 2008. The two stands share an east-west boundary; each transect ran north-south from the second-growth stand into the old-growth stand across this boundary. Since these transects were permanently located, we were able to sample them again during November 2009. The spacing between each pair of transects was chosen to assure that no canopy gap was intersected by more than one transect. Their

placement was designed to avoid all stand edges by at least 50 m, and to sample the entirety of each stand. The transects ranged in length from 574-819 m, primarily because their ending points within the old-growth stand varied in relation to its southern boundary.

Along the length of each transect, the canopy was classified as either closed or a gap. Canopy gaps were areas where the canopy height was < 50 percent of that of the adjacent forest. When a canopy gap was intersected, these characteristics were noted: the number of gap makers (i.e., a single- or a multiple-tree gap), the species and diameter at 1.37 m above the ground (d.b.h.) of each gap maker (trees having a d.b.h. of at least 20.0 cm) (Runkle 1992). A single-tree gap was formed by the death of one gap maker, whereas a multiple-tree gap was created by the death of two or more gap-maker trees. In multiple-tree gaps the main gap maker was the tree with the largest d.b.h. For each gap, the mode of gap origin was determined as tip-up, basal shear, standing dead (snag), limb dead or broken, or due to wet soils (Weiskittel and Hix 2003). The longest axis (A major), along its perpendicular axis (A minor), were measured and used to estimate gap size (Runkle 1992).

Within each gap, the probable replacement tree(s) were determined. These trees were chosen because of their location, height, and health (Runkle 1992). Along with the species of each tree, its height was measured with the aid of a telescoping height pole or a clinometer.

DATA ANALYSES

For each stand, the proportion of land area in canopy gaps was calculated as the proportion of the total transect distance in gaps divided by the total length of transect lines (Runkle 1992). Gap size was determined by using the area formula for an ellipse: $\text{area} = [3.14 \times (A \text{ major} \times A \text{ minor})/4]$. Eccentricity of gap shape was calculated by dividing A major by A minor; a circular shape occurs with a value of one and a value greater than one indicates an elliptical shape (Battles and others 1996).

Proportion of land area in canopy gaps, gap size, d.b.h. of gap-maker trees, and eccentricity of gap shape between the old-growth stand and the second-growth stands were analyzed using t-tests. Mode of gap origin by stand type data was analyzed using the Chi-square method of contingency analysis for categorical data. These analyses used only the 2008 data, and were accomplished using MINITAB® version 15 statistical software (Minitab Inc., State College, PA).

RESULTS

The old-growth stand had a significantly higher ($P = 0.025$) proportion of land area in canopy gaps than did the second-growth stand (9.3 percent vs. 3.7 percent, respectively). A total of 1,958 m of transect was sampled in the old-growth stand, and 17 gaps were encountered in 2008. In the second-growth stand, 728 m of transect was sampled and three gaps were encountered.

Only one additional canopy gap was found on the same transect lines in 2009. This single-tree gap occurred in the old-growth stand and had an area of 148 m².

Most (66.7 percent) of the gaps in the second-growth stand were small (<100.0 m²), whereas half of the gaps in the old-growth stand were large (≥ 100.00 m²) (Fig. 1). The mean canopy gap size in the old-growth stand was larger (145.6 ± 94.0 m²), but it was not different ($P = 0.793$) from the mean size in the second-growth stand (126.8 ± 101.6 m²) (Table 1).

All gaps had eccentricity values greater than one, indicating elliptical shapes. The mean eccentricity value for canopy gap shape was not significantly different ($P = 0.103$) between the two stands.

In the old-growth stand, multiple-tree gaps were 52.9 percent of the canopy gaps, whereas 66.7 percent of the canopy gaps in the second-growth were multiple-tree gaps. Tip-ups and snags were the dominant modes that originated gaps for both the old-growth and second-growth stands (Table 2). No difference in the observed vs. expected frequencies of mode of gap origin were detected between the two stands ($P = 0.130$).

Mean gap-maker size was larger ($P = 0.081$) in the old-growth stand (52.3 cm d.b.h.) than in the second-growth stand (45.8 cm d.b.h.) (Table 3). Sugar maple and American beech were the dominant gap-maker species in the old-growth stand, whereas northern red oak caused most of the gaps in the second-growth stand (Fig. 2).

Sugar maple will be the most common probable replacement tree species in canopy gaps in both the old-growth and second-growth stands. This species represented 83 percent and 54 percent, respectively, of the dominant stems in canopy gaps in the second-growth and old-growth stands. In the old-growth stand, American beech assumed this role in 32 percent of the canopy gaps.

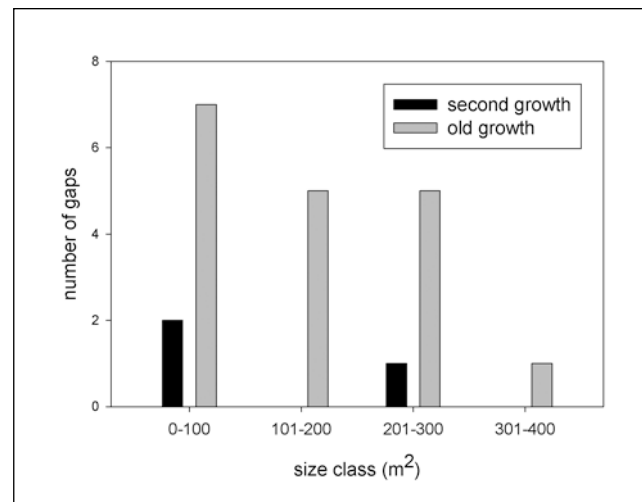


Figure 1.—Size distributions (m²) of canopy gaps in old-growth and second-growth stands at Crall Woods.

Table 1.—Gap size characteristics (m²) for the old-growth and second-growth stand gaps at Crall Woods.

	Old-growth	Second-growth
Mean ± standard deviation	145.6 ± 94.0	126.8 ± 101.6
Median	106.2	71.4
Maximum	302.0	244.0
Minimum	41.2	64.8

Table 2.—Number of gap makers by mode by which they originated canopy gaps for the old-growth and second-growth stands at Crall Woods.

	Old-growth	Second-growth
Tip-up	10	8
Basal shear	5	0
Standing dead (snag)	14	3
Limb dead or broken	2	0
Wet soil	2	0

Table 3.—Gap-maker size (cm d.b.h.) for old-growth and second-growth stands at Crall Woods.

	Old-growth	Second-growth
Mean $\pm$ standard deviation	53.4 $\pm$ 24.3	42.7 $\pm$ 13.3
Median	56.0	46.1
Maximum	113.9	67.2
Minimum	18	23.3

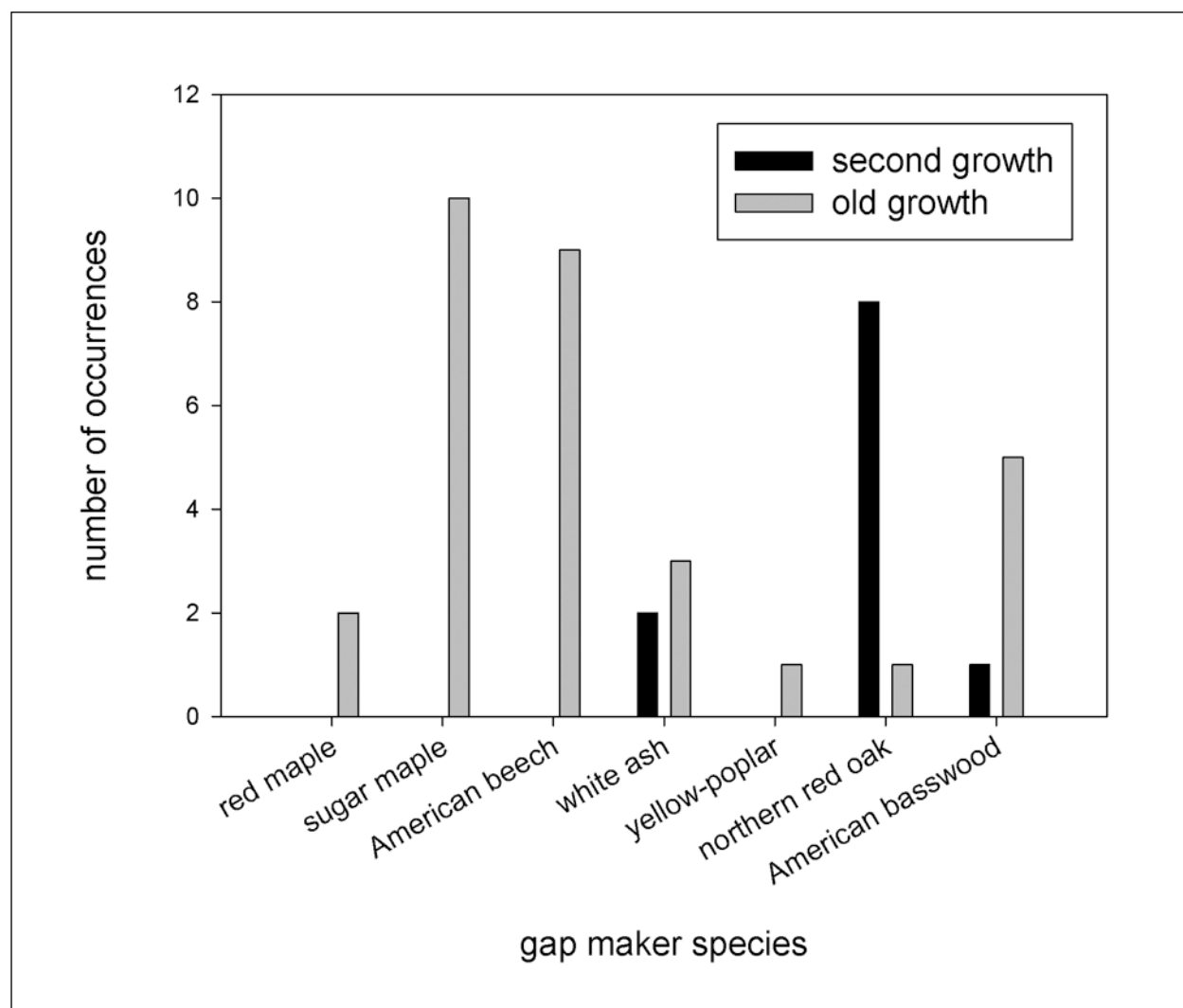


Figure 2.—Gap-maker species in the old-growth and second-growth stands at Crall Woods.

DISCUSSION

OLD-GROWTH STAND

The proportion of land area in canopy gaps in the old-growth stand was similar to those reported in other studies of old-growth Central Hardwood forests (Runkle 1982, Runkle 1990). Single- and multiple-tree canopy gap sizes in the old-growth stand at Crall Woods are typical of other old-growth beech-maple forests studied in Ohio (Runkle 1990, Forrester and Runkle 2000). However, the average gap size found in this study was smaller than the 280- to 375-m² range Lorimer (1989) found was typical of old-growth deciduous forests. Mean canopy gap size was also smaller than that for a nearby old-growth oak-beech-maple, wet-mesic stand in Johnson Woods in northeastern Ohio (Weiskittel and Hix 2003). Johnson Woods also had a larger gap fraction, possibly due to the deteriorating state of its very large, old (~400 years old) oaks (Weiskittel and Hix 2003). Gap shape for the old-growth stand at Crall Woods was more eccentric than in another study in an old-growth beech-maple stand in southwestern Ohio (Runkle 1990), where most gaps in Hueston Woods (65 percent) were approximately circular (Runkle 1990).

It is typical of old-growth forests for gaps to be created by snags and broken limbs resulting from single-tree deaths (Runkle 1982, Runkle 1990). Crall Woods follows this trend, since snags caused about half of the canopy gaps in the old-growth stand. This old-growth stand nonetheless differs from some other similar stands in the region since it has many tip-ups that have resulted in the formation of multiple-tree gaps. The above-mentioned study of Johnson Woods also found a high proportion of tip-ups (Weiskittel and Hix 2003). It is possible that the large crowns, somewhat poorly drained soils, and presence of surrounding agricultural land resulting in a 'hard' edge makes trees more susceptible to windthrow in the remaining old-growth stands of north-central Ohio.

The old-growth stand contained more canopy gaps than did the second-growth stand, but this finding was expected. Old-growth forests tend to have more canopy gaps than younger forests because such gaps have had more time to develop (Lorimer 1989).

SECOND-GROWTH STAND

Gap sizes between the two stands at Crall Woods were not different, possibly due to the similarity in size of the gap makers. Gap sizes in these stands, however, may have been overestimated. Battles and others (1996) found the more elongated the ellipse, the greater the overestimation. Gap shape was elliptical for all gaps in both the second-growth stand and the old-growth stand. The mean eccentricity value for all gaps was 2.5. When eccentricity values are 2 or less, the error is less than 10 percent (Battles and others 1996). Weiskittel and Hix (2003) also found most gaps were eccentric in shape.

The second-growth stand gaps were created primarily by tip-ups, which resulted in multiple-tree gaps. The high number of tip-ups resulting in single- and multiple-tree gaps in both the second- and old-growth stands is probably attributable to the somewhat poorly drained soils of Crall Woods, which cause the trees to be shallow-rooted and more susceptible to wind disturbance (Lorimer and Frelich 1994). In many forests, wind is a primary disturbance agent (Franklin and others 2007). The somewhat poorly drained soils and wind have interacted to create many of the canopy gaps in both stands in Crall Woods. These windthrown trees create pit and mound micro-topography, which affects species compositions and soil processes (Franklin and others 2007).

COMPOSITION OF CRALL WOODS

A recent study of Crall Woods was conducted to determine the current composition and structure of the old-growth and second-growth stands (Pinheiro and others 2008). This study found that the overstory of the old-growth stand is dominated by sugar maple, American beech, American basswood, and yellow-poplar (Pinheiro and others 2008). The overstory of the second-growth stand contains many of the same species, as well as northern red oak and various hickory species (Pinheiro and others 2008). Sugar maple and American beech were the most important gap-maker species in the old-growth stand, whereas northern red oak was the most important gap maker species in the second-growth stand. The significant differences in overstory species composition (Pinheiro 2008) help explain the differences in gap-maker species between the two stands. Despite the differences in overstory species, the understories of the two stands are both dominated by American beech and sugar maple (Pinheiro and others 2008). These two species were also identified as the most common probable replacement tree species in canopy gaps in both stands.

LIMITATIONS OF THE STUDY AND CONCLUSIONS

Our research results constitute a case study of the canopy gap characteristics of a very significant forest in north-central Ohio. Crall Woods presents an important opportunity to examine ecosystem processes of adjacent old-growth and mature second-growth stands, both possessing very similar environmental conditions (i.e., edaphic and physiographic properties). It should be noted that the somewhat poorly drained soil characteristic of Crall Woods may predispose trees to wind throw. Another limitation is the relatively small size of both stands, particularly the second-growth stand.

The information obtained from the study of the composition of Crall Woods, in conjunction with our results, suggests that the composition and structure of both stands are converging, largely in response to individual species' life-history traits and disturbances in the form of canopy gap formation. Despite the small size of Crall Woods, this forest ecosystem represents one of the best remaining examples of a relatively undisturbed old-growth forest in the region. Based upon our results and the conclusions of other studies (e.g., Runkle 1991), we suggest that forest managers may emulate canopy gap dynamics in similar beech-maple forest ecosystems of the Central Hardwood Forest using the selection regeneration method. We found that relatively small gaps occur naturally as trees die and (or) fall over. By harvesting small groups of mature trees (e.g., two to three) to both create new gaps and expand old gaps, managers may emulate the natural disturbance regime.

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COMPOSITION AND STRUCTURE OF RIPARIAN AREAS ALONG A LAND-USE GRADIENT IN AN AGRICULTURAL WATERSHED OF NORTHEASTERN OHIO

P. Charles Goebel, David M. Hix, and Heather L. Whitman¹

Abstract.—The restoration of riparian areas along many streams often proceeds with little existing information on the composition and structure of woody riparian vegetation. We examined the woody riparian vegetation in three subwatersheds of the Sugar Creek watershed in Ohio, each with different environmental characteristics (e.g., glacial history, physiography, soils, stream features) and surrounding land-use practices (e.g., forest, pasture, row crops, lawns). At the community level, we found that the overstory (stems ≥ 10.0 cm diameter at breast height [d.b.h.]) and the understory (stems 2.5–10.0 cm d.b.h.) species compositions were not significantly different among riparian areas with different environmental characteristics or surrounding land-use types (multiple response permutation procedure; $P = 0.365$ and $P = 0.325$, respectively). However, canonical correspondence analyses suggest relationships among individual overstory species, physiographic and stream variables, and surrounding land use. For example, northern catalpa (*Catalpa speciosa* [arder] Warder ex Engelm.) and American beech (*Fagus grandifolia* Ehrh.) were most frequently associated with riparian areas with surrounding wooded pasture land uses. In terms of structure, we found canopy openness to be significantly different among riparian areas adjacent to different land uses (Kruskal-Wallis, $P = 0.007$); we also found differences in understory stem density. These results suggest that environmental factors interacting with surrounding land-use types are associated with woody riparian vegetation, often resulting in riparian areas with simplified canopy structures and reduced complexity that may complicate riparian restoration efforts.

INTRODUCTION

It has long been recognized that riparian areas provide many important functions that are critical to overall watershed health (Naiman and others 1993). Although riparian areas often constitute a small proportion of a watershed, the ecosystem functions they provide illustrate their importance (Gregory and others 1991, Meyer and others 2007). Despite this importance, many riparian areas are highly disturbed, and in the Central Hardwood Forest region, many agricultural watersheds lack natural riparian forests (Naiman and others 1993, Tockner and Stanford 2002). Furthermore, frequent flooding and the linear structure of riparian forests lead to high levels of species richness, but also makes them highly susceptible to invasion by exotic, non-native species (Planty-Tabacchi and others 1996).

In an effort to better understand the factors that regulate riparian forest development and to help guide the management and restoration of riparian areas, many studies have focused on the influence of fluvial

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geomorphology and flooding on riparian communities' composition, structure, and diversity (e.g., Holmes and others 2005, Goebel and others 2006). These studies, however, have focused on relatively undisturbed forested settings, not areas where land-use practices may have significantly impacted the development of riparian plant communities. As many current riparian restoration efforts are occurring in highly disturbed watersheds, including agricultural watersheds of the Central Hardwood Forest region that were once forested, it is also important to understand plant community development in these disturbed riparian areas. It is likely that the extensive fragmentation and anthropogenic modifications made to landscapes in the Central Hardwood Forest region have generated complex disturbance gradients associated with surrounding land use; these disturbances have been shown to influence plant community development in other regions (Moffat and others 2004). Elucidating differences in plant community development in disturbed vs. less-disturbed watersheds is an important first step in developing appropriate management and restoration techniques for riparian areas in disturbed landscapes (Goebel and others 2003).

The objective of this study was to compare the vegetation among riparian areas along a disturbance gradient in an agricultural watershed of northeastern Ohio. Specifically, we assessed: 1) differences in riparian composition and structure associated with different land-use types; and 2) the relationships between environmental factors (including surrounding land use) and riparian overstory vegetation.

STUDY AREA

We focused our efforts in three representative subwatersheds of the Sugar Creek watershed in northeastern Ohio. The Upper and North Fork subwatersheds are located in the glaciated Erie Drift Plains Ecoregion (U.S. Environmental Protection Agency [EPA] 2007). The landscape of this Ecoregion, dominated by glacial landforms of Wisconsinan age, is characterized by gently sloping topography and poorly drained soils (Ohio EPA 2002). These areas are highly productive and are characterized by a mixture of both large- and small-scale dairy and row-crop farming. The South Fork subwatershed is located in the unglaciated Western Allegheny Plateau Ecoregion and is characterized by steeper slopes and higher stream gradients than the Upper and North Fork subwatersheds (Ohio EP 2002). Due to the limitations of steep terrain on row-crop agriculture, many small-scale dairy farms operate in this subwatershed.

The climatic conditions of all three subwatersheds are similar. Mean summer temperatures range from 20.6 to 21.1 °C, and mean winter temperatures range from -1.7 to -2.8 °C (Bureau and others 1984, Waters and Roth 1986, Seaholm and Graham 1997). Average annual precipitation ranges from 90 to 98.3 cm, with increasing levels as one moves south from the Upper subwatershed to the South Fork subwatershed (Bureau and others 1984, Waters and Roth 1986, Seaholm and Graham 1997). More than half of the precipitation falls between April and September in all three subwatersheds (Bureau and others 1984, Waters and Roth 1986, Seaholm and Graham 1997).

METHODS

FIELD AND LABORATORY ANALYSES

Within each subwatershed, we utilized 31 existing sample sites established as part of a long-term project examining the effects of land use on water quality and ecosystem function in headwater streams of the Sugar Creek watershed. Sites were carefully selected to represent the range of riparian and stream conditions associated with each subwatershed, with streams characterized as perennially flowing first- and second-order streams (Strahler 1957). As part of our long-term project, we have classified each site based upon the

dominant surrounding land-use types arrayed along the following developed-to-natural gradient: row crop, lawn, grassed pasture, wooded pasture, and wooded land-use types. The surrounding land use for each site was determined using digital aerial photographs and a geographic information system (GIS) following methods described in Moffat and others (2004). Specifically, within the GIS, the proportion of each land-use category was estimated within 1,000-m radius concentric circles centered on each site. In almost all cases, sites were selected that had homogenous surrounding land use. In those cases where there was more than one dominant surrounding land use, we characterized these 'mixed' land-use types based upon the two dominant land uses. In these instances, there was often a different land-use type on each side of the stream and the three 'mixed' categories encountered were row crop/lawn, row crop/grassed pasture, and lawn/wooded.

At each location we established two 100-m transects, one on each side of the stream, 2 m from the stream-bankfull stage. Along each transect at 20-m intervals, we sampled woody vegetation using the point-centered quarter method (Bonham 1989) from June-August 2008. This method has been shown to be an effective method to sample narrow riparian areas in other regions (e.g., Palik and others 1998). At each sample point, we divided the area into four 90° quarters using the four cardinal directions (i.e., north, east, south, and west). Within each quarter, we recorded the species, size, and crown class (overtopped, intermediate, codominant, dominant) of the closest living overstory tree (stems ≥ 10.0 cm diameter at breast height [d.b.h.], as well as the distance from the center of the tree to the point. At the center of each point, we established a 1.78-m radius plot and sampled saplings (stems < 10.0 cm d.b.h. and ≥ 1.0 m tall) and recorded species and d.b.h. Nomenclature follows the PLANTS database (U. S. Department of Agriculture, Natural Resources Conservation Service 2009).

To characterize canopy closure, we collected hemispherical photographs using a Nikon Coolpix 8400 digital camera with a Nikon FC-E9 Fisheye lens mounted 1 m above the ground and 2 m to the right of every other woody vegetation sampling point, totaling six photographs per sampling site. In the laboratory, we utilized the software program WinSCanopy Pro 2006 (Regent Instruments, Inc., Ste-Foy, QUE) for digital image processing and determining percent canopy openness of each sample site (using a hemispherical threshold compensation to rectify a bright sun in the photograph, causing canopy to be classified as sky). We utilized the six estimates of canopy openness to calculate mean percent openness at each sampling site.

We utilized the U.S. Department of Agriculture Web soil survey to characterize the environmental characteristics of each sampling location. Specifically, we classified the parent material (glacial till or alluvium), general soil texture (sandy loam, sandy to silt loam, silt loam, or fine loamy to silt loam), and soil color (as an indication of drainage class) for each sampling location. To verify this information in the field, we collected a soil sample from each sampling location using a split tube sampler to a maximum depth of 30 cm, and compared texture and color with what the Web soil survey indicated, where possible depending on landowner consent.

In addition to vegetation sampling and basic soil description, we also characterized stream channel and stream valley characteristics at each sampling location. Specifically, we measured bankfull width (m) and depth (m), flood-prone area width (maximum bankfull depth divided by the bankfull width and multiplied by 2, then the width of the channel re-measured at this calculated depth), and percent stream slope. With these data, we calculated an entrenchment ratio (flood-prone area width divided by bankfull width) to determine the vertical containment of a stream, as well as the width-to-depth ratio (bankfull width divided by bankfull depth), which indicates the channel cross-section shape (Rosgen 1996).

DATA ANALYSES

Using the overstory data, we first calculated importance values (IV; sum of relative density, relative dominance as expressed by basal area, and relative frequency divided by 3) for each species at each sampling location following methods described in Bonham (1989). To determine whether there was a difference in riparian overstory composition (as expressed by IVs) associated with different surrounding land-use types, we used a multi-response permutation procedure (MRPP) in PC-ORD (Version 5.0 MjM Software, Gleneden Beach, OR). MRPP is a non-parametric test, which requires neither normally distributed data nor homogeneity variance (McCune and Grace 2002). With MRPP, we used Sorenson's distance and a weighting factor, and made pairwise comparisons.

To test for differences in overstory structure among riparian areas with different surrounding land use, we calculated mean (± 1 standard deviation) dbh and mean canopy openness (± 1 standard deviation) for each type using MINITAB®, Version 15.1.2 (Minitab Inc., State College, PA). Finally, we used a non-parametric Kruskal-Wallis test in MINITAB Version 15.1.2 to examine whether there were significant differences in overstory and understory composition and structure, as well as in environmental variables among riparian areas.

Using the understory data, we calculated species richness (the number of species present), number of saplings per hectare, and relative abundance by sapling species among riparian areas. To determine whether there was a difference in understory composition associated with the different riparian areas, we used MRPP in PC-ORD Version 5.0 (MjM Software, Gleneden Beach, OR). We used the species abundance with a weighting factor and Sorenson's distance, and made pairwise comparisons in PC-ORD. The row crop/grassed pasture land-use type was deleted from this analysis because saplings occurred at only one sampling location. Because MRPP is used for testing for differences among two or more groups, we were not able to use this statistical analysis for the row crop/grassed pasture riparian type (McCune and Grace 2002).

To examine the relationship between overstory species' IVs and environmental factors, we used canonical correspondence analysis (CCA) using CANOCO software Version 4.53 (Biometris-Plant Research International, Wageningen, The Netherlands). Included in the analysis were three major groups of environmental factors: surrounding land use (each land-use type as a 'dummy' variable), soil characteristics (soil parent material and soil texture), and stream characteristics (entrenchment ratio, width-to-depth ratio, and stream channel gradient). While CCA is a direct gradient analysis that constrains the distribution of samples in ordination space based upon the 'best' fit between the environmental factors and species composition, it does require that the environmental factors included in the analysis be representative of the underlying gradients that structure the plant community.

RESULTS

STREAM, PHYSIOGRAPHIC, AND SOIL CHARACTERISTICS

We found no significant differences in stream characteristics, including entrenchment ratio ($P = 0.3550$), width-to-depth ratio ($P = 0.1500$), and stream gradient ($P = 0.5220$), among riparian types (Table 1). The soil profiles of most riparian types were formed in alluvium (about 81 percent), while the remainder developed in glacial till parent material. Soil texture for all riparian type soils was predominantly (about 90 percent) silt loam, while the remaining soils consisted of fine loamy to silt loam, and sandy loam texture.

Table 1.—Mean (± 1 standard deviation) entrenchment ratio, width-to-depth ratio, and gradient for riparian areas classified by surrounding land use in the Sugar Creek watershed, northeastern Ohio.

Adjacent Land Use	Stream Characteristics		
	Entrenchment Ratio	Width-to-Depth Ratio	Stream Gradient (%)
Row crop	1.31 (0.27)	9.20 (3.21)	1.38 (0.75)
Lawn	2.58 (1.30)	7.98 (6.44)	1.17 (0.29)
Grassed pasture	3.10 (1.83)	10.70 (8.87)	2.21 (1.60)
Wooded pasture	4.07 (3.89)	7.91 (5.97)	3.25 (2.47)
Wooded	1.86 (1.11)	16.27 (5.00)	1.17 (0.29)
Row crop/lawn	1.86 (0.64)	10.61 (6.06)	0.75 (0.35)
Grassed pasture/row crop	3.65 (3.82)	4.60 (0.74)	1.50 (0.58)
Lawn/wooded	2.53 (1.70)	5.85 (1.74)	1.50 (0.71)

OVERSTORY COMPOSITION AND STRUCTURE

We sampled 43 overstory species in the riparian areas of the Sugar Creek watershed; a little more than a third (37 percent) of these species occurred frequently (at least 10 percent of the sample plots; see Appendix for complete list of species). The MRPP results indicate no significant differences in overstory composition among riparian areas when classified according to their surrounding land-use type ($T = -0.272$; $P = 0.3650$). The most ubiquitous overstory species was black cherry (*Prunus serotina* Ehrh.) with the highest IV in six of the eight surrounding land-use types (Table 2). Other common species included black walnut (*Juglans nigra* L.), red maple (*Acer rubrum* L.), and green ash (*Fraxinus pennsylvanica* Marsh.). Red maple had the highest IV in the riparian areas associated with the row crop land-use type, black cherry in the lawn land-use type, and both white ash (*F. americana* L.) and black cherry in the grassed pasture land-use type (Table 2). At the opposite end of the disturbance gradient, riparian areas associated with the wooded pasture land-use type, northern catalpa (*Catalpa speciosa* [Warder] Warder ex Engelm.), American beech (*Fagus grandifolia* Ehrh.), and black walnut were the dominant species. In contrast, riparian areas associated with the wooded land-use type were characterized by sugar maple (*Acer saccharum* Marsh.), green ash, and black cherry (Table 2). Other species dominated the mixed land-use types (Table 2).

Table 2.—Mean importance value of most common (those occurring on at least 10 percent of the sample plots) riparian overstory (stems ≥ 10 cm diameter at breast height) species for riparian areas classified by surrounding land use in the Sugar Creek watershed, northeastern Ohio.

Species Name	Row crop	Lawn	Grassed pasture	Wooded pasture	Wooded	Row crop/ lawn	Grassed pasture/ row crop	Lawn/ wooded
<i>Acer negundo</i>	-	2.1	-	-	-	18.8	-	2.5
<i>A. rubrum</i>	14.0	-	2.2	5.4	10.1	-	-	12.6
<i>A. saccharinum</i>	-	2.5	-	-	-	-	-	8.9
<i>A. saccharum</i>	-	14.8	-	-	22.7	-	-	-
<i>Carya ovata</i>	-	5.1	6.2	5.4	-	-	14.0	-
<i>Catalpa speciosa</i>	-	-	-	19.5	-	6.2	9.4	-
<i>Cercis canadensis</i>	-	11.8	-	-	1.2	-	-	-
<i>Fagus grandifolia</i>	-	-	1.1	25.3	5.0	-	-	-
<i>Fraxinus americana</i>	2.8	16.7	10.6	-	-	-	-	-
<i>F. pennsylvanica</i>	4.3	-	3.5	9.7	25.8	-	-	4.1
<i>Juglans nigra</i>	1.1	-	2.4	20.8	1.2	29.6	-	18.7
<i>Morus alba</i>	1.5	1.7	-	-	-	-	-	5.6
<i>Morus rubra</i>	-	-	-	-	1.3	-	-	7.1
<i>Prunus serotina</i>	10.5	35.2	17.7	2.6	39.6	31.6	3.0	13.8
<i>Quercus rubra</i>	-	3.1	-	6.1	-	4.2	-	-
<i>Salix nigra</i>	-	-	7.9	-	-	3.1	-	-

We detected no significant differences in mean tree d.b.h. ($P = 0.0640$) and found that canopy openness differed significantly ($P = 0.0070$) among surrounding land-use types (Fig. 1). Specifically, we found that the riparian areas with the most open canopy were associated with the row crop/grassed pasture land-use type (83.42 percent $\pm$ 7.06 percent), while the wooded pasture and wooded land uses had mean openness values of 16.86 percent ($\pm$ 8.86 percent) and 13.56 percent ($\pm$ 6.63 percent), respectively (Fig. 1).

UNDERSTORY COMPOSITION AND STRUCTURE

We found no significant differences in understory composition among riparian areas ($T = -0.173$; $P = 0.3250$). Black cherry occurred at 4 sites, the most of any of the 19 understory species sampled, and was the most common species in both the row crop and wooded land-use types. In riparian areas associated with the lawn riparian type, American cranberrybush (*Viburnum opulus* L.) was the most common species in the understory layer. The most common species in riparian areas associated with the grassed pasture land-use type was willow (*Salix* spp.). In riparian areas associated with the wooded pasture land-use type, both American beech and black walnut were the most common species in the understory layer, while hawthorn (*Crataegus* spp. L.) and northern red oak (*Quercus rubra* L.) were the most common species in the row crop/lawn land-use type. In the lawn/wooded land-use type, American hornbeam (*Carpinus caroliniana* Walter) was the most common species.

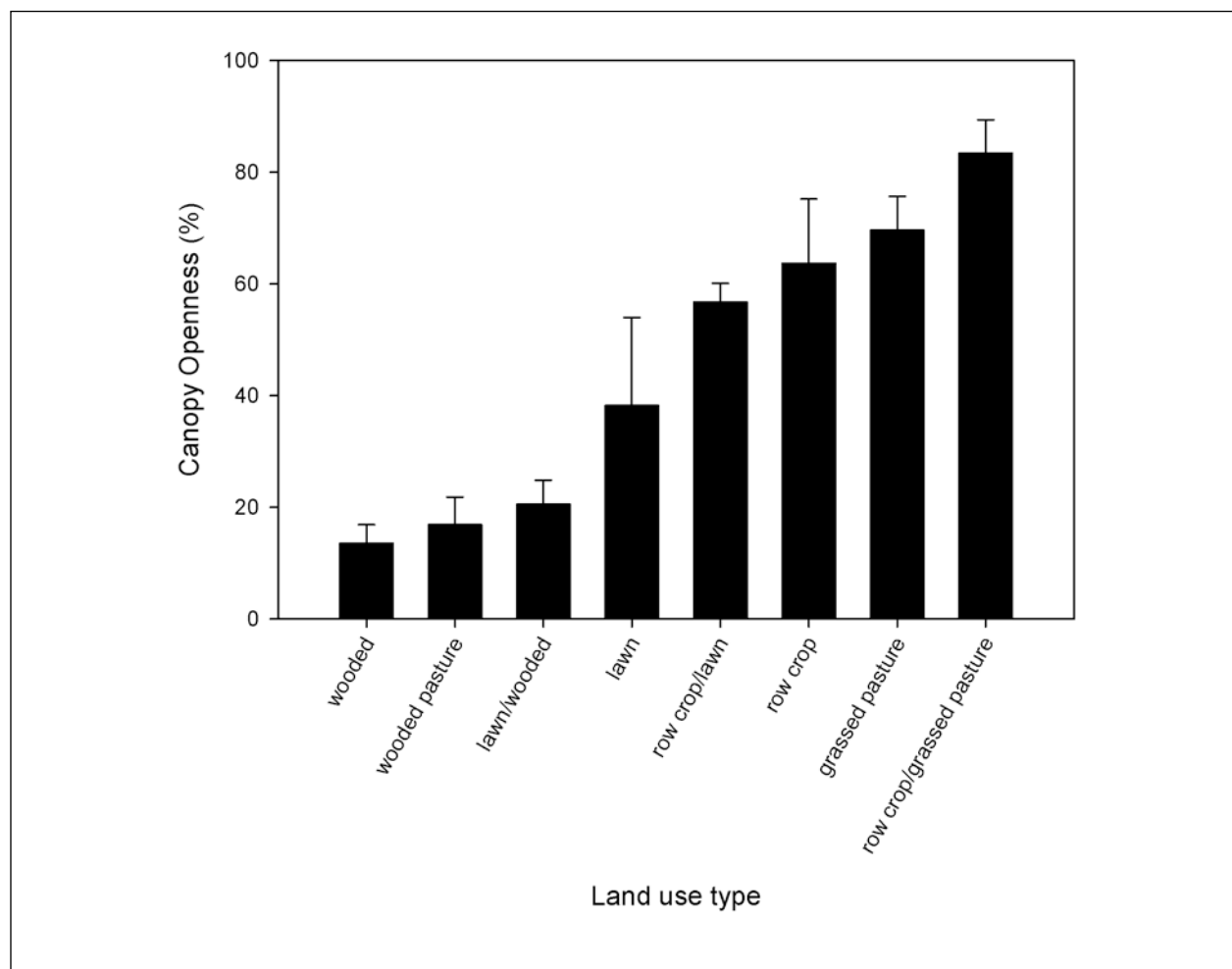


Figure 1.—Mean canopy openness (percent) ($\pm$ 1 standard deviation) for riparian areas classified by surrounding land uses in the Sugar Creek watershed, northeastern Ohio.

Understory density was significantly different ($P = 0.0020$) among riparian areas. Those surrounded by a wooded land-use type had the highest understory density ($774.19/\text{ha} \pm 2.33/\text{ha}$), while the row crop/lawn land-use type had the lowest density ($64.52/\text{ha} \pm 2.22/\text{ha}$). Both wooded and lawn/wooded land-use types had the highest understory species richness (six species in both riparian types) of all riparian areas.

VEGETATION-ENVIRONMENT RELATIONSHIPS

CCA results identified associations between overstory vegetation and environmental factors, explaining 18.4 percent of the variation in the overstory vegetation along the first two canonical axes. For the most part, overstory species were arrayed along the first axis (eigenvalue = 0.664) (Fig. 2). Of the variation in environmental factors, 9.6 percent was explained by CCA axis 1; CCA axis 2 explained an additional 8.8 percent. Along the first CCA axis, riparian areas associated with row crop and lawn-wooded mixed land use-types were positively associated with more deeply entrenched headwater streams in finer-textured soils associated with alluvium. Red maple, red mulberry (*Morus rubra* L.), and white mulberry (*M. alba* L.) all were associated with these land-use types (Fig. 2). Conversely, the row crop-lawn, grassed pasture, and lawn land-use types were characterized by less entrenched stream channels and more sandy soil textures developed in glacial parent material rather than alluvium. A variety of species were associated with these land-use types, including black willow (*Salix nigra* Marsh.), black cherry, and boxelder (*Acer negundo* L.). These land-use types also had the most diverse riparian overstory layers as we encountered more than 25 different overstory species in these riparian areas. We encountered most of these species infrequently and do not believe they are characteristic of mature riparian forests of the study area. Examples include eastern hemlock (*Tsuga canadensis* [L.] Carr.), eastern white pine (*Pinus strobus* L.), Callery pear (*Pyrus calleryana* Decne.), and a variety of apple species (*Malus* spp.). Along the second CCA axis, the mixed grassed pasture-row crop land-use type was associated with higher stream gradient systems and dominated largely by northern catalpa. Many of these riparian areas were located in the South Fork subwatershed within the unglaciated portion of the study area, where the topography is more undulating than in the Upper and North Fork subwatersheds. Riparian areas associated with surrounding wood land-use types were located near the origin of the ordination diagram (Fig. 2).

DISCUSSION

Although it is well understood that riparian forests provide many important ecological functions, riparian vegetation removal and degradation have resulted in poorly functioning riparian areas (Knutson and Klass 1998). In agricultural watersheds formerly dominated by forest, riparian loss has negatively affected stream water quality and aquatic food webs (Allan and others 1997). In response to these changes, a variety of organizations have been focused on restoring riparian forests across the Central Hardwood Forest region. In many instances, these efforts have been undertaken on a case-by-case basis without adequately prioritizing restoration efforts or considering the influences of surrounding land use on the composition and structure of riparian areas targeted for restoration.

Our results from headwater riparian areas in the Sugar Creek watershed, a typical agricultural watershed in the Central Hardwood Forest region, suggest that the composition and structure of riparian areas are associated with differences in environmental factors and surrounding land use. Other studies of riparian vegetation have also observed that surrounding land use is a critical factor along gradients of disturbance (Moffatt and others 2004). However, we should be clear that although the ordination analysis demonstrates a relationship between overstory composition and anthropogenic disturbance (as represented by surrounding

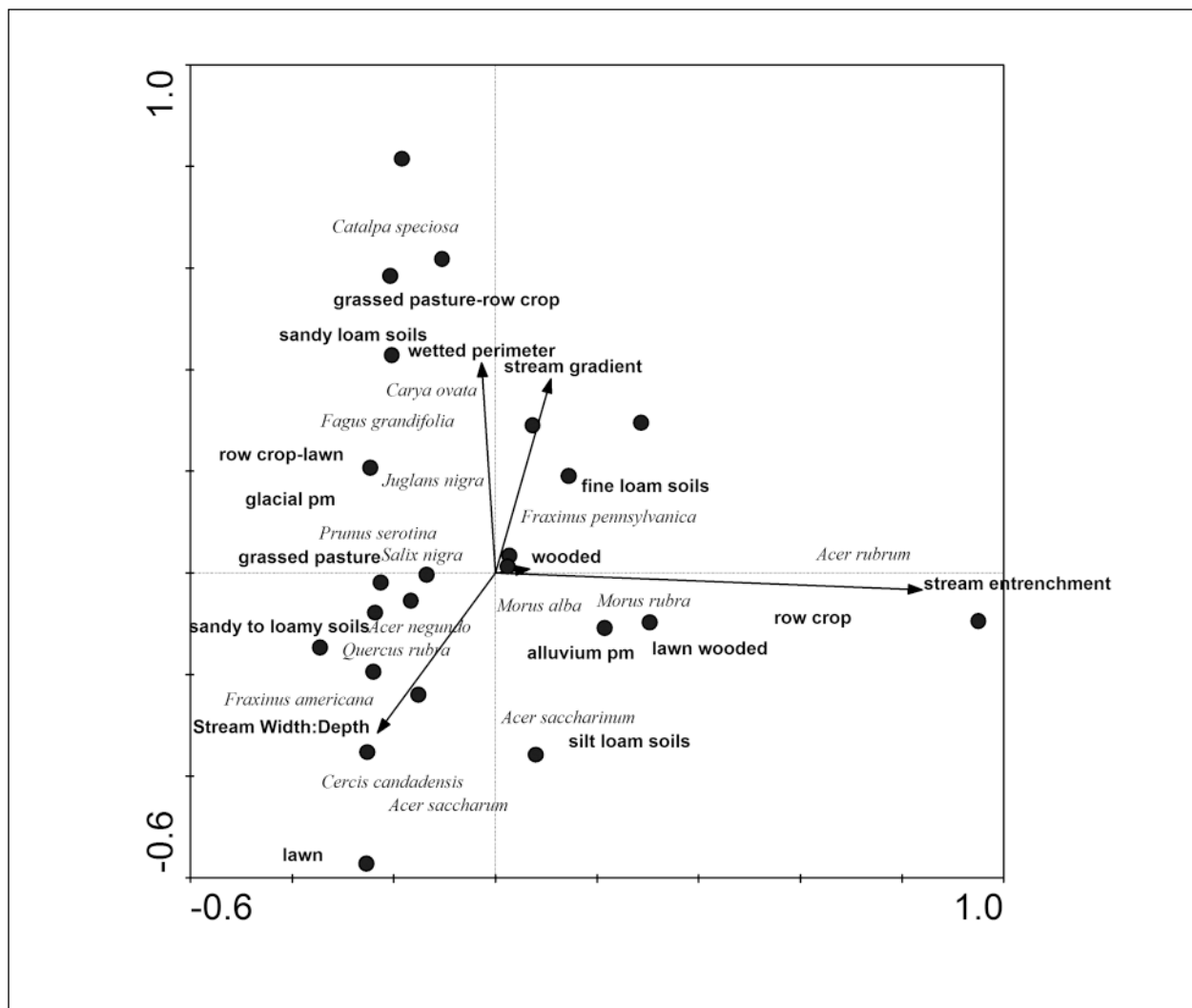


Figure 2.—Canonical correspondence analysis biplot relating mean riparian overstory species' importance values to environmental characteristics in the Sugar Creek watershed, northeastern Ohio. Only those species occurring frequently (those occurring on at least 10 percent of sample plots) are included. Continuous environmental variables are designated by vectors; nominal environmental variables are designated by label only following guidelines of Lepš and Šmilauer (2003).

land-use types) and environmental factors, additional research is needed to determine the strength and frequency of the relationships between disturbance and environmental factors across this landscape.

The patterns we observed in the composition of overstory species illustrate the potential influence of surrounding land use on riparian areas. Overstory species typically associated with disturbed sites, including eastern cottonwood, hawthorn, and northern catalpa, were associated with more disturbed surrounding land-use types (i.e., wooded pasture, row crop/lawn, and row crop/grassed pasture). This relationship is not surprising given the life-history traits of these species. Eastern cottonwood is a very shade-intolerant species which prefers wet soils, as does northern catalpa. Although not as intolerant to shade, hawthorn is characterized by an intermediate shade tolerance. As with the previously listed land-use types, species associated with the lawn/wooded land-use type are intermediate to intolerant of shade and thus grow well in open, disturbed areas. In addition, American sycamore prefers wet soils along streams and lakes. Although the

wooded riparian type was not associated with either axis 1 or 2 in the ordination, two sampling locations, i.e., U20A and U24C, were negatively associated with axis 2 and are both characterized as wooded riparian types. A few examples of species associated with these sites are American elm, blackgum, cherry spp., northern red oak, and willow spp., and range in shade tolerance from fairly tolerant to intolerant. Many of the common overstory and understory species characteristic of riparian areas in each land-use type (e.g., black cherry) are also shade-intolerant. The overstory of these riparian areas associated with wooded surrounding land-use types is similar to that of nearby reference or benchmark systems (Holmes and others 2005).

IMPLICATIONS FOR RESTORATION

The Sugar Creek watershed is a highly disturbed agricultural watershed with poor water quality due to land-use practices and poorly functioning riparian areas (Ohio EPA 2002). Because of the important functions riparian areas provide, restoration is necessary to reduce the impacts of land-use practices. Despite some differences in the importance of individual species, we found no differences in the stream characteristics or overall overstory composition among riparian areas associated with surrounding land-use types. We did, however, find differences in the structural attributes among riparian areas associated with different surrounding land-use types. These results suggest that enhancing the structure of those riparian areas surrounded by more disturbed land-use types may be an effective restoration treatment. Furthermore, the information gathered in this study can be used as baseline conditions by which to compare riparian vegetation composition and structure following restoration efforts in this watershed.

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APPENDIX. List of all overstory species in the Sugar Creek watershed, northeastern Ohio.

Scientific Name	Common Name
<i>Acer negundo</i> L.	boxelder
<i>Acer rubrum</i> L.	red maple
<i>Acer saccharinum</i> L.	silver maple
<i>Acer saccharum</i> Marsh.	sugar maple
<i>Aesculus hippocastanum</i> L.	horse chestnut
<i>Carpinus caroliniana</i> Walter	American hornbeam
<i>Carya cordiformis</i> (Wangenh.) K. Koch	bitternut hickory
<i>Carya ovata</i> (Mill.) K. Koch	shagbark hickory
<i>Catalpa speciosa</i> (Warder) Warder ex Engelm.	northern catalpa
<i>Cercis canadensis</i> L.	eastern redbud
<i>Crataegus</i> spp. L.	hawthorn
<i>Fagus grandifolia</i> Ehrh.	American beech
<i>Fraxinus americana</i> L.	white ash
<i>Fraxinus pennsylvanica</i> Marsh.	green ash
<i>Gleditsia triacanthos</i> L.	honeylocust
<i>Juglans nigra</i> L.	black walnut
<i>Liquidambar styraciflua</i> L.	sweetgum
<i>Liriodendron tulipifera</i> L.	yellow-poplar
<i>Malus</i> spp. Mill.	apple
<i>Maclura pomifera</i> (Raf.) C.K. Schneid.	osage orange
<i>Morus alba</i> L.	white mulberry
<i>Morus rubra</i> L.	red mulberry
<i>Nyssa sylvatica</i> Marsh.	blackgum
<i>Ostrya virginiana</i> (Mill.) K. Koch	hophornbeam
<i>Pinus strobus</i> L.	eastern white pine
<i>Platanus occidentalis</i> L.	American sycamore
<i>Populus deltoides</i> Bartram ex Marsh.	eastern cottonwood
<i>Prunus serotina</i> Ehrh.	black cherry
<i>Prunus</i> spp. L.	cherry
<i>Pyrus calleryana</i> Decne.	callery pear
<i>Quercus bicolor</i> Willd.	swamp white oak
<i>Quercus muehlenbergii</i> Engelm.	chinkapin oak
<i>Quercus palustris</i> Münchh.	pin oak
<i>Quercus rubra</i> L.	northern red oak
<i>Robinia pseudoacacia</i> L.	black locust
<i>Sassafras albidum</i> (Nutt.) Nees	sassafras
<i>Salix</i> spp. L.	willow
<i>Salix nigra</i> Marsh.	black willow
<i>Salix × pendulina</i> Wender.	Wis. weeping willow
<i>Tilia americana</i> L.	American basswood
<i>Tsuga canadensis</i> (L.) Carrière	eastern hemlock
<i>Ulmus americana</i> L.	American elm
<i>Ulmus rubra</i> Muhl.	slippery elm

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

LANDFORM AND TERRAIN SHAPE INDICES ARE RELATED TO OAK SITE INDEX IN THE MISSOURI OZARKS

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Abstract.—In the Southern Appalachians, metrics for quantifying the geometric shape of the land surface (terrain shape index or “tsi”) and of the landform (land form index or “lfi”) were developed and found to be correlated to yellow-poplar site index. However, the utility of these metrics for predicting site index for oaks in the Ozark Highlands has not been evaluated. We examined the relationship between black oak site index and the tsi and lfi with other measures of terrain, including the slope position and the (transformed) slope-aspect using data collected in 120 0.12-ac experimental units at the Sinkin Experimental Forest in southeastern Missouri. The tsi and lfi were each significant ($P < 0.05$) parameters in models predicting site index. Concave land forms had greater tsi and lfi values and higher site index values than did convex land forms. The AICC scores indicated that the best models included parameters for slope position and aspect along with the tsi, lfi, or both. Models with the tsi were statistically indistinguishable from models that included the lfi. Overall, the tsi and lfi each appear to be useful for quantifying terrain shape and its influence on the productivity of upland hardwood stands in the Ozark Highlands.

INTRODUCTION

Site index remains the most commonly used measure of site quality by practicing foresters. In the Central Hardwood Region, it is used to determine species suitable for management and to predict oak regeneration potential and stand development patterns (Johnson and others 2002). Volume equations often include site index along with diameter to estimate stand volume (Hahn and Hansen 1991). Despite its utility and common use, site index cannot be determined unless suitable trees are present in the stand. Suitable site index trees are those that have remained as canopy dominants or codominants; are relatively free of disease, pests, or other health problems; and, ideally, are growing in even-age stands that are approximately the index age (McQuilkin and Rogers 1978). Field estimates of site index are difficult to make when adequate site index trees cannot be found because of past disturbances or past pest or disease problems, or because the age of the trees in the stand greatly exceeds those used to develop site index curves (Monserud 1984, Berguson and others 1994). To deal with these situations, equations have been developed to estimate site index from soil, topographic, and other site factors. Although some of these equations can account for 70-85 percent of the observed variation in site index, these equations can sometimes be difficult to apply because they require detailed site-specific information about soil characteristics (Johnson and others 2002).

In the eastern United States, variation in site index, species composition, and forest productivity has long been attributed to changes in topography (Trimble and Weitzman 1956; McNab 1989, 1993; Johnson and others

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2002). Slope aspect also affects forest productivity due to its influence on solar radiation and microclimate, but aspect alone does not completely explain the variability in productivity across a landscape (Trimble and Weitzman 1956, Trimble 1964, Rosenberg and others 1983, Fekedulegn and others 2003). Soil scientists use slope shape in addition to slope position to infer water movement and subsequently to map the kinds of soils formed in different locations on the landscape (Ruhe 1975, Hudson 1990). Taking into account the influence of slope shape on water movement, forest scientists have developed methods to correlate those relationships with measures of forest productivity such as site index (Johnson and others 2002).

Despite the documented relationships between slope shape and forest productivity, descriptions of slope shape have largely remained qualitative. Land units are allocated into a slope position class such as summit, shoulder, backslope, footslope, or toeslope, defined by their position along a hillslope and by the degree of concavity or convexity they exhibit in the down-slope direction. Slope positions are sometimes assigned an additional modifier to describe the degree of concavity or convexity they exhibit along the contour (e.g., headslopes, noseslopes, or sideslopes) (Schoenenberger and others 2002). In the southern Appalachians, McNab (1989, 1993) developed methods for quantifying the geometric shape of the land surface with the land form index (lfi) and the terrain shape index (tsi). These metrics are determined in the field by making a few measurements with a clinometer and do not require the application of geographic information system (GIS) terrain modeling techniques. Each of these indices was found to be correlated to yellow-poplar (*Liriodendron tulipifera* L.) site index in the southern Appalachians and therefore useful for predicting site index in stands where suitable yellow-poplar site index trees were lacking. Because suitable site index trees frequently are not available in oak stands in the Missouri Ozarks due to high-grading or other past land use, a simple means for predicting site index would be helpful for forest management planning in this region. McNab's (1989, 1993) success correlating simple terrain metrics with yellow-poplar site index suggested that this approach may be applicable to the Missouri Ozarks. However, these metrics have not been evaluated in the drier oak-hickory (*Quercus* L.-*Carya* Nutt.) forests, where relationships between site quality and terrain characteristics potentially differ from those of the southern Appalachians. Our objectives were to examine relationships between the tsi and lfi, and black oak site index, in the Missouri Ozarks. In this region, site index for most oak species is commonly converted to and expressed on a black oak basis for comparing stand site quality.

STUDY SITE

The study was conducted in the Sinkin Experimental Forest located about 30 miles south of Salem, MO, within the Current River Hills Subsection of the Ozark Highlands (Keys and others 1995). This region has narrow ridges and steep side slopes with a relief of around 200 ft and soils formed from Ordovician and Cambrian dolomite and sandstone (Nigh and Schroeder 2002). Data accessible through the Missouri Cooperative Soil Survey (www.soilsurvey.org) indicated that the soils on the ridge tops and upper hillsides developed in parent materials derived from the Roubidoux and upper Gasconade formations and are highly weathered, droughty, and strongly acid, and contain a high percentage of rock fragments. Common soil series on ridge tops and upper slopes were Coulstone and Clarksville (both Typic Paleudults), Hobson (Oxyaquic Fragiudalts), Lebanon (Typic Fragiudults), and Nixa (Glossic Fragiudults). Soils on the lower hillsides developed in parent materials that were derived from the lower Gasconade or Eminence formations generally are less weathered. Some of these soils are influenced by the underlying dolomite and consequently contain clayey residuum that has a greater cation exchange capacity, fewer rock fragments, and a greater water holding capacity. The soil series on lower hillsides were Clarksville and Doniphan (both Typic Paleudults) and Moko (Lithic Hapludolls). Outcrops of dolomite occur with some of the Moko soils.

We examined the relationships between topographic variables and site index in the Missouri Ozarks as part of a comprehensive oak regeneration study named the “Regional Oak Study” (ROS). The ROS is a regional, multi-disciplinary research project being implemented by both the Northern and Southern Research Stations of the U.S. Forest Service at sites located in North Carolina, Tennessee, and Missouri (Greenberg and others 2007). The study consists of 20 rectangular 12-acre treatment units that were each established across a moisture gradient from ridge top to footslope. Each treatment unit has six circular, 0.12-acre sampling plots positioned evenly within the 20 12-acre treatment units, providing a total of 120 sample locations. The treatment units are in oak-pine and mixed oak forests having an average age of 82 years for the overstory. Inventories in these plots revealed that oaks were the dominant species, contributing 59 percent of the basal area. Of the oak species present, white oak (*Quercus alba* L.) contributes 22 percent of the basal area, black oak (*Q. velutina* Lam.) 21 percent, scarlet oak (*Q. coccinea* Muenchh.) 12 percent, and northern red oak (*Q. rubra* L.) 4 percent. Shortleaf pine (*Pinus echinata* Mill.) makes up 22 percent of the basal area. Other species by basal area include hickory species (*Carya* spp.) (7 percent), slippery elm (*Ulmus rubra* Muhl.) (2 percent), flowering dogwood (*Cornus florida* L.) (2 percent), blackgum (*Nyssa sylvatica* Marsh.) (2 percent), black walnut (*Juglans nigra* L.) (2 percent), and red maple (*Acer rubrum* L.) (1 percent).

METHODS

Measurements at each of the 120 sampling plots included site index, aspect, slope position, tsi, and lfi. Site index was determined using a representative dominant or co-dominant tree within each plot. Site index trees included red oaks (*Quercus* spp. L.; section *Lobatae*), white oaks (*Quercus* spp. L.; section *Quercus*), and shortleaf pine. The height of each site index tree was measured to the nearest foot using an Impulse™ 200 (Laser Technology Inc., Centennial, CO) laser. A core was removed at breast height using an increment borer and rings were counted with the aid of a hand lens. Site index in feet was calculated for each species using relationships developed by McQuilkin (1974, 1978) for oaks and Nash (1963) for shortleaf pine. Following the convention, all site index values were converted to black oak site index basis using the relationships developed by McQuilkin (1976).

Aspect was measured from plot center using a handheld compass and taken in the direction of the steepest down-slope direction. Slope position was also recorded in each plot and given a categorical value of shoulder, backslope, or footslope following the definitions originally proposed by Ruhe (1975) and now routinely used in soil mapping (Schoeneberger and others 2002). Briefly, shoulders occur on upper slopes and generally are convex in profile. Backslopes occur mid-slope and are nearly linear in profile but can vary in shape along the contour and include sideslopes (linear along the contour), noseslopes (concave along the contour), and headslopes (convex along the contour). Footslopes occur on lower slopes and generally are concave in profile.

We measured the tsi and lfi from the center of each plot using a hand-held clinometer. At each plot, measurements were taken starting in the direction of the aspect and then rotating clockwise in 45° increments for a total of eight slope measurements. For the tsi, the slope from the plot center to the plot perimeter (41 ft) measured parallel to the land surface was recorded. For lfi, the slope from the plot center to the horizon was measured. Slopes were recorded as a percent; down-slope measurements were recorded as negative values and up-slope measurements as positive values. All eight measurements were then averaged to determine the tsi or lfi value. McNab (1989, 1993) found that estimates of tsi or lfi almost always stabilized when measuring four directions and changed little when eight or more measurements were taken.

DATA ANALYSIS

We used an Information-Theoretic approach (Burnham and Anderson 1998) to compare the models predicting black oak site index (response variable) and the lfi, tsi, and other measures of terrain including slope position and aspect (explanatory variables). As per the Information-Theoretic approach, we developed a series of candidate models for estimating site index by including the lfi or tsi singly or in combination with the slope position and aspect. These models were used to evaluate the hypotheses that including the lfi or tsi improves predictions of site index compared to using only the traditional descriptive measures of slope position and/or slope aspect. Models were compared using Akaike's information criterion (or AIC). The $AIC = n\ln(RSS/n) + 2k$, where n is the sample size, $\ln$ is the natural log, RSS is the residual sums of squares, and k is the number of parameters in the model. Lower AIC scores indicate a better fit and models having AIC scores more than two units apart are generally considered significantly different from each other (Burnham and Anderson 1998). We applied the correction for small sample size (creating an AICC score) by adding $2k(k+1)/(n-k-1)$ to the AIC score as per the recommendations of Burnham and Anderson (1998).

We used the MIXED procedure in SAS[™] statistical software, version 9.1 (SAS Institute, Inc. Cary, NC) to estimate model parameters and to generate AICC scores for each model. The mixed procedure was used to accommodate models that included categorical and continuous data and to account for both fixed effects (i.e., terrain variables) and random effects (i.e., 12-acre treatment units). The model form was $Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \dots + \beta_n X_n$ where Y was the black oak site index in feet, β_0 was the intercept, and subsequent parameters were for the lfi or tsi, slope position, and/or slope aspect. Prior to analyses, the distribution of site index was examined; this variable was found to be normally distributed so there was no need for transformation. During analyses, the slope position was included as a categorical variable (i.e., shoulders, backslopes, and footslopes) and the slope aspect was included as a continuous variable after it was transformed to a linear value ranging from 0 (azimuth of 225 degrees) to 2 (azimuth of 45 degrees) following the model by Beers and others (1966) where the Transformed aspect = $\cos(45 - \text{Aspect}) + 1$.

We used the Information-Theoretic approach for model selection because the AICC score accounted for the number of parameters in models. Thus, the method discriminated against models having more parameters and thereby reduced the risk of overfitting the models. This approach offered an advantage over the stepwise regression procedure, where overfitting generally increases the apparent goodness of fit. One disadvantage of the method, however, is that typical measures of goodness of fit (e.g., R-values) are not provided. As a remedy, we also fit models using the glm procedure (SAS version 9.1). The glm procedure produced model parameters similar in magnitude to those produced by the mixed procedure (they were not exactly the same because glm cannot account for random effects) and provided R values as an additional estimate of goodness of fit.

RESULTS

Both the terrain shape index (tsi) and landform index (lfi) were related to the slope position with a few outliers (Fig. 1). Generally, the tsi and lfi were each positive and larger in magnitude on footslopes, indicating a more concave land surface shape, and were lower in magnitude or negative in value on backslopes and shoulders, indicating a linear or slightly convex shape. The lfi was generally greater in magnitude and more positive than the tsi. This relationship occurred largely because the lfi is determined by measuring the slope angle to the surrounding horizon, which usually appears above the surrounding ridge tops. Consequently, slope measurements made from below a ridge top are very likely to have large and positive values. There also were differences in the magnitude of the variation between the lfi and tsi. The lfi was most variable on backslopes while the tsi was most variable on footslopes.

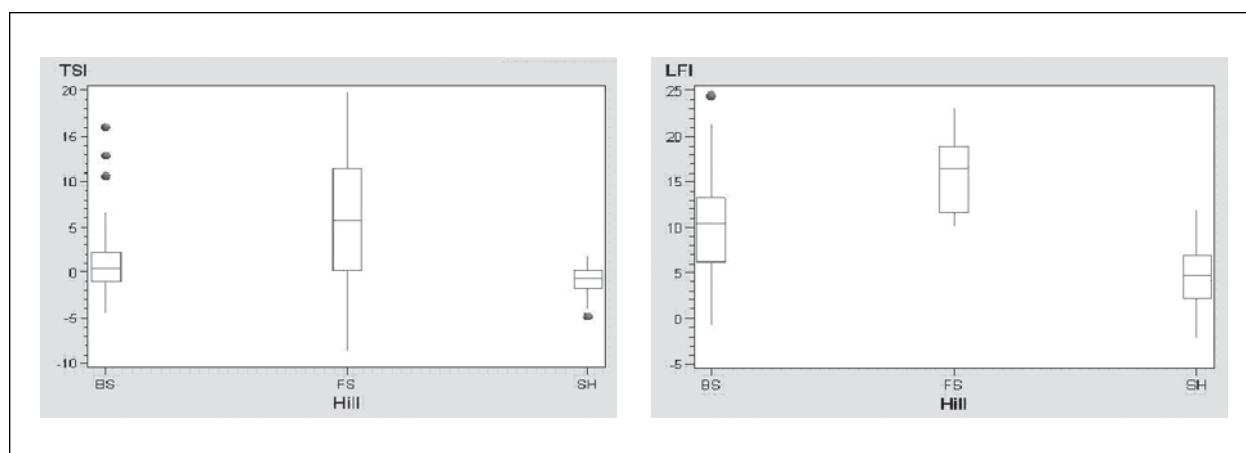


Figure 1.—Box-and-whisker diagrams displaying terrain shape index (tsi) and landform index (lfi) values by slope position. Note: “BS” denotes backslope, “FS” denotes footslope, and “SH” denotes shoulderslope. Both tsi and lfi values are in percent gradient.

The tsi and lfi were each significant ($P < 0.05$) parameters in models predicting site index (Table 1). The AICC scores indicated that the best models included parameters for the slope position and aspect along with the tsi (model 2), the lfi (model 3), or both the lfi and tsi together (model 1). However, the AICC scores also indicated that models including either the tsi or lfi along with slope position and aspect were statistically indistinguishable from each other. Furthermore, models including both the slope position and aspect in addition to the tsi or lfi were better than those that did not include slope position and aspect.

The parameters for models that included the slope position, aspect, and the tsi (model 2) or the lfi (model 3) shown in Table 2 were used to illustrate relationships between them and the black oak site (Figs. 2 and 3). These models indicated that northeast-facing slopes generally had greater site index values than southwest-facing slopes and that shoulder and backslopes had greater site index values than did footslopes. They also indicated that for a given slope position or aspect, increasing the tsi or the lfi (i.e., increasing the concavity) generally increased the site index estimate. However, the wide scatter in the actual data plotted in Figures 2 and 3 and low R values for the models (Table 1) indicated considerable variation in the site index data.

Table 1.—Site index models with effects for slope position (including shoulder, backslope, or footslope), Beers-transformed aspect (taspect), terrain shape index (tsi), and land form index (lfi).

Model ^a	AICC	Delta AICC ^b	R
1. Slopeposition* + taspect* + tsi + lfi	861	0	0.36
2. Slopeposition + taspect + tsi***	862	1	0.32
3. Slopeposition* + taspect** + lfi***	862	1	0.33
4. Slopeposition + taspect	867	6	0.22
5. Slopeposition* + tsi**	867	6	0.29
6. Slopeposition* + lfi**	867	6	0.27
7. Taspect* + tsi**	874	13	0.25
8. Taspect* + lfi**	874	13	0.24
9. Tsj**	878	17	0.20
10. Lfi**	880	19	0.17

^a Model form $Y = \beta_0 + B_1X_1 + B_2X_2 + \dots + B_nX_n$

^b Change in AICC score from the lowest. Lower AICC scores indicate better models and a change in AICC score greater than 2 indicates a model is statistically different from other models.

* effect significance $P \leq 0.10$

** effect significance $P \leq 0.05$

*** effect significance $P \leq 0.01$

Table 2.—Parameters for site index models 2 and 3 in Table 1 for estimating black oak site index in feet.

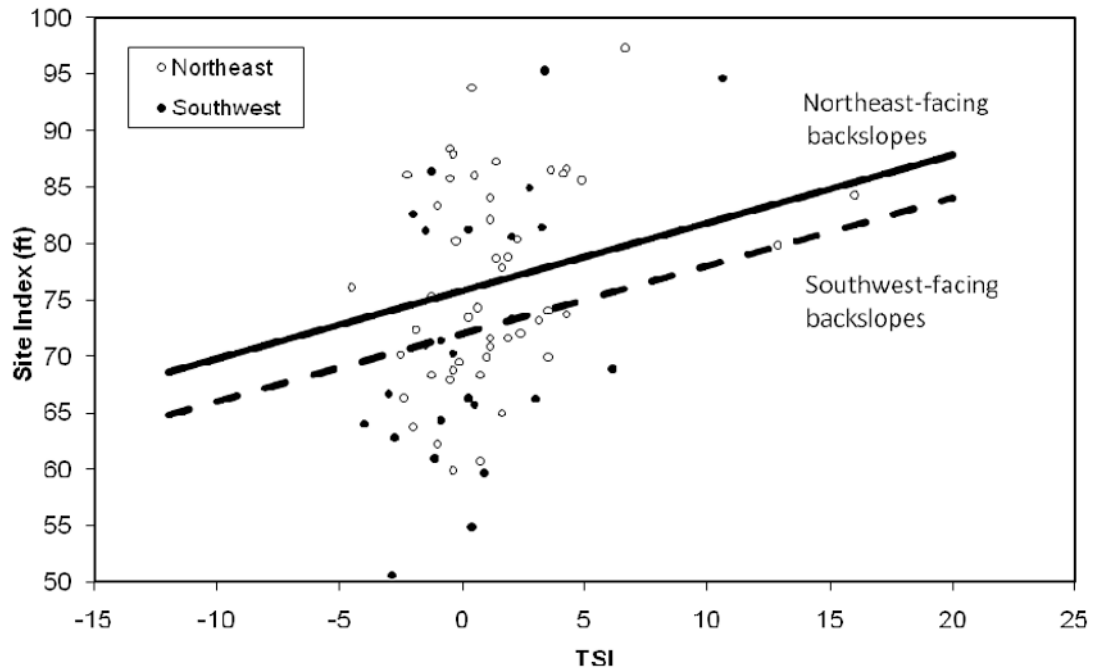
Effect	Estimate	Standard error	P-value
Model 2—TSI			
Intercept	70.60	2.35	<0.01
Slope position			
Backslope	1.41	1.92	0.47
Foothslope	-4.72	3.42	0.17
Shoulder	--	--	--
Transformed aspect	1.93	1.27	0.13
TSI	0.57	0.23	0.01
Model 3—LFI			
Intercept	67.40	2.60	<0.01
Slope position			
Backslope	-0.19	2.14	0.93
Foothslope	-7.02	3.82	0.07
Shoulder	--	--	--
Transformed aspect	2.27	1.28	0.08
LFI	0.50	0.20	0.01

DISCUSSION

Much has been written about the effects of slope shape on the height growth or site index of trees (Carmean 1967, Hannah 1968, Hartung and Lloyd 1969, Auchmoody and Smith 1979, McNab 1989). Most of these studies showed that height growth or site index is generally greater on lower slopes or in concave “cove” positions, where the soil’s ability to supply nutrients and particularly water is generally greater (White 1958, Fralish 1994). Despite the influence of terrain on site productivity, few have developed approaches for quantifying terrain shape to accompany other more descriptive assessments of topography for predicting site index. McNab (1989, 1993) developed a method for quantifying terrain shape by averaging the slopes in four or more directions from a plot center to its perimeter parallel to the land surface (tsi) or to the horizon (lfi). In the mesophytic forests of the southern Appalachians, yellow-poplar site index was significantly correlated with tsi (McNab 1989) and lfi (McNab 1993). In the Missouri Ozarks, we also found significant relationships between black oak site index and tsi and lfi. Our three most significant models for explaining variation in black oak site index were parameters for slope position; Beers-transformed aspect; and tsi, lfi, or both. These three models were statistically indistinguishable from each other, suggesting all were equally useful for predicting site index. However, foresters are more likely to measure a single index and our findings suggest that tsi and lfi work equally well. Slope position and aspect were also important parameters in these models and our analysis suggested that both should be included with either tsi or lfi to render the best estimates of site index in similar landscapes of the Missouri Ozarks.

Despite the statistical significance of the parameters, the R values for our data were generally quite low, indicating considerable variation not accounted for with the models that we developed. McNab (1989, 1993) generally found stronger relationships between the yellow-poplar site index and the lfi (R between 0.45 and 0.65) and the tsi (R=0.71) in the southern Appalachians. This difference in R values may be due to the wider range in site index values and in the tsi and lfi values encountered in the Appalachian region compared to the

a) slope aspect



b) slope position

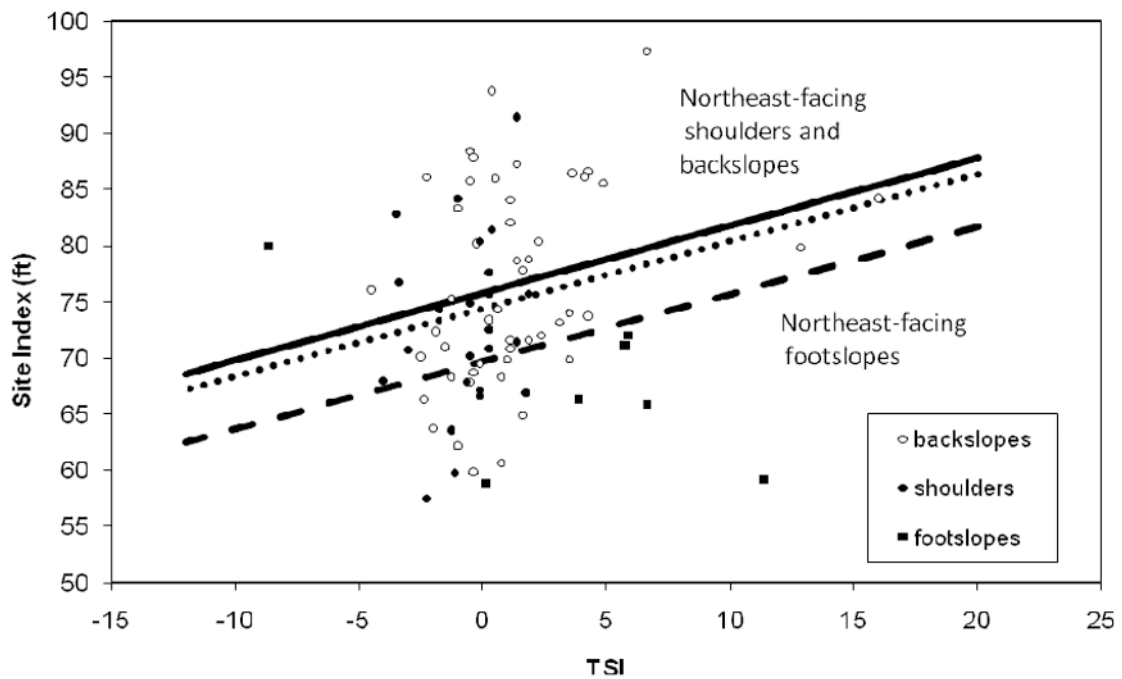
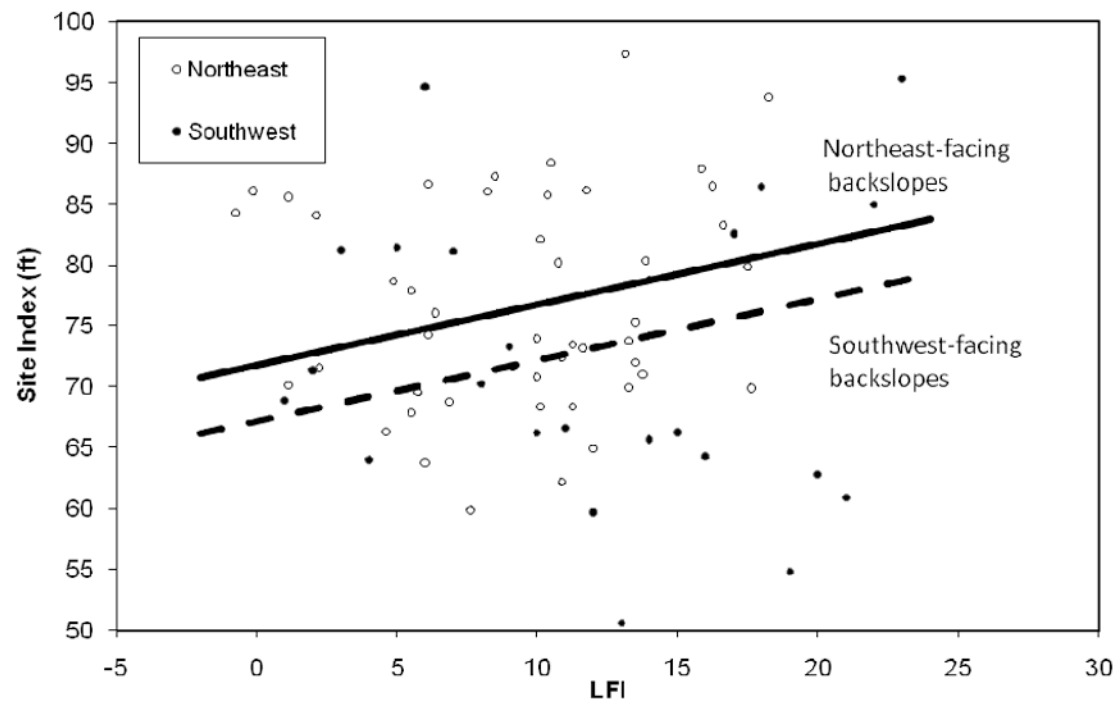


Figure 2.—Black oak site index and terrain shape index (tsi) by a) slope aspect, and b) slope position.

a) slope aspect



b) slope position

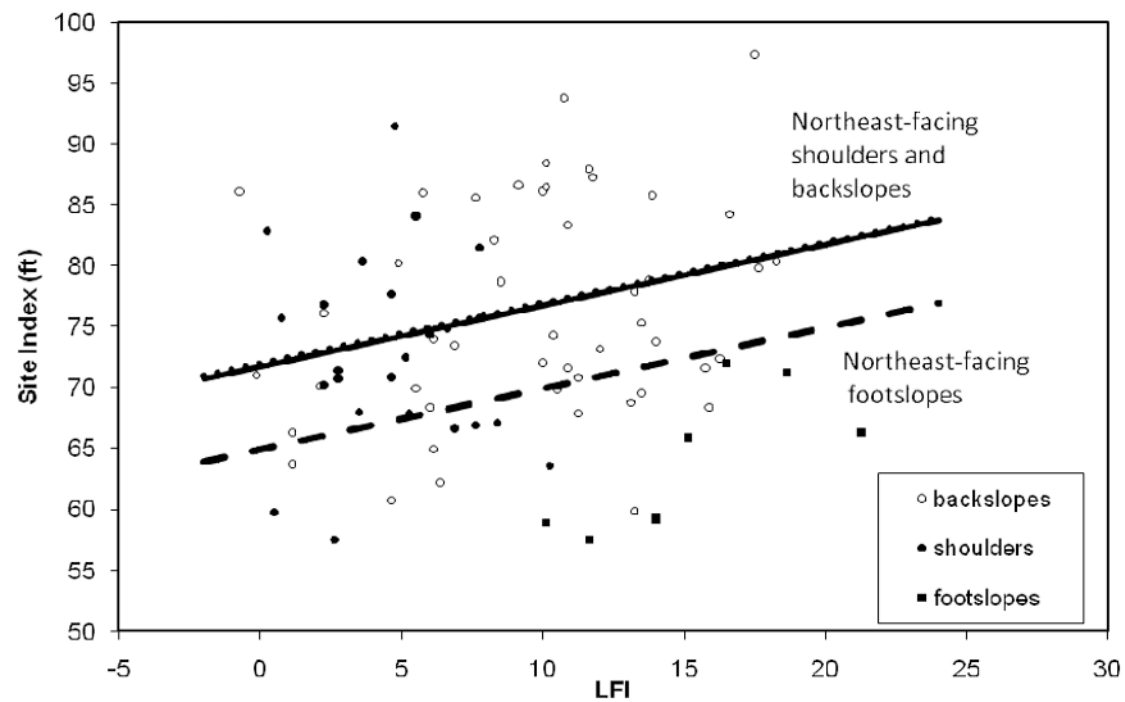


Figure 3.—Black oak site index and land form index (lfi) by a) slope aspect, and b) slope position.

Missouri Ozarks, where the terrain is gentler. In the Missouri Ozarks, greater correlations may have occurred had we accounted for variation in soil properties related to water and nutrient supply.

The models that we developed appear to have utility for estimating black oak site index in the Missouri Ozarks for stands lacking suitable site index trees. For each inventory plot, the slope position and aspect are recorded and the tsi or lfi is determined as described in the “Methods” section of this paper and applied to the models in Table 2. For example, for shoulder slopes having an azimuth of 83° (Beers transformed = 1.79) and a tsi = 2.1, the black oak site index = 75 feet. For backslopes having an azimuth of 180° (Beers transformed = 0.29) and a lfi = 10.7, the black oak site index = 73 feet.

Although the tsi and lfi are determined in the field, the increasing availability of high-resolution terrain information and GIS software has provided alternatives for estimating these and other metrics of slope shape. Some of these terrain metrics have been used along with other remotely-sensed data to estimate site index elsewhere in the Central Hardwood Region. For example, Iverson and others (1997) used GIS software to develop a moisture index model based on soil water-holding capacity, topographic shape, and slope-aspect for the Vinton Experimental Forest in southern Ohio. This moisture index was highly correlated ($R > 0.79$) to the oak site index, particularly with terrain models derived from fine-resolution (<1:24,000 scale) source data.

Our interest in predicting site index is not limited to the need for estimating potential merchantable volume or biomass production of a stand. In the Central Hardwood Region, site index is also important for identifying where future oak regeneration problems may occur. Further research is needed to determine whether measures of terrain shape such as the tsi or lfi do provide an additional metric for identifying where oak regeneration problems are likely to occur, particularly where suitable site index information is lacking.

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

SITE CHARACTERISTICS OF AMERICAN CHESTNUT, OAK, AND HICKORY WITNESS TREES ON THE MONONGAHELA NATIONAL FOREST, WEST VIRGINIA

Melissa Thomas-Van Gundy and Michael Strager¹

Abstract.—Early metes and bounds surveys of the area that would become the Monongahela National Forest in West Virginia were used to create a digital database of corner witness trees. With these electronic and geo-referenced data, physical characteristics were obtained to describe species' locations on the landscape. To characterize the physical environment, variables associated with the corner points were extracted from existing spatial datasets and derived from a digital elevation model. The terrain components that were assessed included topographic roughness, moisture index, aspect, and landform. Calculations were made for American chestnut, several oak species, and hickory species. Species-site associations differed by subsection. American chestnut, chestnut oak, and scarlet oaks, however, were generally found in areas of low moisture and at moderate elevations. White oak was more likely on areas of high moisture and valley floor landforms. Northern red oaks were more likely on S-SE facing benches with low moisture levels. While few differences between subsections were noted for these species, construction of predictive models should consider subsectional differences.

INTRODUCTION

Information is lacking for much of the eastern United States on historical forest conditions due to almost complete forest clearing at the turn of the 19th century. Descriptions and quantitative assessments of early forests are useful in restoration ecology and can come from many sources, such as deeds, traveler's accounts, and photographs. Although these descriptions represent a static point in time, they can be a source of information for the time of European settlement (Whitney and DeCant 2001).

The recording of witness trees in deeds as corner markers for parcel boundaries has often been used to describe European settlement-era vegetation (Abrams and Ruffner 1995; Black and Abrams 2001a, 2001b; Whitney and DeCant 2003; Rentch and Hicks 2005). Land transfers in most eastern forests followed survey methods called "metes and bounds," consisting normally of bearings, distances, and species of tree used to witness property corners. Several authors have discussed the possibility of bias in witness tree databases, mainly pertaining to rectangular land surveys. Nonetheless, the usefulness of the data has been shown to generally outweigh the potential biases (Bourdo 1956, Siccama 1971, Whitney 1994).

Previous witness tree analysis in the Monongahela National Forest (MNF) determined that pre-European settlement Northern Ridge and Valley Section forests were dominated by mixed oak (*Quercus* spp.),

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pine (*Pinus* spp.), American chestnut (*Castanea dentata* [Marsh.] Borkh.), and hickories (*Carya* spp.) on ridges while the valley floors were dominated by white oak (*Q. alba* L.), sugar maple (*Acer saccharum* Marsh.), pine, basswood (*Tilia* spp.), and hemlock (*Tsuga canadensis* [L.] Carrière) (Abrams and McCay 1996). In the Allegheny Mountains Section, beech (*Fagus grandifolia* Ehrh.)-hemlock-pine forests dominated mountain tops and hemlock-maple-birch (*Betula* spp.) forests were found on valley floors (Abrams and McCay 1996).

In the 1930s, staff of the MNF collected deeds and land grants of the earliest granted tracts of land within the MNF proclamation boundary. Dates of these deeds and land grants range from 1752 to 1910. Between 2005 and 2009, these data were converted to a spatial database in a geographic information system (GIS). With these geo-referenced data, physical characteristics associated with species locations can be obtained to describe where species were found on the landscape.

The goal of this research is to characterize the European settlement-era forests based on these witness tree data. Specifically, we intend to: 1) characterize the spatial distribution of selected species; and 2) characterize the early forests of the MNF in terms of relationships between species and physical variables. The tree species assessed were American chestnut, white oak, chestnut oak (*Q. prinus* L.), northern red oak (*Q. rubra* L.), scarlet oak (*Q. coccinea* Münchh.), black oak (*Q. velutina* Lam.), and hickory species. The oak and hickory species were selected for analysis because they often dominate the central hardwoods forest (Fralish 2003) and American chestnut was selected because of its historical extent in the central hardwoods forest and study area. This information may then be used in studies for occurrence modeling and future restoration efforts.

STUDY AREA

Most of the witness tree locations are within 5 km of the proclamation boundary, creating a study area of approximately 689,000 ha. The MNF in eastern West Virginia consists of complex topography in both the Allegheny Mountains and Northern Ridge and Valley physiographic sections (Fig. 1). The MNF includes the highest point in West Virginia and many valleys which quickly drop more than 800 m in elevation. This complexity results in a great variety of landforms and conditions for vegetative diversity.

Subsections represented on the MNF include: Eastern Allegheny Mountain and Valley (EAMV), Eastern Coal Fields (ECF), Northern High Allegheny Mountain (NHAM), Ridge and Valley (RV), Southern High Allegheny Mountain (SHAM), Western Allegheny Mountain (WAM), and Western Allegheny Mountain and Valley (WAMV) (Cleland and others 2007).

The moisture and temperature regimes of the subsections are described in detail in Table 1 (Cleland and others 2007). In general, the RV subsection is warm and dry and the WAMV subsection dry with moderate temperatures. The ECF subsection is warm and moderate in overall moisture. The EAMV subsection is moderate in both moisture and temperature regimes, and the WAM subsection is cool with moderate moisture. The SHAM and NHAM subsections are both wetter than the other subsections; however, NHAM has the lowest average temperatures while SHAM is more moderate.

Appalachian oak forest is the primary potential natural vegetation for the RV, EAMV, and WAMV subsections. In contrast, Northern hardwood is the primary potential natural vegetation for the NHAM and SHAM subsections. The mixed mesophytic type is the primary potential natural vegetation for the WAM and ECF subsections (Cleland and others 2007; Table 1).

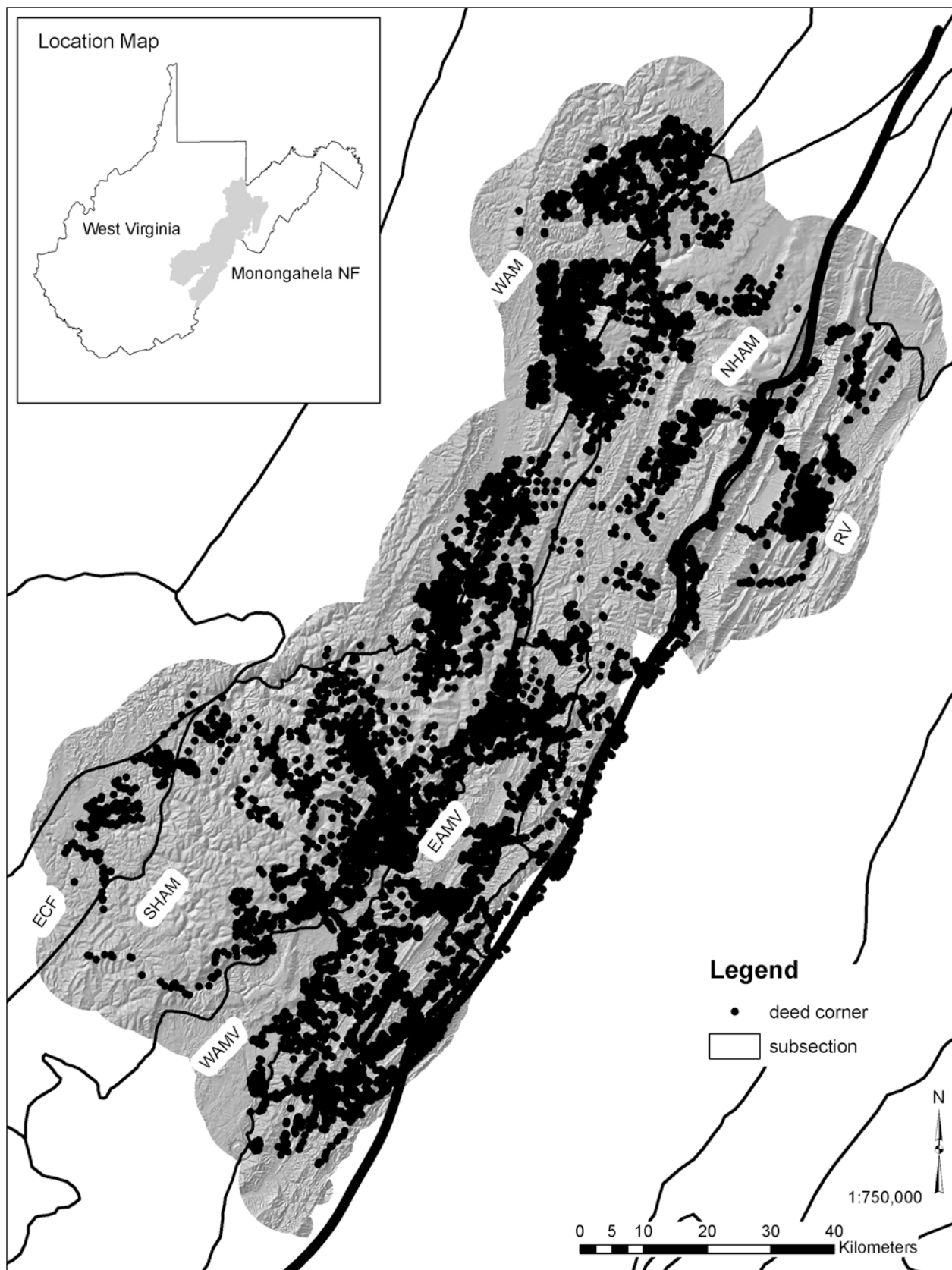


Figure 1.—Study area location with corner locations. Thicker line is boundary between Northern Ridge and Valley and Allegheny Mountains Sections. EAMV = Eastern Allegheny Mountain and Valley, ECF = Eastern Coal Fields, NHAM = Northern High Allegheny Mountain, RV = Ridge and Valley, SHAM = Southern High Allegheny Mountain, WAM = Western Allegheny Mountain, and WAMV = Western Allegheny Mountain and Valley.

Table 1.—Selected subsection climate and potential natural vegetation attributes (Cleland and others 2007).

Subsection	Avg. Annual Max. Temp. (°C)	Avg. Annual Min. Temp. (°C)	Avg. Jan. Min. Temp. (°C)	Avg. Annual Snowfall (cm)	Avg. Annual precipitation (cm)
RV	17.3	4.1	-7.1	82.6	102.3
NHAM	14.5	2.0	-9.4	263.5	128.2
WAM	15.1	3.0	-8.8	187.8	122.2
SHAM	15.2	2.4	-9.0	224.5	138.1
EAMV	16.5	2.6	-8.8	120.5	111.0
WAMV	17.1	4.3	-7.0	85.6	99.7
ECF	17.5	4.5	-6.8	94.9	113.7

Potential Natural Vegetation (%)					
	Appalachian oak forest	Mixed mesophytic forest	Northeastern spruce-fir forest	Northern hardwoods	Oak-hickory-pine forest
RV	49.1		0.7	9.2	40.9
NHAM			14.3	80.4	5.4
WAM	26.6	38.0		35.3	0.1
SHAM	4.5	23.3	21.1	51.2	
EAMV	58.9		0.2	40.9	
WAMV	85.9	1.1		13.0	
ECF	28.1	51.7		20.2	

EAMV = Eastern Allegheny Mountain and Valley, ECF = Eastern Coal Fields, NHAM = Northern High Allegheny Mountain, RV = Ridge and Valley, SHAM = Southern High Allegheny Mountain, WAM = Western Allegheny Mountain, and WAMV = Western Allegheny Mountain and Valley.

METHODS

DATA COLLECTION

The parcel descriptions from the old deeds (1752 to 1910) were used in the 1930s to plot the parcels on maps. The resulting 83 planimetric maps of uniquely identified individual parcels cover an area of approximately 10,832 ha each; the maps overlap and non-MNF area was included. The full dataset includes deeds up to 1910, but this analysis includes deeds/corners up to 1900.

In 2005, the 1930s maps were scanned and geo-referenced, using at least four tie points. Geo-referencing was done using the GIS software in ArcMap (ESRI, Redlands, CA, 2002) by linking coordinates on the scanned maps to the 1:4800 State Addressing Mapping Board orthophotos. The maps were used as a base for digitizing the corners of each parcel. In 2009, manual digitizing of corners was completed using GIS at a scale of 1:5000. The hand-drawn features and printed map line of the 1930s maps, now scanned, were used to create and place corners, creating points in a digital file. The deeds were used to determine corners on the maps and enter tree species or other markers in the attribute field. Trees mentioned in the deeds as rock oak were treated as chestnut oaks (Strausbaugh and Core 1977), and those named span, Spanish, or pin oaks were tallied as scarlet oak (Burns and Honkala 1990).

At each corner, a relative frequency of each species was calculated by dividing the number of trees of each species at the corner by the total number of trees at the corner (Wang and others 2009). The relative compositions of the corners allowed for the use of spatial autocorrelation statistics. For indicator species analyses these relative frequencies were converted to presence or absence of a species at each corner.

SPATIAL AND STATISTICAL ANALYSES

To assess the diverse topography, climate, and soils of the study area, the analyses were stratified by ecological sub-section (Black and Abrams 2001a). This stratification should also address the possible bias for some landforms (Black and Abrams 2001a). Spatial statistics were used to describe the distribution of the witness trees by species and to discern patterns in the distribution.

To identify overall patterns in the witness trees, Moran's I was calculated in ArcGIS (ESRI, Redlands, CA 2002) using the species' relative frequency as the attribute value and calculating the spatial autocorrelation from just those points with the selected species present (the number of points used differed by species). This measure of spatial autocorrelation is a broad comparison of the study locations to a theoretical random distribution of points and associated values. Moran's I ranges from -1 to +1. Scores closer to +1 indicate clustered distributions, scores closer to -1 identify uniform or dispersed patterns, and values close to 0 indicate random patterns. The strength of any pattern discerned in the spatial data is reflected in a calculated Z-score; a 95-percent confidence interval was used ($\alpha = 0.05$).

To characterize the physical site characteristics associated with tree species of the European settlement-era forests, environmental variables at the corner points were extracted from existing spatial datasets and variables derived from an 18-m digital elevation model (DEM). Physical components assessed were topographic roughness, a moisture index, aspect, landform, and elevation. The corner points were buffered by 21 m, which is the spatial error of the dataset calculated from measurements of images on the scanned maps used to place digital points.

Topographic roughness (TRI) was calculated as the square root of the sum of squared differences in elevation between a cell and its eight neighboring cells (Riley and others 1999). A moisture index was calculated as the natural log of flow accumulation + 1, divided by slope + 1 of a target cell (Anderson and Merrill 1998). Elevation, moisture index, and topographic roughness index were all calculated as an average for a 21-m radius around the corner location. These averages were then classified into high, medium, and low groups based on quantiles. Low elevations ranged from 1 to 698 m, moderate elevations were between 698 and 872 m, and more than 872 m was considered high elevation. Low TRI ranged from 1 to 32.4 m, moderate TRI from 32.5 to 68.1 m, and high from 68.2 to 224.7 m.

Aspect, elevation, topographic roughness, and the moisture index were derived from the 18-m DEM of the study area. Landforms were extracted from an existing spatial dataset of the MNF. Landforms include: ridge/peak, bench/plateau, toe slope, side slope, cove, and valley floor. Buffering the corners by 21-m resulted in species-landform tallies that duplicated some corners as the buffer could encompass more than one landform.

To determine significant associations, indicator species analysis on the presence/absence of species by site variable was conducted in PC-ORD (McCune and Mefford 2006). Indicator species analysis is a multivariate, non-parametric method to describe relationships between species and environmental groups based on proportional frequencies of the species by groups; groups are based on the categorical site variables. Statistical significance ($\alpha = 0.05$) is computed through a Monte Carlo permutation method where the data matrix is shuffled 4,999 times with results compared to the results from real data (McCune and Grace 2002). While only results for selected species are reported here, the indicator species analysis and significance testing included all major species ($n > 30$) found in the dataset.

RESULTS

The analysis was made on 15,591 corners (1,279 included markers other than trees) representing 22,107 witness trees. Each corner represented up to six witness trees; about 8 percent of the corners were not witnessed by any trees. About 24 percent of the corners date to the late 1700s. The greatest numbers of corners were established in the 1840s through the 1850s, at 17.8 percent and 29.3 percent, respectively.

The selected species constituted 44 percent of all tallied witness trees (Table 2). All oak species combined totaled 32 percent of species tallied, with white oak making up the majority.

Of the 49 combinations of subsections and species, 28 were found to have a clustered pattern (positive spatial autocorrelation), 20 were found to have random patterns (no spatial autocorrelation), and one (scarlet oak in ECF) had too few occurrences to calculate spatial autocorrelation (Table 2). All species showed a clustered pattern when assessed across the study area.

In the RV subsection, no significant associations with aspect, TRI, and moisture were found based on the Monte Carlo tests. Monte Carlo tests found no significant correlation between frequencies of any species and moisture index or TRI in the ECF subsection. Frequencies of the selected species were not found to be correlated with aspect in the WAMV and WAM subsections. Table 3 contains the summarized results for all indicator species analyses.

AMERICAN CHESTNUT

American chestnut witness trees were associated with areas of low moisture (WAM and EAMV) and high (RV and EAMV) and moderate (WAM) elevations. American chestnut witness trees were more likely to occur on areas with moderate TRI in the EAMV subsection. American chestnut was significantly associated with ridge landforms (WAM, ECF, WAMV, and EAMV). In the SHAM subsection the species was more likely on toe slopes and in the NHAM subsection it was more likely on benches. On the study area as a whole, American chestnut trees were more likely found on ridges with high TRI, low moisture, and high elevations.

Table 2.—Species' relative frequency by subsection. Highlighted estimates denote random spatial pattern; all others occurred as clustered spatial pattern (global Moran's I; $\alpha = 0.05$)

Species	EAMV	ECF	NHAM	RV	SHAM	WAMV	WAM	Study area
American chestnut	6.4	18.4	4.0	4.0	4.7	6.1	9.3	6.3
White oak	32.5	10.7	5.7	28.3	3.9	38.2	17.0	19.4
Chestnut oak	6.1	3.0	2.3	12.3	1.0	2.2	7.6	5.2
Northern red oak	2.9	2.9	3.3	4.8	3.1	2.2	2.6	3.1
Scarlet oak	0.9	0.5	2.1	2.0	0.9	3.8	4.3	2.0
Black oak	2.8	4.0	0.2	5.6	0.7	5.0	1.4	2.2
Hickory	6.1	2.2	1.3	4.5	1.7	5.1	5.3	4.1

EAMV = Eastern Allegheny Mountain and Valley, ECF = Eastern Coal Fields, NHAM = Northern High Allegheny Mountain, RV = Ridge and Valley, SHAM = Southern High Allegheny Mountain, WAM = Western Allegheny Mountain, and WAMV = Western Allegheny Mountain and Valley.

Table 3.—Indicator species analysis results for witness trees and moisture index, TRI, elevation, aspect, and landform across the study area. Differences between subsections noted in parentheses.

Species	Moisture	TRI	Elevation	Aspect	Landform
American chestnut	low	high (moderate EAMV)	moderate (high RV, EAMV)	none (N-NW-W EAMV)	ridge (toe slopes SHAM; bench NHAM)
White oak	high (low NHAM and SHAM)	low (high NHAM; moderate EAMV)	low (moderate NHAM, SHAM, WAMV)	none (SW SHAM)	valley floor (toe slopes SHAM, RV, WAMV)
Chestnut oak	low	high	moderate (high RV, ECF, EAMV)	none (N-NW-W EAMV)	ridge (bench NHAM)
Northern red oak	low	high (moderate WAMV)	high (moderate WAM)	S-SE	bench (toe slope RV; ridge EAMV; valley ECF)
Scarlet oak	low (high WAMV)	high	moderate (high WAM, EAMV)	none (SW NHAM; S-SE SHAM)	none (cove RV)
Black oak	low	none (high EAMV)	low (high EAMV)	none (S-SE EAMV)	none (toe slope RV; ridge EAMV; valley ECF)
Hickory	none	high	low	none (SW NHAM and ECF; S-SE SHAM)	none (side slope SHAM)

EAMV = Eastern Allegheny Mountain and Valley, ECF = Eastern Coal Fields, NHAM = Northern High Allegheny Mountain, RV = Ridge and Valley, SHAM = Southern High Allegheny Mountain, WAM = Western Allegheny Mountain, and WAMV = Western Allegheny Mountain and Valley.

WHITE OAK

White oaks were more likely on areas of low moisture (NHAM and SHAM). The association of white oaks with TRI varied by subsection with white oaks significant on low TRI areas in the WAM subsection, high TRI areas in the NHAM subsection, and moderate TRI in the EAMV subsection. On the NHAM, SHAM, and WAMV subsections, white oaks were associated with areas of moderate elevation. They were significantly associated with low elevations on the WAM and RV subsections. White oaks were more likely to be located on SW aspects (SHAM). White oaks were associated with valley floors (WAM) and toe slopes (SHAM, RV, and WAMV). Across the study area white oaks were more likely on valley floor landforms at low elevations with high moisture and low TRI.

CHESTNUT OAK

Chestnut oaks were associated with areas of low moisture and areas of high TRI (WAM, NHAM, and EAMV). Chestnut oaks were more likely on high (RV, ECF, and EAMV) or moderate (WAM) elevations and were associated with ridges (EAMV and ECF). Across the study area chestnut oak witness trees were more likely to be found on ridge landforms with high TRI, low moisture, and moderate elevations.

NORTHERN RED OAK

Northern red oaks were found on areas of low moisture (WAMV, SHAM, and EAMV) and high (RV and EAMV) to moderate (WAM) elevations. Associations with landforms varied between subsections. Toe slopes were significant in the RV subsection, benches in the SHAM and NHAM subsections, and ridges in the EAMV subsection. Taken across the study area, northern red oak witness trees were more likely on S-SE facing benches at high elevations, with high TRI and low moisture.

SCARLET OAK

Surprisingly, the cove landforms were found to be significant for scarlet oak in the RV subsection. Scarlet oak were more likely on SW (NHAM) and S-SE (SHAM) aspects. In the WAM subsection, scarlet oaks were associated with higher elevations and high TRI. An association with areas of high TRI was also found for the EAMV subsection. Across the study area low moisture index, moderate elevation, and high TRI were found to be significant for describing the site-species relationship for scarlet oak.

BLACK OAK

Black oak was associated with areas of low moisture and high TRI (EAMV). Black oak was associated with low elevations in the NHAM subsection and areas of high elevation in the EAMV subsection. Black oak was associated with toe slopes (RV), valleys (ECF), and ridges (EAMV). In the EAMV subsection, black oaks were more likely on S-SE slopes. Across the study area, black oaks were associated with low moisture and low elevations.

HICKORY

Hickory witness trees were associated with areas of high TRI (NHAM) and low elevations (NHAM and EAMV). Hickories were more likely to occur on SW (NHAM and ECF) and S-SE aspects (SHAM). Hickory was more likely to be found on side slopes (SHAM). Across the study area, hickories were more likely to be found on low elevations with high TRI.

DISCUSSION AND CONCLUSIONS

All species locations were found to be spatially auto-correlated when assessed across the study area. This is not surprising as tree species are not generally randomly located across a landscape as large as the study area. The lack of spatial autocorrelation in some combinations of species and subsections was not expected and may be due to low numbers of points in some subsections (ECF) or the shape of the subsections (RV).

Positive spatial autocorrelation is common in ecological data. Ignoring spatial autocorrelation can cause problems in hypothesis testing, especially in studies that are not from planned experiments (Lennon 2000). There is a risk of inflating Type I errors (false positives) if simple confirmatory statistics are used on spatially auto-correlated data (Diniz-Filho and others 2003) because the degrees of freedom are overestimated (Lennon 2000). Permutation tests, where significance is based on random shuffling of the observations, are appropriate when positive spatial autocorrelation exists (Legendre 1993). For these reasons, analyses such as correlation and linear regression were not used to explain the relationships between species and sites. Regression models can be modified to account for spatial autocorrelation (Lennon 2000) and any future analysis of this dataset should include consideration of this spatial property.

While the subsection boundaries define areas based on observed ecological differences, there were relatively few differences in site-species associations between the subsections. Perhaps other physical or ecological variables such as soil series would show differences between subsections. Although few differences between subsections were noted for these species and site characteristics, construction of predictive models should consider subsectional differences.

One interesting difference between subsections is the variety of landforms associated with black oaks—ranging from toe slopes and valley to ridges. These differences are likely due to the different moisture regimes of the subsections; the RV is the driest of the study area. Another interesting difference between subsections but within a species is the association of northern red oaks to different landforms. Across the study area, northern red oaks are more likely on benches, but significant associations to ridges and toe slopes were also determined. These findings again may reflect the different moisture or temperature regimes in these subsections and the species' optimal habitat for competition.

As in the previous analysis of witness trees in the study area (Abrams and McCay 1996), chestnut oaks, northern red oak, scarlet oak, black oak, and American chestnut were found in areas of low moisture and ridge positions. In the current analysis, white oak was found to be more likely to occur on valley floors than in other landforms, while no landform preference was found to be significant previously.

Across its range, white oak can occur on all upland aspects and slope positions except for extremely dry ridges, poorly drained flats, and wet bottomland (Burns and Honkala 1990). This conclusion would seem counter to the findings presented here, where white oak was more likely on moist sites and valley floor landforms. White oak does not occur at higher elevations across its range (Burns and Honkala 1990) and this observation is reflected in the current analysis of witness trees.

Northern red oak currently occurs across all topographic positions, under mesic to dry-mesic conditions; it grows best on N and E aspects (Burns and Honkala 1990). Based on the variety of landforms found significant at the subsection level, the witness trees reflect the species' suitability to many topographic positions. In contrast, S-SE aspects were found to be the only significant aspect for northern red oak trees.

These differences between current species distributions and this analysis of witness trees may be due to the comparison of the entire range with a portion of a species range. Comparing current distributions with past distributions also may explain these differences. The witness trees reflect species distributions before substantial influence of European settlement activities that may be responsible for the current distribution of the species, including fire suppression.

That American chestnut was associated strongly with ridge landforms and areas of low moisture is consistent with previous witness tree analysis (Abrams and McCay 1996). Before the near-elimination of the species from the chestnut blight, American chestnut was found on the hillsides in most of West Virginia (Hough 1878) and was dominant on ridges in western Maryland (Russell 1987). In West Virginia, the abundance of American chestnut dropped as elevation increased (Hough 1878). In this analysis, American chestnut was associated with moderate elevations. That no species was found to be significantly associated with side slopes, the most common landform in the study area, may indicate that side slopes were generally a mixture of these, and possibly other, species.

Indicator species analysis does not allow for the determination of site characteristics negatively associated with species, as can be made using the contingency table analysis. However, given that the witness tree corners exhibit positive spatial autocorrelation at the level of the study area—indicating the observations are not independent—the nonparametric indicator species analysis and Monte Carlo testing for significance are more appropriate.

Considering the possibility of confusion over common names used in the deeds, the creation of a red oak species grouping may be useful in future analyses. Future use of this database will include spatial interpolation of species occurrences between points to create continuous coverage and comparisons of past species abundances to current forest conditions.

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EFFECTS OF NATURAL GAS DEVELOPMENT ON FOREST ECOSYSTEMS

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Abstract.—In 2004, an energy company leased the privately owned minerals that underlie the Fernow Experimental Forest in West Virginia. The Fernow, established in 1934, is dedicated to long-term research. In 2008, a natural gas well was drilled on the Fernow and a pipeline and supporting infrastructure constructed. We describe the impacts of natural gas development on the natural resources of the Fernow, and develop recommendations for landowners and land managers based on our experiences. Some of the effects (forest clearing, erosion, road damage) were expected and predictable, and some were unexpected (vegetation death from land application of fluids, an apparent increase in white-tailed deer presence). Although this is a case study, and therefore the results and conclusions are not applicable to all hardwood forests, information about gas development impacts is sufficiently rare that forest managers, research scientists, and the concerned public can learn from our experience.

INTRODUCTION

The increased demand for natural gas during recent decades, coupled with advances in extraction technology, has led to an increase in exploration and development in many previously unexplored areas in the United States, most notably areas underlain by Marcellus shales in the eastern part of the nation. The impacts of increased development pressure on other natural resources, including soil water, wildlife, and vegetation, are relatively undocumented for forest ecosystems in the eastern United States. Landowners, land managers, and policy makers require such information about these effects to help them make decisions about mineral resource development.

In 2008, drilling for natural gas and subsequent pipeline construction were implemented to extract the privately owned minerals underneath the Fernow Experimental Forest in West Virginia. The Fernow is well known for long-term silviculture, watershed, and ecological research (Kochenderfer 2006). In this paper, we describe some of the impacts of this natural gas development on the natural and scientific resources of the Fernow, and identify opportunities to mitigate possible impacts based on our experiences. This report describes results from a single case study, and the data described herein generally come from post-hoc monitoring, not designed experimentation. In addition, many of the potentially sensitive components of the ecosystem were not monitored; information on fauna, in particular, is lacking. Nonetheless, conclusions can be drawn which may be useful to other land managers.

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SITE DESCRIPTION

The well site is located within the 1,902-ha Fernow Experimental Forest, which lies in the Allegheny Mountain section of the mixed mesophytic forest (Kochenderfer 2006). The Fernow was set aside to “make permanently available for forest research and the demonstration of its results a carefully selected area representing forest conditions that are important in Northeastern West Virginia” (Fernow Establishment Order, dated March 23, 1934; on file at the Timber and Watershed Laboratory, Parsons, WV).

The gas well site and access road are located on karst topography within the Ellick Run watershed. Surficial bedrock geology is the Greenbrier Group, which is composed of interbedded limestones and calcareous shales, and includes cavernous limestone. Groundwater emerges at two springs located approximately 1,000 m northeast of and at a lower elevation than the well site. Known underground passageways within the Big Springs Cave are located approximately 600 m away from the well site. Big Springs Cave is the largest hibernaculum on public land in West Virginia for the endangered Indiana Bat.

PROCESS OF DEVELOPMENT OF THE WELL SITE

An abbreviated timeline of the well and pipeline development process is provided in Table 1 (for a detailed timeline and for details of monitoring methods, see Adams and others in process). A similar time frame might be expected for well development on other lands, but the timing of the various activities was controlled by weather conditions by the availability of equipment required at each stage of well development. Trees and all vegetation were cleared from an area of about 1.4 ha, followed by earthwork to construct a 4.6-m-wide road, a level drill pad of about 30 m by 61 m, and a pit 24 m by 37 m by 3 m deep. The purpose of the plastic-lined drilling pit was to hold rock cuttings and drilling residues.

Table 1.—Abbreviated chronology of activities on the natural gas well site and associated pipeline, Fernow Experimental Forest.

Date	Activity
May 11, 2007	Energy company contacts the U.S. Forest Service and requests input.
Sept. 11, 2007	FS receives copy of well work permit application.
Sept. 21, 2007	WV Department of Environmental Protection, Office of Oil and Gas, approves well work permit application.
Jan. 2008	Timber on access road/well site cut by Fernow logging crew.
Feb.-April, 2008	Construction of access road/well site.
April 8, 2008	Well drilling begins.
April 30, 2008	Well reached total depth of 2,387 m. Proved gas in the Huntersville Chert portion of the Oriskany and associated strata, and possibly the Oriskany sandstone. Drill pipe broke off and stuck in the well bore. Directional drilling around stuck pipe.
May 15, 2008	Drilling completed.
May 23, 2008	Hydrofracing begins, completed on 5/31.
June 13, 2008	Road repairs completed.
June 21, 2008	Land application of drill pit fluids completed to Site 1.
July 28, 2008	Pit contents solidified and back-filled.
September 9, 2008	Recontouring of the well site begins.
October 2, 2008	Cutting of pipeline right-of-way begins.
October 7, 2008	Recontouring of well site completed.
December 15, 2008	Well pad and access road seeded and mulched.
December 31, 2008	Cutting of pipeline right-of-way and removal of harvested stems completed.
January 29, 2009	Pipeline installation completed.

Drilling required approximately 3 weeks, and was completed to a depth of 2,387 m in the Oriskany sandstone. After completion of drilling, the rock layer was fractured to release natural gas in the formation—a process known as hydrofracturing. Following hydrofracturing, the hydrofracturing fluids were recovered and temporarily stored in the pit. Subsequently, the majority of the drill pit fluids (302,800 L) were land-applied to an area of about 0.2 ha within a nearby research compartment (Fig. 1). This fairly small area was negotiated in an effort to minimize impacts on ongoing research nearby. Foliar injury and death of vegetation during this application necessitated a change of application site, and the remaining 75,700 L was applied within the pipeline right-of-way to an area of about 0.04 ha. The remaining pit contents were then solidified by adding cement, and buried in place.

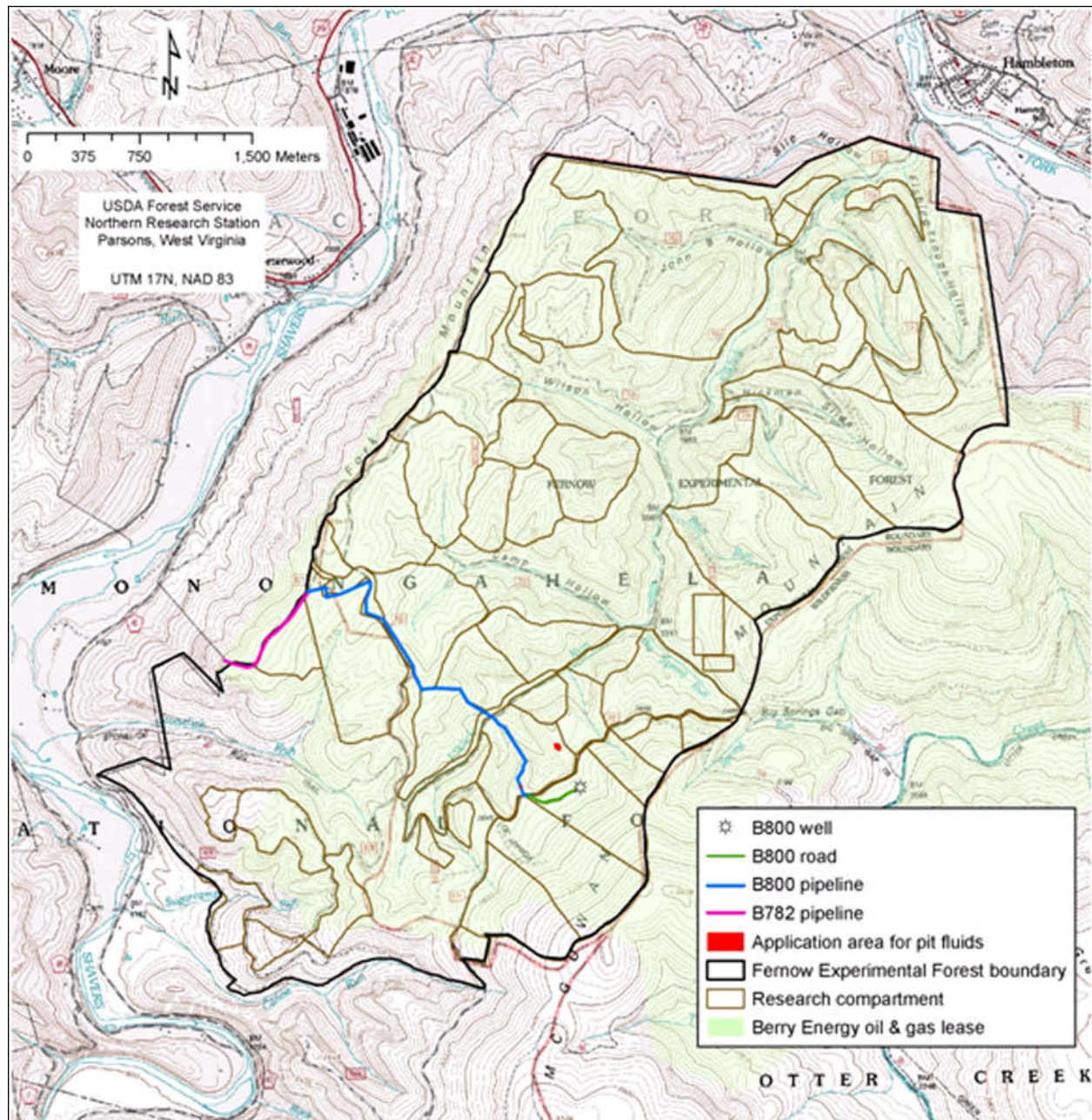


Figure 1.—Fernow Experimental Forest, West Virginia, showing location of features related to natural gas exploration and development project.

OBSERVATIONS

CLEARING AND SITE DEVELOPMENT

An area of about 0.63 ha was cleared for the well-pad and drill pit. All vegetation within the perimeter was removed, including 94 trees (>28 cm diameter at breast height [d.b.h.]) and 126 pulpwood stems (52,200 bd ft, or 123.2 m³). Most of the well site will remain in herbaceous cover, except for the well-pad itself and the access road, which were graveled.

The pipeline corridor is estimated at 2 ha in extent (8 to 9 m wide, and 1,800 m long). All vegetation was cleared from the area and stumps removed. Brush was piled along the edges of the right-of-way. In total, 714 trees were removed, 310 of them (43 percent) being greater than 28 cm d.b.h. (volume of 84,600 bd ft, or 199.6 m³). The pipeline is to be maintained as a grassy corridor by mowing, and no trees will be allowed to become established.

EROSION AND SEDIMENT

A total of 1,330.12 kg of soil material was collected behind silt fences located below the well site. This number represents total soil loss during summer 2008, from a cleared area of about 0.63 ha, and equates to a loss rate of 2.1 metric tons per hectare during a 3-month period. This figure likely underestimates total soil erosion losses, however; site soil was also lost prior to silt fence installation, and soil loss occurred off the well site at other locations that were not monitored (Edwards 2008). For comparison, rates of erosion from road surfaces in eastern forests range from 1.9 metric tons per hectare (Dickerson 1975) to 337 metric tons per hectare (Ursic and Douglas 1979). Chen (1974) estimated erosion losses from urban construction sites to range from 11 to 450 metric tons per hectare per year. Given that we measured soil loss from a portion of the site for only a portion of the year, it is apparent that soil erosion from gas well development can be significant.

There is no evidence that soil eroded from the well site reached nearby springs (Edwards 2008). More than 99 percent of the reported turbidity values from the springs were less than 40 NTUs. All of the turbidity values observed during monitoring of these two springs were within ranges commonly reported within this region, and variability was notably low.

ROAD DAMAGE

Damage to the Fernow's graveled roads was anticipated and did occur. Road use was estimated to exceed 9,300 metric tons during the approximately 6-month development phase, compared with about 4,500 metric tons annually for logging on the Fernow. Ruts of 30 cm or more developed, roadside drainage ditches were filled or collapsed, and erosion of the road surface subsequently occurred. Damage was repaired satisfactorily after completion of well site construction and drilling, but temporary repairs during the drilling process were not always sufficient to keep the road open to the general public.

DRILLING AND HYDROFRACING FLUIDS

Fluids from the drill pit were land-applied at two locations on the Fernow in June 2008, with nearly immediate impacts on vegetation. The fluid from the drill pit was sprayed into the air and onto the vegetation, although this application method may not be the standard protocol. The assumption was that if the drill pit fluids met the standards specified in the permit, there would be no damage to the vegetation or soil. Permit levels, as established by the West Virginia Division of Environmental Protection, dictate land-applied drill pit

fluids must have chloride concentrations less than 12,500 mg/L and a pH between 6 and 10. State regulations require only one sample of the effluent be taken from the distribution hose during land application. The drill pit fluids met the permit concentration levels, but total vegetation mortality was observed on the first fluid application site almost immediately after land application. Because of this unexpected effect, a second fluid application site was negotiated, and a smaller amount of fluids applied to this site.

At the first fluid application site, many trees, shrubs, and understory plants showed immediate responses to the fluid application, with leaves turning brown, wilting, and subsequent leaf and bud mortality. We also observed that taller trees, whose leaves were not contacted by the fluids, also began showing decline symptoms about 10 days after the ground vegetation; these symptoms included leaf browning, leaf curling, and premature leaf drop. Premature leaf fall ranged from 227 to 1,395 kg ha⁻¹, or about 10 to 45 percent of annual autumn leaf fall biomass (Adams 2008). We inferred from these observations that some of the vegetation was damaged immediately by contact with the drill pit fluids, but most likely many of the larger trees were killed as a result of uptake of the fluid through roots from the soil.

All herbaceous and shrub vegetation within the perimeter of the application area showed damage symptoms in 2008; 115 trees (>2.54 cm d.b.h.) exhibited decline symptoms. In 2009, that number had increased to 147 trees (basal area 3.8 m²). Some recovery of understory vegetation occurred in 2009. Note, however, that more than 50 percent of the trees had no foliage in 2009, and 65 percent of the trees had less than one-third full crown. Mortality was most evident in American beech, with bark sloughing from the bole and branches on 38 percent of the beech trees within the application area perimeter.

Damage symptoms on the second fluid application site were less dramatic (browning of leaves, particularly of northern red oak seedlings), most likely because a much lower volume of fluids was applied, and the operator took care to apply the fluids onto the ground, rather than spraying it onto the vegetation.

We hypothesized that the vegetation on fluid application site 1 was killed as a result of very high salt loading to the site. Although concentrations of chlorides met the permit criteria, an estimated 302,800 L of fluid were applied to this site, resulting in a load of 11,355 kg chlorides per ha. Such a loading far exceeds load limits established elsewhere, such as Oklahoma (450 kg ha⁻¹), Wisconsin (275 kg ha⁻¹ on a 2-year basis; www.dnr.state.wi.us/org/water/wm/ww/gpindex/57665_permit.pdf), and in Saskatchewan (400 kg ha⁻¹; http://eps.mcgill.ca/%7Ecourses/c550/Environmental-impact-of-drilling/Sask_Drilling_Waste_Guidelines.pdf).

Monitoring of soil chemistry over time has confirmed that high levels of chlorides, sodium, and calcium were found in the application areas immediately after the fluids were applied (Fig. 2). Although concentrations have decreased over time, soil chloride and sodium levels were still significantly elevated in May 2009 in the application area relative to the control area.

DEER ACTIVITY

We observed extensive use of the area by white-tailed deer, and to a lesser extent black bear, as represented by high density of hoof prints, browse sign, turned-over logs, and disturbed soil, on the areas adjacent to the gas well, and on the fluid application areas. The levels of wildlife sign greatly exceeded what is normally observed on the Fernow in a localized area. After reclamation of the drill pit, there were several seeps on the lower side of the well pad where the contractor was unable to establish vegetative cover because deer utilized this area as

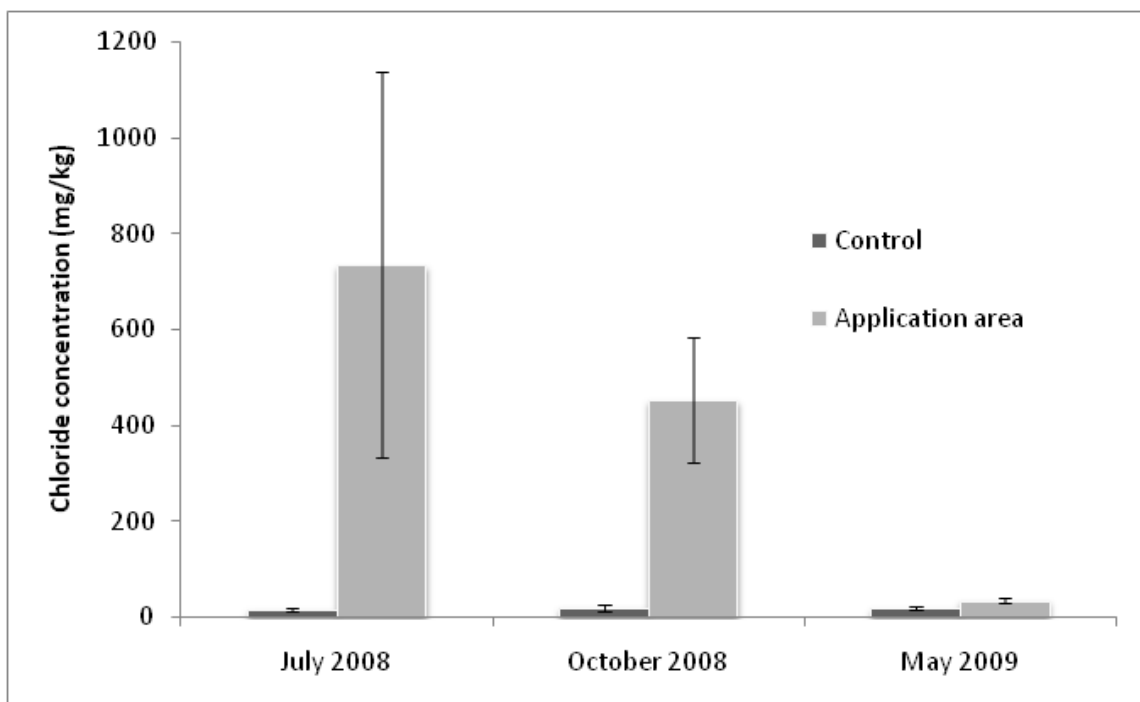
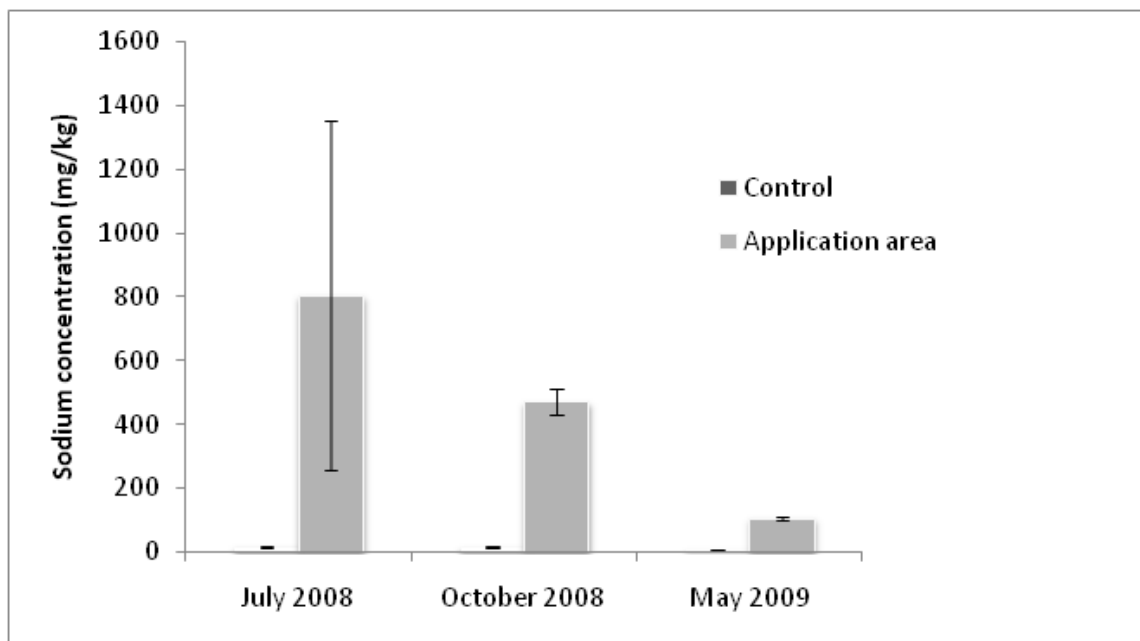


Figure 2.—Sodium and chloride concentrations in soil from fluid application area 1, and an adjacent control area, Fernow Experimental Forest, West Virginia, over time. Fluids were applied in late June 2008. Soil samples were collected from the surface 10 cm from three transects per area.

a mineral source or “lick”. Covering of these sites with slash was moderately successful in allowing germination of the seeded ground cover to proceed, but high levels of deer activity around these seeps continued into late summer 2009. Similar deer behavior has been previously documented as an unintended consequence of gas development in the region (Campbell and others 2004).

RECOMMENDATIONS

All discussions about development of energy resources, including natural gas, should include careful consideration of the effects on natural resources. From our experiences on the Fernow Experimental Forest, we present some of the lessons learned that perhaps should be incorporated into these important discussions.

Some of the impacts of natural gas development can be predicted, and these impacts can be lessened through planning and cooperation. For example, the area cleared for the well site, and for the pipeline in particular, was negotiated so as to minimize impacts. Soil loss from construction of the well site was inevitable, but silt fences installed by the contractor and by the U.S. Forest Service minimized transport of that sediment to nearby water sources. These mitigating activities required negotiation and communication by all parties involved.

The unexpected will always happen, and should be part of any planning discussions. The unexpected did occur during the development of the gas well and pipeline on the Fernow. We attribute such occurrences to accidents, equipment failures, and misconceptions about what to expect. Our misconceptions may be similar to those of private landowners with little direct experience with gas well development. Discussions of the process between the energy developer and the landowners/managers should happen early and often to be sure that all parties’ expectations are clear. Visits to active development sites can help landowners better understand what to expect, and also help them communicate their wishes to the developer. Landowners who also own the subsurface rights have the authority to require the developer to follow desired procedures. Landowners who do not own the subsurface rights still retain some legal rights during gas well development, and may want to seek council to ensure those rights are respected. Surface owners’ organizations (for example, www.wvsoro.org) can help to educate landowners about their rights and what to expect. While we recognize that it is impossible to foresee every eventuality, we suggest that a thorough analysis of risks to natural resources, using alternative “what-if” scenarios, should be conducted.

Much more research is needed immediately to better understand and predict the effects of natural gas exploration and development on forests, particularly in the eastern United States. We were surprised by the paucity of peer-reviewed research evaluating effects of natural gas development on forest lands in the eastern United States. For example, as mentioned above, the permit standard in West Virginia for land application of hydrofracing fluids is based on a concentration rather than a total amount of chemicals applied, which might provide better resource protection. Unfortunately, little research is available to use in developing a dose-based standard, and such research could improve information available to policymakers, regulators, and energy developers. We also know little about the effects of exploration and development on ground water hydrology and water quality, and surprisingly little about effects on downstream surface waters. In general, information about effects on most fauna also is lacking (but see Moseley and others 2009).

Other questions include:

- How can the risks associated with natural gas development in karst geology be evaluated and mitigated?
- What are the effects of elevated chlorides and salts on water quality and biota in the soil, ground water, or stream water?
- Are there significant edge effects from the pipeline corridor—changes in light, temperature, or moisture regimes? Are these similar to effects of roads and other permanent openings?
- How can we evaluate the cumulative effects of natural gas development within a landscape?

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

SPATIAL IMPACT ASSESSMENT OF CONIFER STANDS IN THE HOOSIER NATIONAL FOREST

Richard Thureau, Craig Wayson, Dale Weigel, and Jeff Ehman¹

Abstract.—Forest management decisions on Federal lands must be administered at many spatial and temporal scales. Forest condition, size class, and cover type at the stand level determine how silvicultural practices today will impact management area and overall forest goals in the future. The Hoosier National Forest (HNF) Land Resource Management Plan lists eight goals for guiding management decisions on the HNF to meet the needs of the people of Indiana and the rest of the United States. Nonnative conifer species were introduced in the HNF in the 1930s-1950s to reclaim abandoned agricultural lands, but perpetual non-native forest stands are in direct conflict with some current goals. The objective of this study is to assess the impact of existing conifer stands on management goals outlined in the HNF Plan. This assessment ranks HNF stands by forest goal values and the relationships with conifer stand cover. The impact of conifer stands on HNF goals was measured at the stand level by examining the relationship between forest goal values and conifer fraction. Forest goal indices are inversely correlated with conifer fraction within Tell City Management Unit stands. This analysis demonstrates how forest goals, and therefore management strategies, vary over both time and space.

INTRODUCTION

Decisions about managing Federal forest land must consider implications at many spatial and temporal scales. Management objectives incorporate national interests (such as directives mandated in the National Forest Management Act²), as well as site-specific conditions impacting local ecology and productivity. Management practices must account for both present conditions and those desired in the future. The way that silvicultural practices implemented today will affect the management area and overall goals in the future depends on forest condition, size class, and cover type at the stand level. However, just as forests evolve over time in response to management practices and local ecological and environmental processes, the demands for forest goods and services also evolve in response to societal needs and attitudes.

To guide management decisions on the Hoosier National Forest (HNF) in Indiana, the HNF Land Resource Management Plan (HNF Plan) contains eight goals intended to meet the needs of residents of the State and the rest of the country (U.S. Department of Agriculture, Forest Service [USDA FS] 2006). The eight goals listed in Table 1 outline a management framework for providing forest resources “now and in the future[,]” including “air quality, cave systems, heritage resources, minerals, natural areas, recreation opportunities, scenery, soil, terrestrial and aquatic ecosystems, timber, water, and wilderness” (HNF Plan p. 2-1).

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²October 22, 1976 P.O. 94-588, 90 Stat. 2949, as amended; 16 U.S.C.

Table 1.—Hoosier National Forest goals described in the 2006 land resource management plan (HNF Plan).

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1. Conservation of Threatened and Endangered Species Habitat
 2. Maintain and Restore Sustainable Ecosystems
 3. Maintain and Restore Watershed Health
 4. Protect our Cultural Heritage
 5. Provide for Visually Pleasing Landscape
 6. Provide for Recreation Use in Harmony with Natural Communities
 7. Provide a Useable Land Base
 8. Provide for Human and Community Development
-

Nonnative conifer species were introduced in the HNF from the 1930s to the 1950s to reclaim abandoned agricultural lands, providing erosion control and a fast-growing forest stock. Of the many conifer species present in the forest today, only eastern redcedar (*Juniperis virginiana*), and eastern hemlock (*Tsuga canadensis*) are thought to have native populations within the HNF. Although fast-growing southern pine species met the objectives at the time, perpetual non-native forest stands are in direct conflict with some current goals described in the HNF Plan.

OBJECTIVE

The objective of this study is to assess the impact of existing conifer stands on management goals outlined in the HNF Plan. This assessment ranks HNF stands by forest goal values and the relationships with conifer stand cover. Key questions answered in this analysis are:

1. What is the relationship between forest goal values and conifer cover?
2. What impact would removal of conifer stands in the HNF have on forest goal values?

Spatial layers representing forest goal values were created by linking goal descriptions in the HNF Plan to available spatial data layer attributes. Forest goal values and percent conifer cover (conifer fraction) were evaluated across individual forest stands in the Tell City Management Unit (TCU).

STUDY AREA

This analysis was confined to the managed forest stands of the Tell City Management Unit on the Hoosier National Forest (see Figures 1a, b). Forests of this region occur within the maple-beech to oak-hickory transition zone of the Eastern Deciduous Forest (Braun 1950, Barrett 1995) and generally contain a high diversity of tree species. The topography can be described as hilly, with slopes typically ranging from 10 to 40 percent. Elevations of the sites vary from 190 to 240 m. The 30-year mean annual temperature is about 11 °C, with mean monthly temperatures ranging from –1 °C in January to 25 °C in July. Mean annual precipitation is 1,100 mm, occurring as rain from March to July and snow from December to February (National Climatic Data Center).

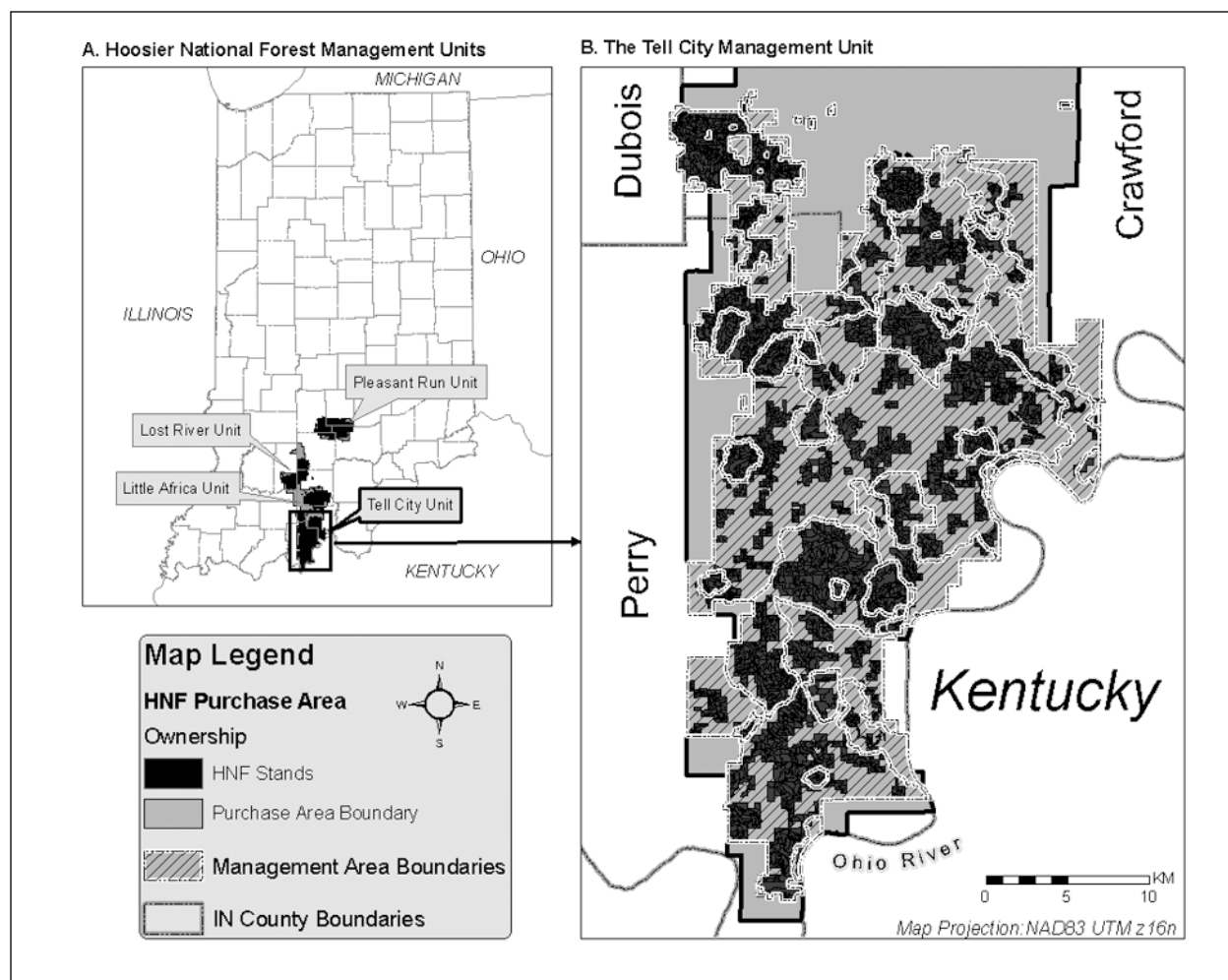


Figure 1.—Geographic units of management within the study area: (A) The Hoosier National Forest in Indiana, and (B) Tell City Management Unit.

METHODS

This analysis used two primary and four tertiary electronic dataset inputs: (1) The HNF stands represent the geographic unit for comparisons between conifer stands and forest goal values. Stand cover types were also an input for forest goal values; (2) Delineated conifer stands represent the most current (2005) and spatially accurate (to within ~ 5 m horizontal accuracy) extent of conifer stands; (3) Ecological land type phase (ELTP) units describe local ecological conditions and are used to characterize ecological goals in constructing forest goal surface values; (4) Management areas; (5) Cave density; and (6) Riparian extent were also utilized to construct forest goal value layers. General spatial summary statistics for all data inputs are presented in Table 2.

DIGITAL LANDSCAPE CHARACTERIZATION

Management Areas and Units

Figure 1 illustrates many levels of geographic management boundaries. Management units (black areas in Figure 1A; USDA FS 2007a) divide the forest latitudinally into four major geographic sections. The purchase area boundary (gray in Figure 1A and B; USDA FS 2007b) defines the congressionally designated geographic

Table 2.—Summary spatial statistics for polygon datasets within the Tell City Management Unit. All area values are in hectares.

	Data Layer	Count	Total Area	Mean Area	STD. Dev. Area
1	*Forest Stands	4,258	32,744	8	8
2	*Delineated Conifer Stands	496	5,192	10	12
3	*Study Area ELTPs	7,947	32,357	4	12
4	Management Areas	74	6,965	941	2,570
5	Cave Density	1	1,366	1,366	0
6	Riparian Areas	1	7,516	7,516	0

* Datasets clipped to HNF stands extent.

area within which HNF authorities can purchase lands to meet forest objectives. Management area boundaries (white dashed outline and hash areas in Figure 1A; USDA FS 2007a) segment the landscape by ecological and social values as they relate to forest goals defined in the HNF Plan (pp. 3-25 – 3-51). Management areas include private and HNF properties.

Properties owned by the HNF are divided into stands (dark areas in Figure 1B) based on forest cover type and local site conditions. Stands were created by the USDA Forest Service³ in 1998 to represent the smallest geographic unit for which management decisions are made. Thirty-two forest types are identified in the TCU. Forest goal values and conifer fraction were evaluated for each stand in the TCU as the primary unit of analysis for this study. Stand type was also used as an input for calculating forest goal values based on stand descriptions and forest types as they relate to goals and conditions described in the HNF Plan.

Delineated Conifer Stands

Conifer stands for the HNF were delineated from high-resolution infrared aerial imagery as part of a project conducted for HNF⁴. Within the TCU, 496 conifer stands were identified (see Table 3 for spatial summary statistics). The high-resolution imagery (1 m pixel size) and manual accuracy assessment provided a high level of confidence in the final output for stand size, shape, and location. However, it was not possible to perform species identification with this dataset; stands are simply identified as “conifers”.

Landscape Classification

Ecological classification systems define expected ecological conditions on the landscape by ecosystem productivity or expected response to management activity (Bailey 1983). The introduction of the National Hierarchical Framework of Ecological Units (NHFEU) established a system for the spatial segmentation of ecosystems in the United States (Cleland and others 1997). The NHFEUs range from continental ecoregions to stand-sized land units called Ecological Land Types (ELTs) and smaller ELTPs.

An ecological classification system created for the HNF defines 16 ELTPs for the TCU (Zhalnin 2004, Zhalnin and Parker 2009). ELTP descriptions are based on statistical comparisons between topographic conditions and forest types across sampling sites in management units (Zhalnin and Parker 2009).

³ Stands dataset on file with the Hoosier National Forest Stands, Bedford, IN (contact authors).

⁴ Image Matters Inc. May 24, 2009. “Report on Phase II Results of Conifer Stand Delineation of the Hoosier National Forest.” Contact author Jeff Ehman.

Table 3.—Input dataset attribute contributions to forest goal value surfaces.

Input Layer (Attribute)	Forest Goal	Positive Impact Areas	Negative Impact Areas
HNF Stands (Cover type)	FG 1 FG 2	Cover type includes a listed species Specified habitat type	Nonnative forest cover type Single species
ELTP (ELTP)	FG 1 FG 2	Ciffs Match forest cover type	-- --
Cave Density	FG 1 FG 2	> 0 caves/square km > 0 caves/square km	-- --
Management Area	FG 1 FG 2	Description supports T/E Natural and diverse character	Support recreation development --
Riparian	FG 1 FG 2	-- 100-ft riparian buffer	-- --

Forest Goal (FG) Value Layers

Spatially explicit values associated with forest goals were derived for the study area. Forest goal value layers were created by linking forest goal descriptions listed in the HNF Plan with available spatial data layer attributes. First, the eight forest goals listed in the Plan (see Table 1) were analyzed for relationships to forest cover type. It was determined that only FG 1 (Conservation of Threatened and Endangered Species Habitat) and FG 2 (Maintain and Restore Sustainable Ecosystems) were directly impacted by the presence of conifers. Next, available spatial datasets were selected based on relationships with goal descriptions. Spatial attributes were ranked by whether their relationship was positive, neutral, or negative (1, 0, -1, respectively) in achieving FG 1 or 2.

Spatial data inputs used to derive forest goal surfaces were selected based on availability, scale, and interpretability toward forest goals. Forest goal value surface input data layers were as follows: the HNF stands dataset, HNF Management Areas (USDA FS 2007a), ecological land types for the HNF (Zhalnin 2004), Indiana streams (U.S. Geological Survey and U.S. Environmental Protection Agency 2008a), Indiana water bodies (U.S. Geological Survey and U.S. Environmental Protection Agency 2008b), and cave densities (Indiana Geological Survey 1997). Layer inputs were summed across the study area, converted to positive integers, divided by polygon area, then normalized to derive spatially explicit (per unit area) independent index values ranging from 0 to 1 for forest goals 1 and 2. Descriptions of spatial data inputs and attribute rankings are presented in Table 3.

The value layers for FG 1 were constructed under the following assumptions related to threatened and endangered (T/E) species. The U.S. Department of Interior, Fish and Wildlife Service has identified five federally listed species with part or all of their range within the HNF: Indiana bat (*Myotis sodalis*), Gray bat (*Myotis grisescens*), rough pigtoe mussel (*Pleurobema plenum*), and eastern fanshell mussel (*Cyprogenia stegaria*) (all listed as endangered) and the bald eagle (*Haliaeetus leucocephalus*) (listed as threatened) (HNF Plan p. C-1). Forest cover types described in the HNF stands containing any favored roosting tree species (HNF Plan p. 3-3) of the Indiana or Gray bat were ranked positively and nonnative forest cover types were ranked negatively. Cave locations (cave density) and ecological conditions favoring bat habitat (e.g., ELTP cliffs) were ranked positively. No sites specifically noted for the fanshell mussel, rough pigtoe mussel, or the bald eagle are present in the TCU. Management areas described in the HNF Plan as containing habitat directly related to T/E listed species were ranked positively and areas designated for the development of recreational activities were ranked negatively for FG 1.

FG 2 calls for the maintenance of ecosystems that are diverse in species variety, genetic make-up, and ecological processes; the enhancement of sustainability through provision of a variety of habitats focusing on native species, cave, and riparian ecosystems; and the use of prescribed burning to restore natural conditions (HNF Plan pp. 2-3 – 2-4). Forest stands containing nonnative species or pure stands (forest types listed as a single species) are ranked negatively. The value layer for FG 2 was constructed under these assumptions: Forest stand types matching FG 2 habitats listed in the HNF Plan or that matched expected ELTP descriptions were ranked positively based on expected ecosystem quality. Forest stands listed as single-tree cover types were ranked negatively. Areas where cave density is greater than 1 per square kilometer were ranked positively. Management areas described in the Plan as possessing enhanced habitat or character were ranked positively for FG 2. Due to greater diversity associated with lakes and streams, riparian areas were also ranked positively for FG 2. See Table 3 for a summary of FG value rankings.

STATISTICAL ANALYSIS

The impact of conifer stands on HNF goals was measured at the stand level by examining the relationship between FG 1 and FG 2 and conifer fraction. Figure 2 illustrates the method for calculating forest goal values and conifer fraction for each stand polygon. For stands with a conifer fraction greater than zero, an arcsine transformation was applied and index values were regressed with conifer fraction. Index values for stands with zero conifer fraction were compared to other stands using a Mann-Whitney rank sum test of FG value.

RESULTS AND DISCUSSION

FOREST GOAL VALUE ANALYSIS

Histogram distributions and descriptive statistics for FG 1 and FG 2 are presented in Figures 3A, B. Both distributions reflect a tri-modal statistical artifact of the original integer-based ranking. Index values for FG 1 are more widely distributed throughout the study area, whereas FG 2 values are more concentrated (also reflected in standard deviations). On average, FG 1 values were slightly higher than FG 2, suggesting stands in the TCU may be better suited for preservation of T/E habitat than for sustainability. Comparing the distributions of both classes suggests FG 1 objectives may be met over a larger number of stands; the FG 2 objectives are targeted to more specific stands.

Relationships between FG values and conifer fraction are presented in Figure 4. Forest stands are plotted as points by respective arcsine-transformed forest goal values (Y-axis) and arcsine-transformed conifer fraction (X-axis). Both forest goal indices are inversely correlated with conifer fraction within TCU stands (regression slope statistically < 0). Although the data clouds are dispersed, a trend is evident: forest stands with a higher conifer fraction tend to have lower FG values. Results from the Mann-Whitney rank sum test comparing stands with and without conifers show no significant difference for FG 1 ($p = 0.244$) but do show a significant difference for FG 2 ($p < 0.001$).

Statistical analysis results were convoluted due to the methods used to rank and combine inputs in forming FG index values. The regression analysis demonstrates a trend where lower conifer fraction is associated with higher forest goal values. Furthermore, the non-parametric Mann-Whitney test indicates that FG 2 was significantly different between stands with and without conifers. These results indicate that removal of conifers could improve the value of stands in the TCU for achieving forest goals. However, further analysis is necessary to confirm this assumption.

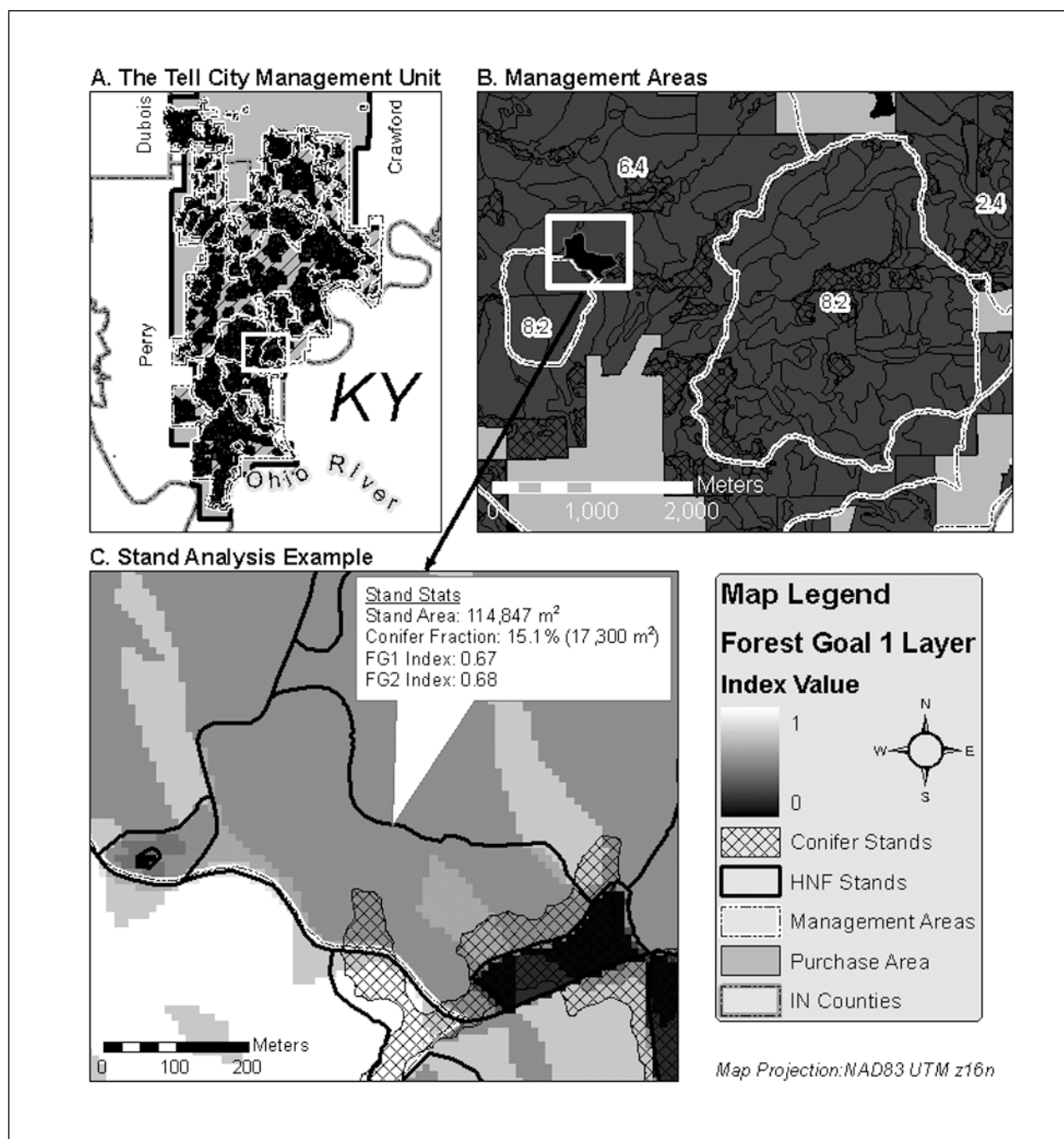


Figure 2.—Example of stand level analysis performed in this study. Multiple scales are displayed to illustrate (A) the TCU study area, (B) relative management areas, and (C) the calculation of stand-level statistics.

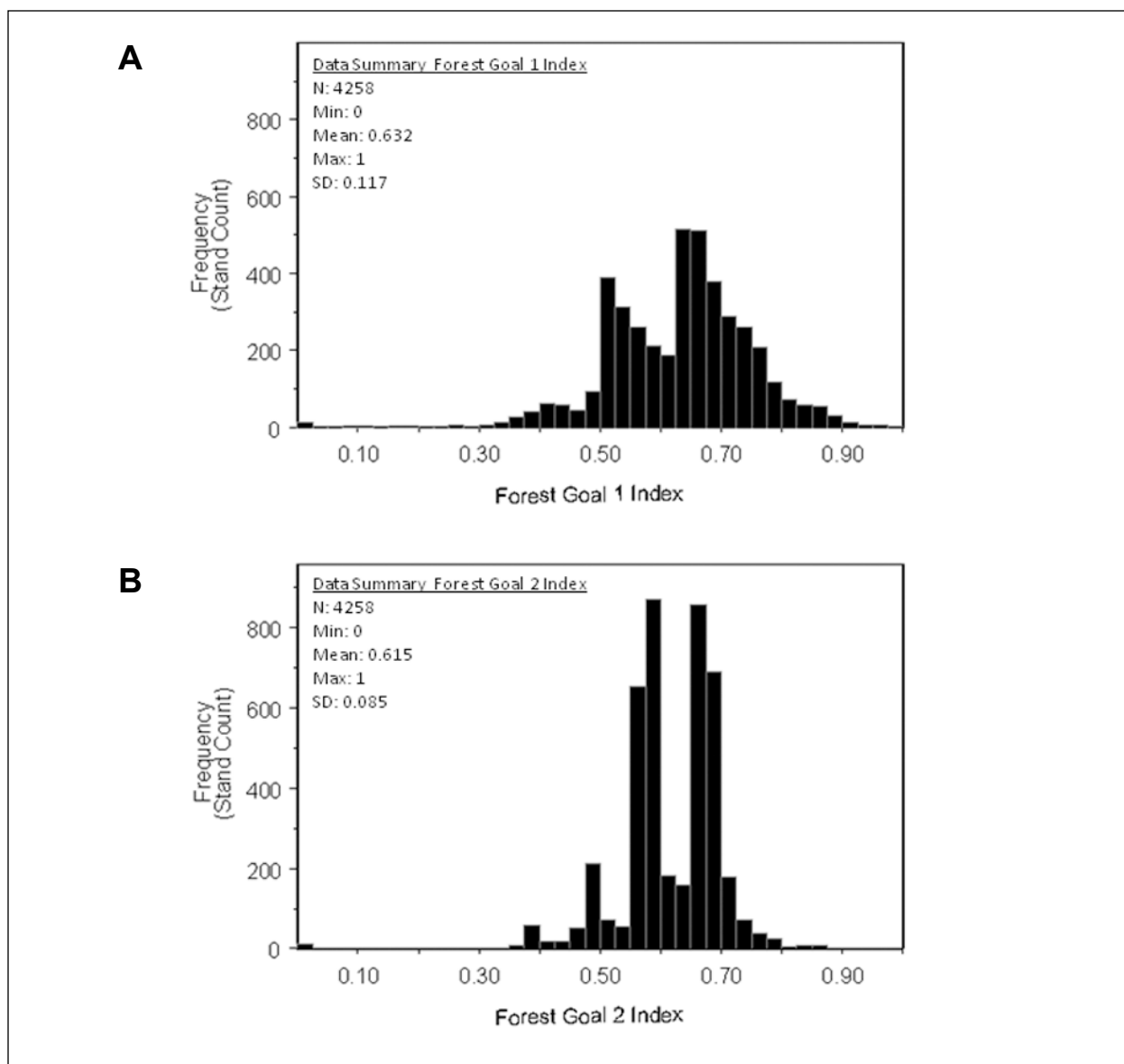


Figure 3.—Frequency distributions and descriptive statistics of forest goal indices across forest stands in the TCU.

CONCLUSIONS

Although this analysis concludes that an inverse relationship exists between land values for some forest goals and the presence of conifers, we do not argue that conifers are “bad” for the forest. Evergreen species provide some ecological and social benefits over hardwoods through year-round ground cover supporting watershed function (FG 3) and desired greenness through the winter (FG 5). Incorporating layers created for additional forest goals could increase the applicability of this analysis. However, a comprehensive study to create spatial layers for all eight forest goals would require extensive data sampling over many disciplines and many spatial scales, and the links between complementary and opposing objectives could confuse rather than resolve the issue of managing conifers. The reasons for examining FG 1 and FG 2 are described above. Sensitivity analysis of inputs and input ranks could quantify the utility of the forest goal evaluation by relating threshold goal values with records of actual management treatments.

Some conifers are native to the study area but were not distinguishable from nonnatives due to remote-sensing processing limitations. As science improves, spatial analysis methods offer possibilities of enhancing

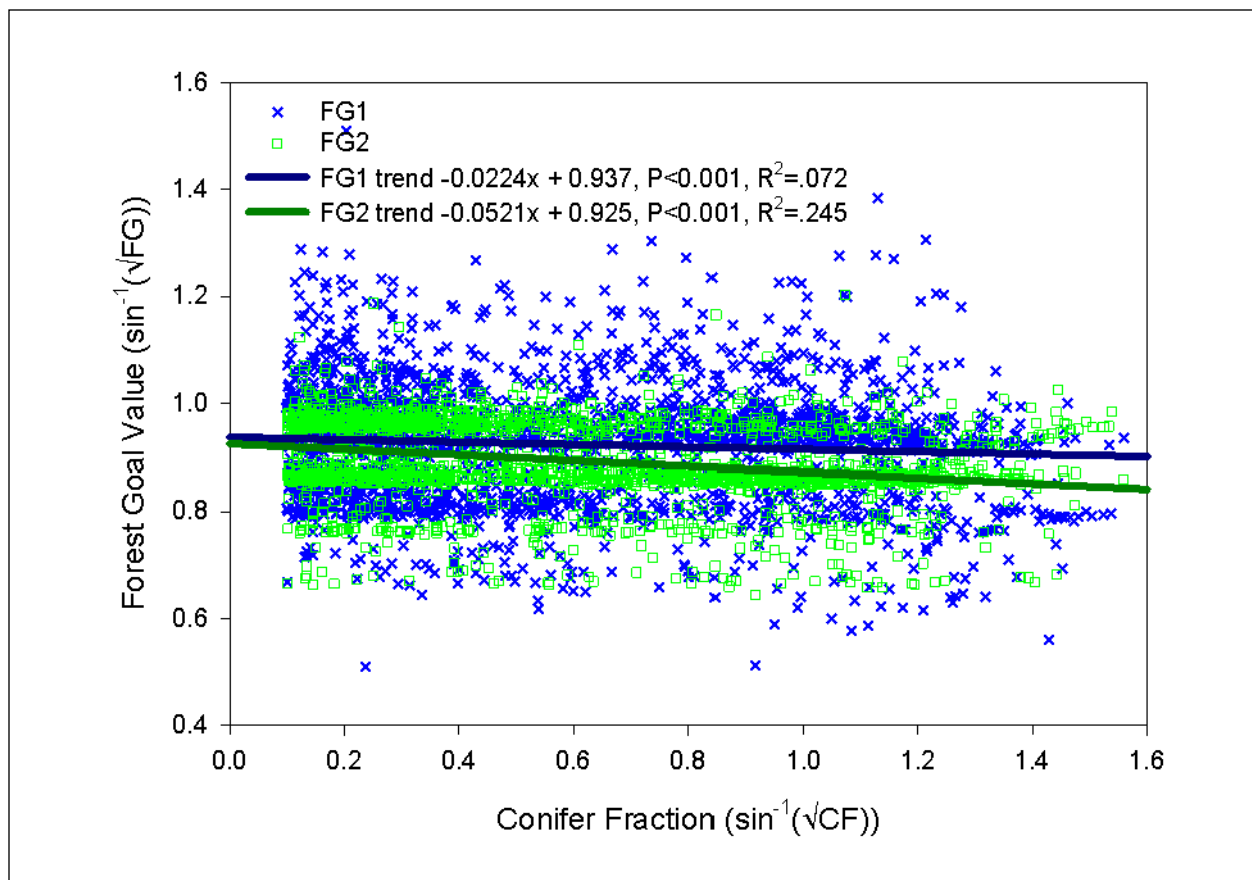


Figure 4.—Forest goal value and conifer fraction plot. Points represent polygons in the TCU. Trend lines represent the results of the linear regression. Data are left transformed to more clearly show the inverse relationship between conifer fraction and forest goal value.

both the identification of stand characteristics such as species and size, and other management considerations important for effective allocation of precious resources.

The primary purpose of this analysis was to demonstrate that forest-wide, spatially explicit information on forest conditions can ultimately inform management decisions. Conifer fraction can be tied with negative or positive landscape impacts, relative to specific management objectives. As spatially explicit management information is developed (e.g., where fire may be used), incorporating information about social (e.g., management area goals) and physical (e.g., presence of conifers) resources will ultimately lead to better informed decisions (e.g., where can minimal burning have maximum impact). This information may be especially important in the HNF, where managers may opt to purchase land to meet their objectives.

The HNF Plan produced in 2006 outlines forest goals that reflect a paradigm shift in forest management in the Central Hardwoods region by defining multiple-use objectives influenced by national, regional, and local stakeholder interests, over purely local objectives. This analysis demonstrates how management actions in the past and present can be linked to future decisions. At the time of planting, fast-growing non-native conifer species provided erosion control on locally degraded sites, and pulp wood for regional markets. Climate-related shifts in species range could change the definition of what is native and therefore the desirability of conifer stands in the HNF. This analysis could provide a value-based assessment of adaptation/mitigation strategies by modeling the ranks of forest types according to expected future conditions.

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ANALYSIS OF HISTORICAL VEGETATION PATTERNS IN THE EASTERN PORTION OF THE ILLINOIS LESSER SHAWNEE HILLS

Suzanne L. Jones and Roger C. Anderson¹

Abstract.—We used General Land Office Survey (GLO) records to reconstruct the historical (1806-1810) vegetation of the eastern portion of the unglaciated Lesser Shawnee Hills Ecoregion of the Shawnee National Forest Purchase Area in southern Illinois. Prairies occurred on 0.4 percent of the area and savannah, open forest, and closed forest occupied 14.5 percent, 25.4 percent, and 59.6 percent of the area, respectively. Average tree density and basal area for the area were 102 trees per ha and 13 m²/ha, respectively. Closed forest occurred at significantly lower elevation than savannah, with average elevation being 139.3 m and 147.1 m for closed forest and savannah, respectively. White oak (*Quercus alba*) and black oak (*Q. velutina*) were the two leading dominant species in all vegetation categories. Savannah had higher importance of xeric species than did closed forest, whereas mesophytic species had higher Importance Values in closed forest. Variation in fire frequency, topography, and moisture availability likely contributed to the landscape vegetation patterns, species composition, and tree density and basal area in the study area. High abundances of oaks and hickories and low tree densities and basal areas throughout the study area indicate that all vegetation types may have experienced periodic fires.

INTRODUCTION

We reconstructed the historical vegetation (1800-1810) of a portion of the Lesser Shawnee Hills in southern Illinois using General Land Office (GLO) survey records. This investigation was part of a larger study that provided information for developing the Shawnee National Forest Plan (Fralish and others 2002).

The Shawnee Hills Natural Division of Illinois (260,620 ha) occurs in an unglaciated portion of Illinois and is divided into two subdivisions, the Greater and Lesser Shawnee Hills. The Greater Shawnee Hills subsection (173,631 ha) is the northern of the two subdivisions and is characterized by a high east-west escarpment of Pennsylvanian-age sandstone. The Lesser Shawnee Hills subsection (86,989 ha) averages 61 m lower in elevation than the more dissected Greater Shawnee Hills, with Mississippian-age limestone as the primary surface bedrock (Willman and Frye 1970, Schwegman and others 1973). Most of the soils were derived from loess that originated primarily from the Mississippi flood plain (Fahrenbacher and others 1967, Willman and Frey 1970). The Lesser Shawnee Hills has thick loess soils along the Mississippi and Ohio rivers and thinner soil in the middle of the state (Willman and Frye 1970, Schwegman and others 1973). Loess deposits vary in depth from more than 7.5 m to 1.2 m or less on scattered rock outcrops (Willman and Frye 1970). The Lesser Shawnee Hills subsection was subdivided into three Land Type Associations (LTAs) based on the thickness

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of the loess deposits: LTA-6 = West Thick Loess, LTA-7 = East Shallow Loess, and LTA-8 = East Thick Loess (Fig. 1). During initial analysis of the witness tree data for the LTAs, the east shallow loess and east thick loess had lower tree density and basal area (88-92 trees/ha and 11-12 m²/ha, respectively) compared to the other LTA in Lesser Shawnee (235 trees/ha and 32 m²/ha) and the Greater Shawnee Hills (195 trees/ha and 24 m²/ha). There were also differences in tree species Importance Values between LTA-7 and LTA-8 compared to the rest of the Shawnee Hills. These two LTAs were studied because of their similarity to each other and the differences between these two LTAs and the remainder of the Shawnee Hills. The GLO witness tree data for LTA-7 and LTA-8 were combined for this study.

The study area is approximately 69,500 ha and was included in the Western Mesophytic region by Braun (1950). The current vegetation is largely characterized by second-growth oak-hickory forest on upland sites and mesic forests dominated by sugar maple (*Acer saccharum*), American beech (*Fagus grandiflora*), and yellow poplar (*Liriodendron tulipifera*) in sheltered locations, with a diversity of tree species occupying bottomland sites (Fralish 1976, 1988; Fralish and others 1991; Fralish 1994; Heikens and Robertson 1994; Ozier and others 2006; Chandy and others 2009). Historically, the upland oak-hickory forests were maintained by periodic fire and frequent timber harvests (Fralish and others 1991, Parker and Ruffner 2004). However, with reduced timber harvest and decreased fire frequency beginning near the middle of the past century, shade-tolerant, fire-sensitive mesophytes, including sugar maple and American beech, are replacing the relatively shade-intolerant oaks and hickories (Fralish and others 1991, Fralish 1997, Zaczek and others 2002, Parker and Ruffner 2004, Ozier and others 2006, Chandy and others 2009).

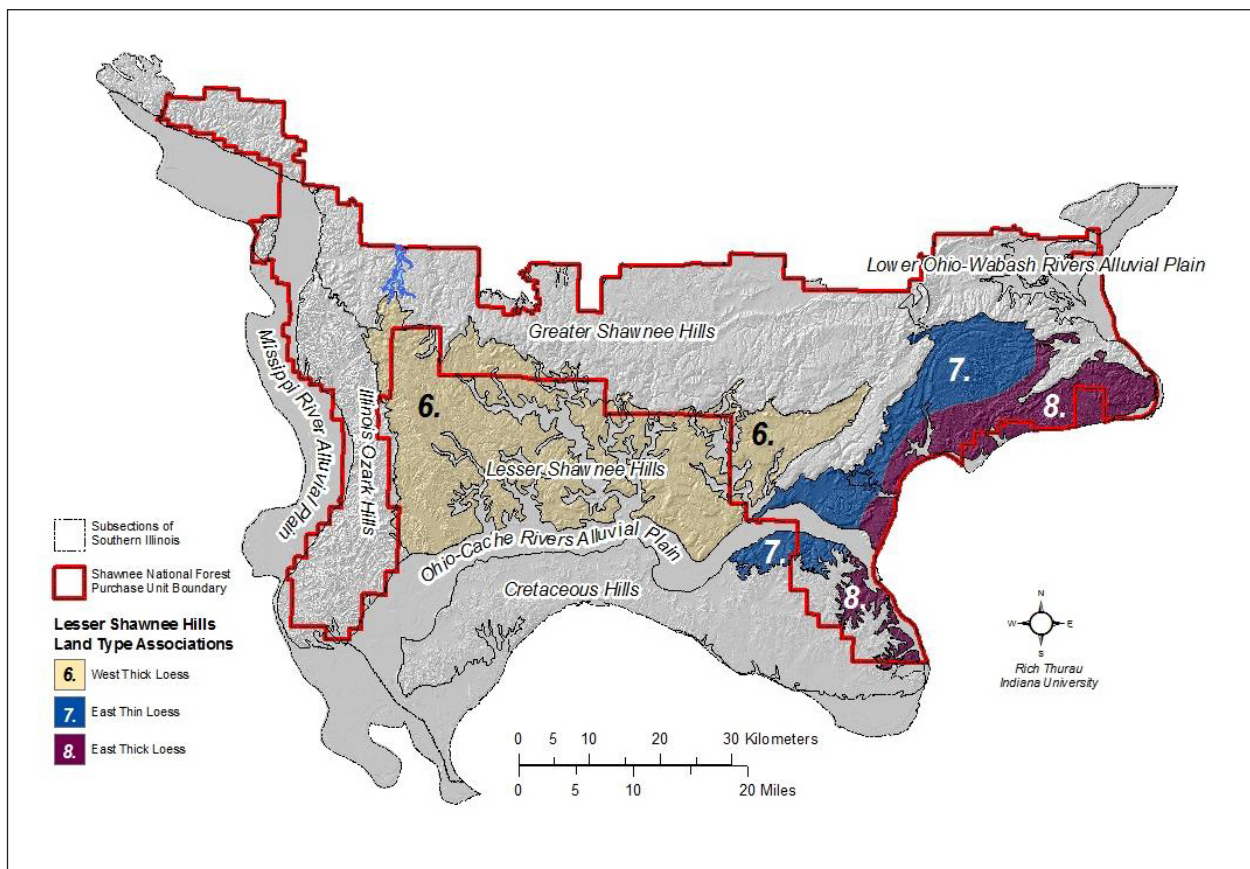


Figure 1.—Land Type Associations (LTA's) for the Shawnee National Forest (Map Produced by Richard Thurau, Indiana University, Bloomington, Indiana). The study areas include LTA7: East Shallow Loess (Lesser Shawnee Hills) and LTA8: East Thick Loess (Lesser Shawnee Hills).

OBJECTIVES

The purposes of this study were to determine: 1) the occurrence of prairie, savannah, open forest, and closed forest vegetation types on the landscape; 2) the composition and abundance of tree species in three vegetation types; and 3) influence of topographic landscape features (slope, aspect, and elevation) on the distribution of three vegetation types.

METHODS

Original survey records are a valuable source of information about historical vegetation. Land surveys were commissioned in 1785 for territories lying northwest of the Ohio River, and were conducted by the GLO before extensive European settlement of the region (Hutchinson 1988). A detailed review of survey instructions and procedures is available in several sources (e.g., Bourdo 1956, White 1983). While some studies have identified limitations in the use of GLO Survey Records for vegetation analysis (Bourdo 1956, Manies and others 2001, Schulte and Mladenoff 2001, Bouldin 2008), it is widely accepted that these records are the best source for historical vegetation data and can be used successfully to determine historical tree density and community composition (Schulte and Mladenoff 2001, Anderson and others 2006).

To calculate the tree densities from the GLO witness tree distance data, the method of Anderson and others (2006) was used. Seven-hundred sixteen points were present in the study area. Outer boundaries of townships are shared with other townships, and to eliminate duplicating measurements, only the southern and eastern boundaries of each township were surveyed. For these survey lines, surveyors tended to select witness trees that were within the township they were surveying rather than in an adjacent one. The average distance for these witness trees was considered to be Q_m (average of $Q_1 + Q_2 + Q_3 + Q_4$) of the quarter method and equal to the square root of the mean area. For points with only one tree or one tree and a corner tree and interior points with two witness trees in the same quadrant or missing directions, the single distance or the shorter of the two distances was treated as the Q_1 distance of the quarter method. Points with no witness trees, points with only a corner tree, or points with missing distances were not used in the calculation of tree density.

For data from interior section or quarter section points with two trees, two distances, and two directions in different quadrants were analyzed by modifying the exclusion angle of the random pairs (Anderson and others 2006). Surveyors measured the distance between the point and the tree, but the distance methods assume that the distance is measured to the center of the tree. Because we calculated tree density at each point, and wanted these data to be as precise as possible, one-half of the witness tree diameter was added to the measured distance. Where distances were given, but no tree diameters recorded, one-half of the average tree diameters for the study area (37 cm) was added to the measured distance. Square root of the mean area and tree density were calculated at each point using the appropriate method for that group (Anderson and others 2006).

Each point was assigned to a vegetation category, based on tree density, according to the categories used by Anderson and Anderson (1975). These vegetation categories were developed by using the GLO survey data from Williamson County, IL and are based on an estimation of the density of average-diameter “open-grown trees” needed to produce canopy cover of <50 percent, >50 percent but <100 percent, and at least 100 percent for savannah, open forest, and closed forest, respectively. In these calculations, it was assumed that the tree crowns do not overlap. The relationship between tree diameter and crown spread was developed by measuring diameter and crown radius of extant open-grown oaks and hickories and using regression analysis to develop a linear relationship between diameter and crown radius, which was used to calculate crown area. Prairies were

arbitrarily defined as having less than 0.5 trees/ha. Points with greater than 0.5 but less than 47 trees/ha were categorized as savannah, and points with 47 to 96 trees/ha or more than 96 trees/ha were categorized as open forest or closed forest, respectively. Within each vegetation category, and the study area as a whole, the tree density and basal area per ha, relative density, relative dominance (basal area) and an Importance Value based on the average of the two relative values were calculated for each species (Anderson and Anderson 1975).

Percent similarity (McCune and Grace 2002) was used to compare species' Importance Values of witness trees in the three vegetation categories. To further assess differences in species importance among vegetation types, species were also grouped according to preferred site characteristics into xerophytic, mesophytic, mid-canopy, and bottomland (Fralish and others 2002). There were not enough mid-canopy or bottomland species in all of the vegetation types to include these groups in the analysis. Xerophytic species included white oak, black oak (*Q. velutina*), hickory species (*Carya* species), post oak (*Q. stellata*), red oak (*Q. rubra*), Spanish oak (*Q. falcata*), oak species (*Quercus* species), and blackjack oak (*Quercus marilandica*). Mesophytic species included gum, elm species (*Ulmus* species), yellow-poplar, blackgum (*Nyssa sylvatica*), white ash (*Fraxinus americana*), sugar maple, sweet gum (*Liquidambar styraciflua*), American beech, maple species (*Acer* species), ash species (*Fraxinus* species), black walnut (*Juglans nigra*), sassafras (*Sassafras albidum*), butternut (*Juglans cinerea*), and hackberry (*Celtis occidentalis*).

Shannon diversity index (H') and evenness were calculated for the three vegetation types using tree Importance Values.

GEOGRAPHIC INFORMATION SYSTEMS (GIS) ANALYSIS

The Digital Elevation Model (DEM) for this project was originally provided by the Natural Resource Conservation Service (NRCS), Major Land Resource Area office and modified by NRCS and researchers at Southern Illinois University-Carbondale prior to our use. The DEM of the Lesser Shawnee Hills is projected in the NAD_1983_UTM_Zone_16N coordinate system. This DEM (resolution 10 m) was used to create elevation, slope, and aspect maps for the study area. A file was created containing tree data (tree species, tree density, and vegetation category) and geographical data (including latitude, longitude, elevation, slope, and aspect) for all sample points. These data were used to create the tree density and vegetation cover maps. It was also used for statistical analysis. To better evaluate the study area, the DEM was used to create an elevation map (10-m resolution). Spatial Analyst (ESRI, Redlands, CA) was used to derive slope and aspect from the DEM. Maps (10-m resolution) were also created for slope and aspect.

The latitude and longitude coordinates and the tree density for each sample point were imported into a gridding software (Surfer, Golden Software, Golden, CO). Sample points were located at 0.81-km intervals along a regular grid. Some densities identified as outliers (>1,500 trees/ha) were removed before proceeding. The tree densities were interpolated into a regular surface using Kriging, with linear variogram (Burrough and McDonnell 1998). The interpolated grid was then exported as an ASCII text file, and imported into ArcView (ESRI) as a table from this format. The table was used to add an event theme (i.e., tree density) to the view with the same projection as the DEM. The inverse-distance weighting method was used to interpolate the event theme, recreating the surface that was previously interpolated in the Surfer software. The density categories were reclassified to reflect the density ranges for vegetation categories defined by Anderson and Anderson (1975). Maps were created showing the location of the sample points and their vegetation category, based on the tree density of each sample point, and these data were projected over an elevation map of the study area (Sauer 2002). Digitized soil data were not available to us for this study.

STATISTICAL ANALYSIS

Importance Values for trees, slope, linearized aspect (Beers and others 1966), and elevation data did not have normal distributions and could not be normalized with transformations (logn, square root, arcsine). Consequently, non-parametric tests were used for analyses. Spearman correlations were used to test for similarities in species' Importance Values among vegetation categories. Chi-square analysis was used to assess differences in Importance Values among the vegetation categories for white oak, black oak, and all other species combined, and for xerophytic and mesophytic species grouped separately. Nonparametric Kruskal-Wallis tests were used to determine whether (1) slope, (2) aspect, and (3) elevation were significantly different among vegetation categories. When the test showed a significant difference among vegetation categories, follow-up Kruskal-Wallis tests were used to perform pairwise comparisons between categories. Statistical significance was accepted at $\alpha = 0.05$ that was adjusted using a Bonferroni correction for multiple comparisons. Data were analyzed using SAS (SAS Institute, Cary, NC).

RESULTS

DISTANCE METHOD APPLICATION-MODIFICATION OF RANDOM PAIRS

Occurrence of witness trees for interior points did not differ significantly from an equal distribution of witness trees in the four quadrants ($\chi^2 = 2.297$, $\chi^2_{0.05, 3} = 7.815$). There were 264, 278, 252, and 284 trees in the NE, SE, SW, and NW quadrants, respectively. The suitability of using the Point Centered Quarter or the Random Pairs Method with an exclusion angle of 180° to determine tree density from the witness tree distances was tested by determining the distribution of second witness trees (second-closest tree to the point) in quadrants opposite or adjacent to the quadrant of the first witness trees (closest tree to the point) (Anderson and Anderson 1975, Anderson and others 2006). However, analysis of second witness tree distribution showed that 395 second witness trees were located in quadrants opposite to the quadrant of the first tree and 144 were located in adjacent quadrants. There is a significant ($\chi^2 = 386.96$, $\chi^2_{0.05, 1} = 3.841$) deviation from the expected distribution of twice as many occurrences of witness trees in adjacent than opposite quadrants for application of the Point Centered Quarter Method to the data and to the expected 50 percent in opposite and 50 percent in adjacent quadrants for the random pairs method ($\chi^2 = 116.885$, $\chi^2_{0.05, 1} = 3.841$). Distribution of second witness trees was found to be 73.3 percent in opposite and 26.7 percent in adjacent quadrants with respect to the quadrant of the first witness tree. When we use the relationship $[90^\circ \times (\text{proportion of total trees in adjacent quadrants} / \text{proportion of total trees in the opposite quadrants})]$ (Sauer 2002, Anderson and others 2006), degrees of azimuth in adjacent quadrants for these data becomes: $[90^\circ \times (0.267/0.733) = 32.81^\circ]$, which results in an exclusion angle of 237.19° , $(360^\circ - 90^\circ - 32.81^\circ)$ and a correction factor for the Modified Random Pairs Method of 0.68.

VEGETATION CATEGORIES AND TREE DATA

There were 104 points with tree densities falling within the savannah range. Open and closed forest categories had 182 and 427 points, respectively. The three prairie points, none of which had any witness trees, were excluded from further analysis based on lack of witness tree information and small sample size. Based on the percentage of the points occurring in each vegetation type, excluding the prairie points, the area was 14.6 percent savannah, 25.5 percent open forest, and 59.9 percent closed forest. For the study area as a whole, the most important species were white oak, black oak, and hickory species (Table 1). White oak alone accounted for 44.5 percent of the Importance Value, and together the three species constituted 72.5 percent of the importance. Average tree density and basal area in the study area were 102.4 trees/ha and 13.0 m²/ha, respectively. Thirty tree taxa (species and genera) were recorded in the study area.

Table 1.—Tree mean diameter, relative density, relative dominance, and Importance Values (IV) by species for the study area. Tree density and basal area were 102.36 trees/ha and 13.03 m²/ha, respectively. Overall mean d.b.h. was 34.41 cm.

Species	No. Trees	Mean d.b.h.(cm)	Relative Density	Relative Dominance	IV
<i>Quercus alba</i>	626	40.92	43.75	45.30	44.52
<i>Quercus velutina</i>	252	48.44	17.61	25.55	21.58
<i>Carya species</i>	111	32.10	7.76	4.94	6.35
<i>Quercus rubra</i>	48	55.86	3.35	6.47	4.91
<i>Cornus species</i>	69	18.48	4.82	1.02	2.92
<i>Quercus stellata</i>	42	37.11	2.94	2.50	2.72
Gum	46	32.03	3.21	2.04	2.63
<i>Ulmus species</i>	43	30.73	3.00	1.75	2.38
<i>Liriodendron tulipifera</i>	29	34.69	2.03	1.51	1.77
<i>Nyssa sylvatica</i>	27	30.29	1.89	1.07	1.48
<i>Fraxinus americana</i>	24	31.62	1.68	1.04	1.36
<i>Acer saccharum</i>	21	33.99	1.47	1.05	1.26
<i>Liquidambar styraciflua</i>	15	44.55	1.05	1.29	1.17
<i>Fagus grandifolia</i>	10	46.04	0.70	0.92	0.81
<i>Morus species</i>	10	31.17	0.70	0.42	0.56
<i>Acer species</i>	9	35.14	0.63	0.48	0.55
<i>Fraxinus species</i>	10	30.73	0.70	0.41	0.55
<i>Platanus occidentalis</i>	3	77.94	0.21	0.79	0.50
<i>Juglans nigra</i>	8	32.65	0.56	0.37	0.46
<i>Quercus falcata</i>	6	37.26	0.42	0.36	0.39
<i>Sassafras albidum</i>	6	23.60	0.42	0.14	0.28
<i>Quercus species</i>	3	32.95	0.21	0.14	0.18
<i>Gleditsia triacanthos</i>	2	44.47	0.14	0.17	0.16
<i>Juglans cinerea</i>	3	22.91	0.21	0.07	0.14
<i>Cercis canadensis</i>	2	24.10	0.14	0.05	0.09
<i>Ostrya virginiana</i>	2	20.95	0.14	0.04	0.09
<i>Celtis occidentalis</i>	1	30.48	0.07	0.04	0.06
<i>Quercus nigra</i>	1	30.48	0.07	0.04	0.06
<i>Acer negundo</i>	1	25.40	0.07	0.03	0.05
<i>Quercus marilandica</i>	1	15.24	0.07	0.01	0.04
Total		34.41	100	100	100

The three leading species in importance in the savannah were white oak, black oak, and hickory species, representing 88.4 percent of importance (Table 2). Post oak was fourth in importance (compared with sixth in the study area), representing 2.8 percent of the Importance Value. Fifteen tree taxa occurred as witness trees in the savannah vegetation. Tree density and basal area were less than the study area as a whole, 19 trees/ha and 3 m²/ha, respectively. In the open forest, white oak, black oak, and hickory species were the three leading species and represented 77.4 percent of Importance Values (Table 3). These species were followed by red oak (*Q. rubra*) and post oak. Average tree density and basal area were low in open forest: 71 trees/ha and 11 m²/ha, respectively. Twenty-two taxa were represented at open forest sites.

The average tree density and basal area for closed forest points was 230 trees/ha and 26 m²/ha, respectively (Table 4). Twenty-eight taxa were represented in the closed forest, with white oak, black oak, and hickory species as the leading three taxa in Importance Value, representing a cumulative Importance Value of 67.6 percent. Post oak was the ninth species in order of importance (Importance Value = 2.11).

Table 2.—Tree mean d.b.h., relative density, relative dominance, and Importance Values (IV) by species for savannah. Tree density and basal area were 19.28 trees/ha and 2.47 m²/ha, respectively. Mean d.b.h. for savannah trees was 35.62 cm.

Species	No. Trees	Mean d.b.h. (cm)	Relative Density	Relative Dominance	IV
<i>Quercus alba</i>	98	42.02	59.04	57.76	58.40
<i>Quercus velutina</i>	37	48.36	22.29	28.89	25.59
<i>Carya species</i>	9	33.76	5.42	3.42	4.42
<i>Quercus stellata</i>	5	40.00	3.01	2.67	2.84
<i>Quercus rubra</i>	2	62.84	1.20	2.64	1.92
<i>Cornus species</i>	5	20.22	3.01	0.68	1.85
<i>Acer species</i>	1	63.50	0.60	1.35	0.97
<i>Ulmus species</i>	2	28.06	1.20	0.53	0.87
<i>Acer saccharum</i>	1	45.72	0.60	0.70	0.65
Gum	1	35.56	0.60	0.42	0.51
<i>Platanus occidentalis</i>	1	35.56	0.60	0.42	0.51
<i>Quercus falcata</i>	1	22.86	0.60	0.17	0.39
<i>Liriodendron tulipifera</i>	1	20.32	0.60	0.14	0.37
<i>Morus species</i>	1	20.32	0.60	0.14	0.37
<i>Sassafras albidum</i>	1	15.24	0.60	0.08	0.34
Total		35.62	100	100	100

Table 3.—Tree mean d.b.h., relative density, relative dominance, and Importance Values (IV) by species for open forest. Tree density and basal area were 70.71 trees/ha and 10.66 m²/ha, respectively. Mean d.b.h. for open forest trees was 37.63 cm.

Species	No. Trees	Mean d.b.h. (cm)	Relative Density	Relative Dominance	IV
<i>Quercus alba</i>	145	42.74	44.75	42.68	43.72
<i>Quercus velutina</i>	72	52.14	22.22	31.54	26.88
<i>Carya species</i>	26	36.55	8.02	5.60	6.81
<i>Quercus rubra</i>	12	54.64	3.70	5.77	4.74
<i>Quercus stellata</i>	15	39.77	4.63	3.82	4.23
<i>Ulmus species</i>	7	32.94	2.16	1.22	1.69
<i>Fagus grandifolia</i>	5	45.11	1.54	1.64	1.59
Gum	8	23.52	2.47	0.71	1.59
<i>Cornus species</i>	8	20.89	2.47	0.56	1.52
<i>Platanus occidentalis</i>	1	121.92	0.31	2.40	1.35
<i>Liriodendron tulipifera</i>	5	31.32	1.54	0.79	1.17
<i>Fraxinus americana</i>	4	41.11	1.23	1.09	1.16
<i>Acer saccharum</i>	4	26.00	1.23	0.44	0.84
<i>Nyssa sylvatica</i>	3	29.95	0.93	0.43	0.68
<i>Quercus species</i>	2	36.14	0.62	0.42	0.52
<i>Acer species</i>	1	30.48	0.31	0.15	0.23
<i>Fraxinus species</i>	1	30.48	0.31	0.15	0.23
<i>Morus species</i>	1	30.48	0.31	0.15	0.23
<i>Quercus nigra</i>	1	30.48	0.31	0.15	0.23
<i>Ostrya virginiana</i>	1	25.45	0.31	0.10	0.21
<i>Acer negundo</i>	1	25.40	0.31	0.10	0.21
<i>Juglans nigra</i>	1	20.32	0.31	0.07	0.19
Total		37.63	100	100	100

Table 4.—Mean tree diameter, relative density, relative dominance, and Importance Value (IV) by species for closed forest. Tree density and basal area were 230 trees/ha and 26.09 m²/ha, respectively. Mean d.b.h. for closed forest trees was 33.14 cm.

Species	No. Trees	Mean d.b.h. (cm)	Relative Density	Relative Dominance	IV
<i>Quercus alba</i>	383	39.92	40.83	43.80	42.31
<i>Quercus velutina</i>	143	46.49	15.25	22.17	18.71
<i>Carya species</i>	76	30.21	8.10	4.98	6.54
<i>Quercus rubra</i>	34	55.85	3.62	7.61	5.62
<i>Cornus species</i>	56	17.95	5.97	1.29	3.63
Gum	37	33.49	3.94	2.98	3.46
<i>Ulmus species</i>	34	30.41	3.62	2.26	2.94
<i>Liriodendron tulipifera</i>	23	35.86	2.45	2.12	2.29
<i>Quercus stellata</i>	22	34.46	2.35	1.87	2.11
<i>Nyssa sylvatica</i>	24	30.33	2.56	1.58	2.07
<i>Liquidambar styraciflua</i>	12	49.81	1.28	2.14	1.71
<i>Fraxinus americana</i>	20	29.36	2.13	1.24	1.68
<i>Acer saccharum</i>	16	34.88	1.71	1.40	1.55
<i>Fraxinus species</i>	9	30.76	0.96	0.61	0.79
<i>Morus species</i>	8	32.35	0.85	0.60	0.73
<i>Juglans nigra</i>	7	34.05	0.75	0.58	0.66
<i>Fagus grandifolia</i>	5	46.96	0.53	0.79	0.66
<i>Acer species</i>	7	29.65	0.75	0.44	0.59
<i>Quercus falcata</i>	5	38.17	0.53	0.52	0.53
<i>Sassafras albidum</i>	5	24.94	0.53	0.22	0.38
<i>Juglans cinerea</i>	3	28.89	0.32	0.18	0.25
<i>Gleditsia triacanthos</i>	2	44.47	0.21	0.28	0.25
<i>Cercis canadensis</i>	2	24.10	0.21	0.08	0.15
<i>Platanus occidentalis</i>	1	38.10	0.11	0.10	0.11
<i>Celtis occidentalis</i>	1	30.48	0.11	0.07	0.09
<i>Quercus species</i>	1	25.40	0.11	0.05	0.08
<i>Ostrya virginiana</i>	1	15.24	0.11	0.02	0.06
<i>Quercus marilandica</i>	1	15.24	0.11	0.02	0.06
Total		33.14	100	100	100

Shannon Diversity Index (H') and evenness (J) increased from savannah to open and closed forests (H' = 1.31, 1.79, and 2.09; J = 0.48, 0.58, and 0.70 for savannah, open and closed forests, respectively).

Spearman correlation showed that species' Importance Values were correlated between the vegetation categories in the closed and open forests (df = 28, Rho = 0.733, p < 0.0001); open and savannah (df = 22, Rho = 0.698, p = 0.0002); and savannah and closed (df = 27, Rho = 0.651, p = 0.0001); Bonferroni adjustment = alpha significant at 0.017. Differences between the groups did not occur because of dominant species, but rather were the result of differences in importance occurring among the less-abundant species. Percent similarity between vegetation types was high: 83.4 percent and 75.7 percent between savannah and open and closed forests, respectively, and 84.8 percent between open and closed forest.

Chi-square analysis showed differences in Importance Values of white oak, black oak combined, and all other species combined among the three vegetation types ($\chi^2 = 14.382$, df = 4, p = 0.0062). Open forest showed no significant difference between closed forest or savannah (open vs. closed $\chi^2 = 2.831$, df = 2, p = 0.243; open vs. savannah $\chi^2 = 6.087$, df = 2, p = 0.048; Bonferroni adjustment = alpha significant < 0.017), but there was

a significant difference between closed forest and savannah ($\chi^2 = 12.206$, $df = 2$, $p = 0.0005$). While white oak and black oak were the first and second most important species in closed forest and savannah, they had higher importance in the savannah than in the closed forest (white oak 42.3 in closed forest vs. 58.4 in savannah; black oak 18.7 in closed forest vs. 25.6 in savannah). The “other species” category had higher combined importance in the closed forest (40.0) than in the savannah (16.0).

A significant difference was found among the vegetation categories for Importance Values of xerophytic and mesophytic species groups ($\chi^2 = 13.198$, $df = 2$, $p = 0.0014$). While there was no significant difference between the closed forest and open forest ($\chi^2 = 4.1$, $df = 1$, $p = 0.0429$; Bonferroni adjustment = significant alpha < 0.017) or open forest and savannah ($\chi^2 = 2.668$, $df = 1$, $p = 0.102$), there was a significant difference in Importance Values of xerophytic and mesophytic species when closed forest and savannah ($\chi^2 = 12.2061$, $df = 1$, $p = 0.0005$) were compared. Xerophytic species were somewhat more important in savannah than in open forest and mesophytic species were more important in the open forest sites than in savannahs (Fig. 2). While not statistically significant, there was a pattern showing higher importance of mesophytic species in closed forest compared to open forest, and greater importance of xerophytic species in the open forest compared to the closed forest (Fig. 2).

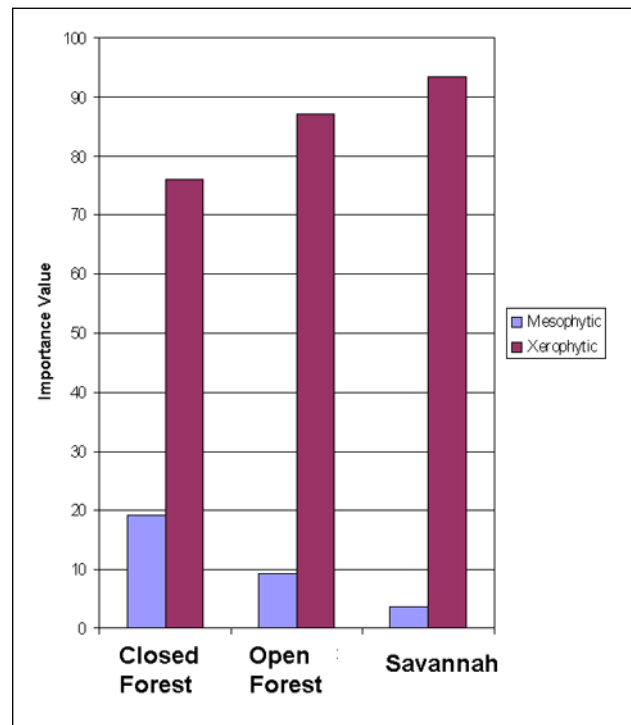


Figure 2.—Sum of Importance Values for mesophytic and xerophytic species in closed forest, open forest, and savannah vegetation categories.

GIS MAPPING AND VEGETATION AND ELEVATION PATTERNS

The study area is not highly dissected, so the slope map showed few steep ridges and there were weak relationships between slope and aspect and vegetation patterns as described later. Historically, the study area was a mosaic of savannah, open forest, and closed forest with the three prairie points located in the extreme southern portion of the study area, as shown in Figure 3.

There were no significant differences among vegetation categories for slope ($\chi^2 = 2.979$, $df = 2$, $p = 0.2255$) or aspect ($\chi^2 = 0.269$, $df = 2$, $p = 0.8741$). There was a weak significant difference in elevation among vegetation categories ($\chi^2 = 5.893$, $df = 2$, $p = 0.052$) (Table 5). Savannah elevations were significantly different from those in closed forest ($\chi^2 = 5.456$, $df = 1$, $p = 0.019$), but open forest elevations were not statistically different from elevations either in savannah ($\chi^2 = 1.927$, $df = 1$, $p = 0.165$) or in closed forest ($\chi^2 = 1.136$, $df = 1$, $p = 0.286$). Savannah occurred at significantly higher elevations (147.1 m) than closed forest (139.5 m) but not open forest (142.6 m); open and closed forests were not significantly different (Table 5).

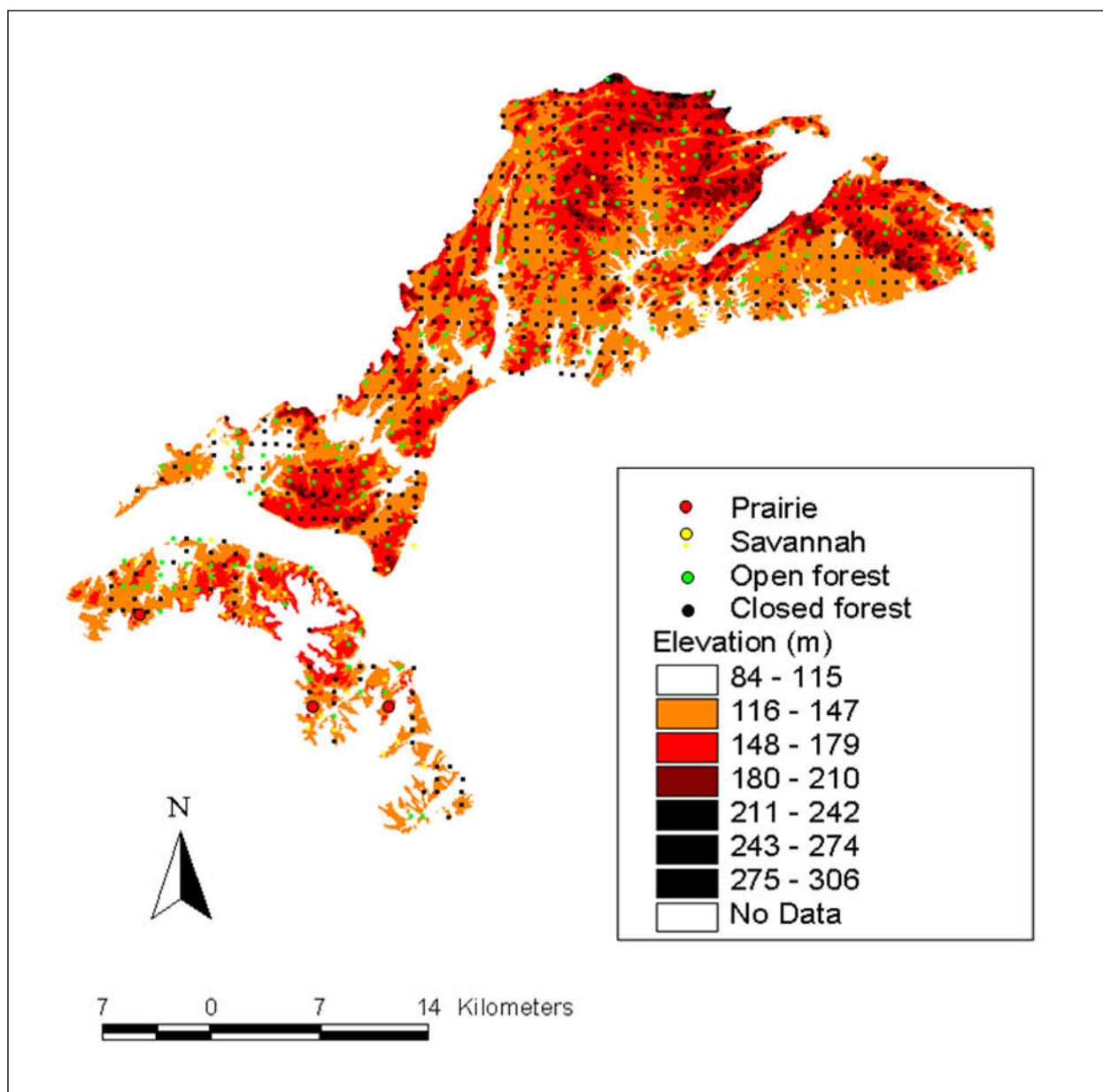


Figure 3.—The distribution of survey points by vegetation categories (prairie, savanna, open forest, and closed forest) in relation to elevation.

Table 5.—Mean values and standard error for slope, elevation, and aspect in each vegetation category.

	Savannah (n=104)	Open Forest (n=182)	Closed Forest (n=427)
Slope (percent)	8.7 ± 0.5	8.8 ± 0.4	9.7 ± 0.4
Elevation (m)	147.1 ± 2.0	142.6 ± 1.7	139.5 ± 1.1
Aspect (degrees)	175.3 ± 10.4	186.9 ± 8.1	181.1 ± 4.9

DISCUSSION

We selected this study area for analysis primarily because of its lower historical (1806-1810) tree densities and basal areas than the rest of the Lesser Shawnee Hills or Greater Shawnee Hills subsection (Fralish and others 2002). Historic vegetation and topography patterns within the study area suggest factors that may have influenced the lower tree density in the area. The average tree density for this study area was 102 trees/ha, which is just within the closed forest category of 96 trees/ha. Forty percent of the points in the area had tree densities within the savannah and open forest ranges, again indicating that open woodlands and savannahs were an important component of vegetation of the area. Post oak, which is characteristic of barrens (Taft 1997) and dry sites (Fralish 1976), has slightly higher Importance Values in the study area than in Greater Shawnee Hills, where tree density is higher (Fralish and others 2002). The study area, especially savannah and open forest sites, was dominated by oak-hickory vegetation rather than species characteristic of more mesic, mixed woodlands. The tree density (303 to 631/ha) and basal area (15 to 29 m²/ha) reported by Fralish and others (1991) for current forests in the Shawnee Region occurring on north- and south-facing slopes and ridgetops are greater than we found for the historic forests. Three interrelated factors would have likely contributed to maintaining this low tree density and basal area: fire, site topography, and moisture availability.

Fire was likely an important factor in maintaining the low tree densities and basal areas of this area. Oaks and hickories, in particular, are resistant to fire and dependent upon periodic fire for their maintenance (Abrams 1992, Heikens and Robertson 1994, Nowacki and Abrams 2008). Fire hinders the growth of non fire-tolerant species, like beech and sugar maple, and reduces the overall tree density. Several studies have shown that fire was an important factor in maintaining oak savannahs and open woodlands (Reich and others 1990, Abrams 1992, Delcourt and Delcourt 1997, Anderson and Bowles 1999, Nowacki and Abrams 2008). Native Americans used fire (Fralish 1997, Parker and Ruffner 2004), as did early European settlers in southern Illinois (Miller 1920, Parker and Ruffner 2004) as a means of clearing land for farming and increasing forb and grass growth. One of the reasons for the lack of a strong relationship between elevation, aspect, and slope with tree density or historic vegetation type could be the landscape homogeneity of the study area. The topography of the study area differs from much of the Shawnee Hills in that it is more rolling and less dissected. While sheltered narrow ravines in the Greater Shawnee Hills would have supported mesic communities, it is likely that the moderately rolling topography of this study area would have allowed fire to travel across most of the landscape (Anderson 1991). Widespread occurrence of fire could account in part for the lower tree density and basal area of the study area compared to the Greater Shawnee Hills.

While variation in fire frequency on the landscape could account for differences in tree density and species composition availability, species composition and dominance can be strongly influenced by moisture availability. However, we presented no evidence for soil moisture as a factor in determining vegetation patterns within our study area. Nevertheless, the distribution of tree species and vegetation communities has been shown to be related to a soil moisture gradient in the Shawnee Hills (Fralish 1976, 1988, 1994). In our study, savannahs occurred on average at higher elevations and slightly more level topography than closed forests. Savannahs appeared to be more common on ridge tops than were closed forests, which were associated more with ravine and valley habitats. For example, 21.1 percent of closed forest points occurred at elevations less than 120 m in elevation, compared with 18.2 percent for open forest and 6.6 percent for savannah. In contrast, 8.7 percent of savannah points were at elevations greater than 180 m, compared to 5.5 percent and 4.5 percent of open and closed forest points, respectively. Higher moisture availability along the lower slopes, in ravines, and along riparian areas may account for areas of higher species diversity and tree densities, as well

as for the presence and importance of mesic and bottomland species in closed forests. Lower moisture would have been associated with upper hilltops and ridges, where soil depth was shallow and favored low-density forests with a higher proportion of xerophytic species than in lower-elevation closed forests (Fralish 1976).

The vegetation mosaic that occurred on the study site at the time of the GLO surveys was likely the result of a combination of interacting factors, including soil moisture, fire, and topography. However, the fire disturbance regime in southern Illinois has changed in the past 55-65 years after the establishment of the Shawnee National Forest and institution of fire control practices (Heikens and Robertson 1994, Fralish 1997, Anderson and others 2000, Parker and Ruffner 2004, Fralish and McCardle 2009). Barrens were present in the study area at the time of the survey, especially at the southern end (Hutchinson and others 1986), and remnants can still be seen today. Nevertheless, studies conducted during the past 40 years have shown that these open forest and barrens undergo transition to closed forest unless fire is used consistently as a management tool (Anderson and others 2000). Similarly, numerous studies have shown that the fire-tolerant oak-hickory community shifts to a community where mesic species like sugar maple increase in importance after fires are suppressed (Lorimer 1985, Abrams 1992, Shotola and others 1992, Zaczek and others 2002, Nowacki and Abrams 2008). For example, Chandy and others (2009) reported that sugar maple in the understory of forests in the Shawnee Region increased (mean $\pm$ SE) from 36 ± 116 to 775 ± 1387 stems/ha over a 3- to 6-year period and sugar maple was the third most abundant tree (basal area = 2.6 ± 4.5 m²/ha), although white oak was the dominant tree species (5.9 ± 10.9 m²/ha). We found that the basal area of sugar maple was 0.1 m²/ha and white oak basal area was 5.9 m²/ha.

The total basal area for all species was 17.9 ± 1.7 m²/ha in the current forest (Chandy and others 2009), whereas we found a total basal area of 13.0 m²/ha in the historic forest. Similarly, in a protected old-growth forest sampled in permanent plots established in 1935, sugar maple density increased from 156 stems/ha to 346 stems/ha and basal area increased from 1.0 m²/ha to 6.8 m²/ha between 1935 and 1997, whereas white oak density declined from 111 to 28 stems/ha and basal area increased from 4.7 to 5.5 m²/ha (Zaczek and others 2002). The authors concluded that the old-growth forest without disturbance for 100 years is succeeding to sugar maple and other shade-tolerant species, even though yellow-poplar and white oak are the two leading species based on relative dominance.

In a process termed “mesophication,” by Nowacki and Abrams (2008), the increased abundance of mesophytic species alters the landscape in such a way that the fire regime is changed so that fires occur less frequently and the microclimate is changed to favor shade-tolerant mesophytic species. This pattern has been documented in the Shawnee Hills and other portions of southern Illinois (Fralish and others 1991, Shotola and others 1992, Ozier and others 2006, Chandy and others 2009). Mesophication has progressed on some portions of the landscape to the point where fire management in the absence of other silvicultural practices, such as cutting, is no longer an effective tool to restore historic conditions (Anderson and others 2000, Zaczek and others 2002, Ozier and others 2006).

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

FOREST REGENERATION

FATE OF THE 2001 ACORN CROP AT CLEAR CREEK STATE FOREST, PENNSYLVANIA

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Abstract.—One of the key steps in the oak regeneration process is the successful germination of acorns into new seedlings. Several factors can greatly reduce or entirely destroy a red oak acorn crop between seed fall in the autumn and germination the following spring. In 2001, a bumper acorn crop occurred on Clear Creek State Forest in Jefferson County, PA. This event coincided with the installation of a root development study and the establishment of an oak silviculture short course for SILVAH. This fortuitous coincidence allowed for a series of three simple experiments to evaluate factors affecting acorn overwintering success and subsequent seedling survival. The crop was estimated at approximately 210,000 acorns per acre for all oak species. Insect infestation of the acorns varied among stands, with infestation rates ranging from 5 to 20 percent. Overwintering success of unburied sound acorns was 20 percent, but burial increased survival to more than 80 percent. Unburied acorns succumbed to desiccation (48 percent), insect infestation (15 percent), disease (15 percent), and consumption by wildlife (2 percent). Seedlings that germinated from acorns outside of deer fences and in dense understory shade had the lowest survival rate (2 percent) during the next 8 years. Excluding the deer or reducing the dense understory shade increased survival rates to approximately 26 percent during the same period. Seedlings in fenced, partial-shade stands had an 8-year survival rate of 56 percent. These data indicate that: 1) oak management activities need to be flexible and opportunistic to take advantage of mast years; 2) there is a several-year window for management following such an event; and 3) dense understory shade and excessive deer browsing are two major obstacles to establishing new oak seedlings, an essential step in the oak regeneration process, at Clear Creek State Forest.

INTRODUCTION

Oak (*Quercus* spp.) forests throughout the eastern United States regenerate through a multi-step process that often takes 10 to 20 years to complete (Sander 1990, Johnson and others 2002). Generally, this process consists of acorn production, seedling establishment from acorns, development of the seedlings into competitive-sized reproduction, and a timely, adequate release of that reproduction (Loftis 2004). Various factors can retard or stop any step in the process, but establishing new oak seedlings from acorns seems especially prone to disruption. For example, acorns and new seedlings can be killed by insects, pathogens, weather, or wildlife. Consequently, the occurrence of mast years is often critical to overwhelm these potential obstacles and start the oak regeneration process in many stands.

Numerous studies, including several conducted in Pennsylvania, have quantified acorn production, documented production periodicity through time, and identified the causes of acorn losses. Sharp (1958) showed that while acorn production is positively correlated with crown size and exposure to sunlight, spring

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weather has a strong influence on annual acorn production, especially on the formation of bumper crops. Marquis and others (1976) documented that acorn production and consumption vary from stand to stand and some locations inherently have less production and/or more consumption than others. Galford and others (1991, 1992) and Auchmoody and others (1994) found that a suite of insects attacks fallen acorns, even during the winter and early spring months. Steiner (1995) and Steiner and Joyce (1999) reported substantial losses of red oak acorns to wildlife, primarily white-tail deer (*Odocoileus virginiana*).

In 2001, a bumper acorn crop occurred throughout northern Pennsylvania. At Clear Creek State Forest (CCSF), this acorn crop coincided with the start of an oak seedling root development study (Brose 2008) and the establishment of a network of regeneration sampling plots for a SILVAH short course emphasizing oak ecology and silviculture. This coincidence allowed for a series of small studies to document 1) the quantity and soundness of the acorns; 2) the proportional losses of acorns to insects, pathogens, weather, and wildlife during the 2001-2002 winter; and 3) survival of the new oak seedlings when subjected to dense shade, partial shade, deer browsing, and no deer browsing through the following 8 years.

STUDY AREA

Clear Creek State Forest (CCSF) is a 13,500-acre property owned and managed by the Pennsylvania Bureau of Forestry in northern Jefferson County, PA (41°18' N, 79°00' W). This location places CCSF in the Appalachian Plateau Province (Schultz 1999), a dissected plateau characterized by broad, flat-topped hills with steep side slopes. Elevations range from 1200 feet a.s.l. in the river valleys to 1800 feet a.s.l. on the summits. Climate is continental; the summers are warm and humid while the winters are cold and snowy. Average annual temperature is 46.4 °F with an average minimum mean of 14.0 °F in January and an average maximum mean of 78.1 °F in July (Zarichansky 1964). Annual precipitation averages 42 inches of rain and 72 inches of snow distributed evenly throughout the year. Growing seasons average 122 days. Generally, soils are loams derived in place from gneiss, schist, and shale parent material. Consequently, these soils are moderately fertile and strongly acidic. Oak site index ranges from 65 to 80 feet at age 50. CCSF contains three forest types: Allegheny hardwood, mixed oak, and northern hardwood. Of these, the mixed oak type is the most common and occupies approximately 85 percent of the land base, and northern red oak (*Q. rubra*) is the most abundant tree species within this forest type. Other common oaks and associated tree species include white oak (*Q. alba*), chestnut oak (*Q. montana*), black oak (*Q. velutina*), red maple (*Acer rubrum*), black birch (*Betula lenta*), black cherry (*Prunus serotina*), American beech (*Fagus grandifolia*), and eastern hemlock (*Tsuga grandifolia*). Nearly all forest stands are about 100 years old, originating after a catastrophic wildfire in 1903.

METHODS

Twelve mature mixed oak stands were selected for this study. Each stand was 10 to 20 acres with oak composing at least 50 percent of the overstory basal area. Six of the stands had been undisturbed for decades and had basal areas ranging from 120 to 145 ft²/ac. The other six had received the first removal cut of a two-step shelterwood sequence 2 to 5 years earlier. This cut was primarily from below (75:25 ratio) and removed poor quality oaks as well as low value and undesirable trees. Residual basal areas ranged from 75 to 90 ft²/ac. Half of the shelterwood and uncut stands had been previously fenced with 8-foot high woven wire fence to exclude white-tailed deer. This arrangement created four treatments—uncut/unfenced (control), uncut/fenced, cut/unfenced, and cut/fenced—with three replications of each. In all 12 stands, a systematic grid of 12-foot-diameter regeneration plots was installed at a density of one plot per acre for a total of 188 plots.

To estimate acorn production and determine seed viability (Study 1), the regeneration plots in the six uncut stands were sub-sampled using four 1-ft² quadrats per plot. These quadrats were placed systematically at the edge of each plot and the acorns in them were collected between mid-October and mid-November 2001. Because of time and work constraints, collections were done only once. Collected acorns were tallied by oak family (red or white) and immersed in water to separate sound from infested acorns (Olsen 1974). In this method, sound acorns sink while infested ones float. A random subsample of 100 sound acorns per oak family from each stand was dissected to determine the infestation rate because the flotation method will not detect all infested acorns (Olsen 1974).

To determine overwintering losses of red oak acorns (Study 2), three fenced and three unfenced stands were randomly chosen and in each, four 12-foot × 24-foot planting plots were established in early November 2001 (after the majority of acorns had fallen). In each stand, all acorns were removed by hand from each planting plot and the plot was then randomly assigned to one of four treatments: litter (control), hardware cloth, buried, and buried/hardware cloth. In each planting plot, 384 sound northern red oak (NRO) acorns from a single mother tree were uniformly placed at a density of 16 acorns per square yard. In the control treatment, the acorns were placed among the leaf litter to simulate naturally fallen acorns. The hardware cloth treatment was prepared the same as the control except ¼-inch wire screen was placed on top of each plot and fastened to the ground with sod staples to exclude predation by wildlife. In the buried treatment, the leaf litter was raked from each plot; then it was tilled with a front-tine rototiller to scarify the soil. The acorns were then placed approximately 1 inch into the soil. The buried/hardware cloth treatment was prepared the same as the buried treatment except ¼-inch wire screen was placed on top of each plot and fastened to the ground with sod staples to exclude predation by wildlife. Acorns that had not germinated by July 2002 were collected and dissected to determine whether they died from disease, insects, or weather. Acorns containing larvae or signs of prior insect infestation (exit holes, frass) were classified as having been killed by insects. Acorns that had begun to germinate, but stopped and had fungal mycelium inside the shell, were classified as having been killed by disease. Weather-killed acorns were those that failed to germinate, but had no evidence of insect infestation or fungal infection. Missing acorns were assumed eaten by wildlife.

To determine the impact of deer browsing and dense understory shade on the long-term survival of the new NRO seedlings (Study 3), they were tallied annually from 2002 to 2009 in late summer in each of the 188 regeneration plots.

For the acorn production and viability study, analysis of variance (ANOVA) with the Student-Keuls mean separation test (SAS Institute, Inc. Cary, NC, 2000) was used to detect differences among the six stands regarding the average number of acorns by family and by soundness. For the acorn overwintering study, the proportion of acorns germinating in each planting plot was calculated. These data were arcsine transformed and ANOVA was used to detect differences among the eight treatments (buried, buried/fenced, in litter, in litter/fenced, buried/protected, in litter/protected, buried/fenced/protected, and in litter/fenced/protected). In addition, the proportion of acorns that failed to germinate was calculated for all the planting plots in unburied treatments. These data were arcsine transformed and ANOVA was used to detect differences among the four mortality agents (disease, insect, weather, and wildlife). For the seedling survival study, the annual survival rate relative to the initial (2002) seedling counts was calculated for each of the 12 stands, and then repeated-measures ANOVA was used to test for differences among the control, no deer, reduced shade, and no deer/reduced shade treatments. For all tests, residuals were analyzed to ensure ANOVA assumptions were met and alpha was 0.05.

RESULTS

Stand-level acorn production in fall 2001 varied among stands (Table 1). Average acorn production ranged from $169,000 \pm 36,000$ acorns per acre at Spring Creek to $247,000 \pm 41,000$ acorns per acre at Pine Run. However, these differences were not significant ($p = 0.0641$). Acorn production differed between red and white oaks within stands and reflected the composition of the overstory trees. The Edeburn, Jimtown, and Lonestar stands were dominated by chestnut and white oaks and their acorns accounted for 61 to 94 percent of the acorns collected. Conversely, Pine Run and Spring Creek were red oak stands and those acorns accounted for 69 percent of the acorns collected. McNeil had an approximately even mix of red and white oaks and the ratio of their acorns was 51:49. There was no difference in the distribution of sound acorns among stands as acorn soundness ranged from 80 to 95 percent. Likewise, no differences were found in the soundness rate between red and white oak acorns in any of the stands. The flotation method did a good job separating sound from unviable acorns as the subsampling of sound acorns revealed an insect infestation rate between 1 and 3 percent.

Overwintering survival of red oak acorns was affected only by whether the acorn was buried or in the leaf litter on the forest floor (Fig. 1). Acorns placed in the leaf litter had overwintering survival rates of 15.7 to 22.6 percent while buried acorns survived at significantly higher rates (73.0 to 91.5 percent). Addition of deer fencing and/or hardware cloth did not alter overwintering survival rates, regardless of whether the acorn was in the litter or buried in the forest floor. Examination of the red oak acorns placed in the leaf litter that failed to germinate revealed differences among the four mortality agents (Fig. 2). Winter weather, specifically desiccation, killed 48 percent of the acorns, the most of any of the mortality agents. Disease and insects each accounted for 15 percent of the overwintering losses. Two percent of the acorns were not recovered and their removal is attributed to consumption by wildlife.

Deer fencing and reducing dense understory shade affected survival of the new red oak seedlings from 2002 to 2009 (Fig. 3). Seedlings in dense shade and exposed to deer browsing had the lowest survival rate, regardless of year; by 2009, only 2 percent of the seedlings were still alive. There was no difference in seedling survival rates between fencing and partial shade removal, regardless of year, as both increased seedling survival rates relative to the control. This difference became evident in 2005 and persisted until 2009, when the average survival rate of the two treatments was 26 percent. The combination of fencing and partial shade removal had the highest survival rate (56 percent) of the four treatments. The difference between fence/shade reduction and the other treatments became evident in 2003 and persisted to the end of the study. Acorn crops from 2002 to 2008 were almost nonexistent and no new oak seedling establishment was observed in any of the plots.

Table 1.—Acorn production (mean $\pm$ 1 SE) of six mixed-oak stands at Clear Creek State Forest in Fall 2001. Means followed by different letters *within a column* are statistically different at the 0.05 level.

Stand	Red oak	White oak	Total	Sound
		1000/acre		(%)
Edeburn	$10 \pm 3d$	$162 \pm 37a$	$172 \pm 38a$	$83 \pm 7a$
Jimtown	$80 \pm 11bc$	$160 \pm 40a$	$241 \pm 38a$	$94 \pm 8a$
Lonestar	$67 \pm 8bc$	$106 \pm 40ab$	$173 \pm 35a$	$80 \pm 5a$
McNeil	$125 \pm 17ab$	$119 \pm 41ab$	$244 \pm 40a$	$95 \pm 7a$
Pine Run	$172 \pm 43a$	$75 \pm 7b$	$247 \pm 41a$	$92 \pm 8a$
Sp. Creek	$118 \pm 30ab$	$51 \pm 14b$	$169 \pm 36a$	$82 \pm 5a$

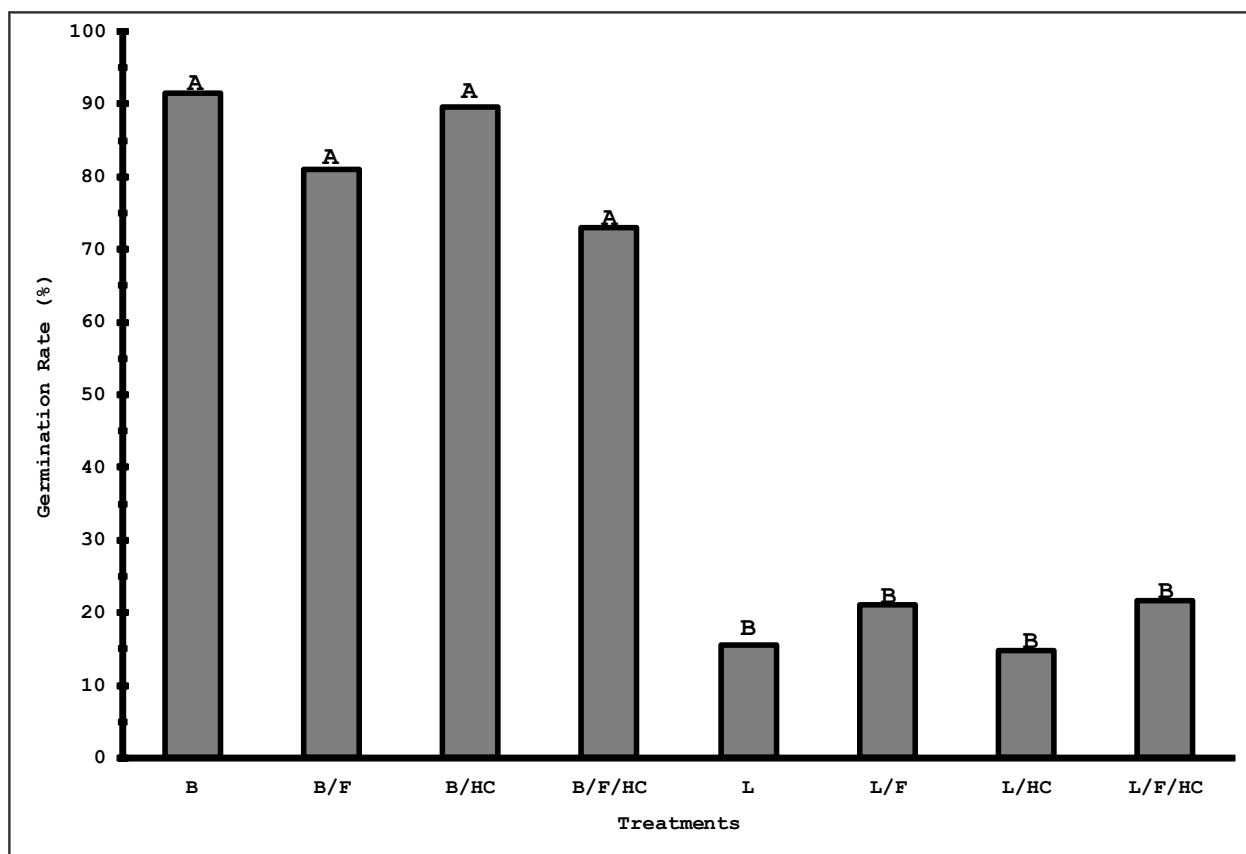


Figure 1.—The effects of position (B and L) and protection from wildlife (F and HC) on the overwintering survival and subsequent germination of red oak acorns at Clear Creek State Forest, Pennsylvania. Treatment abbreviations are: B = Buried in soil, L = placed in leaf litter, F = inside a deer fence, and HC = covered with hardware cloth. Bars with different letters above them are statistically different at the 0.05 level.

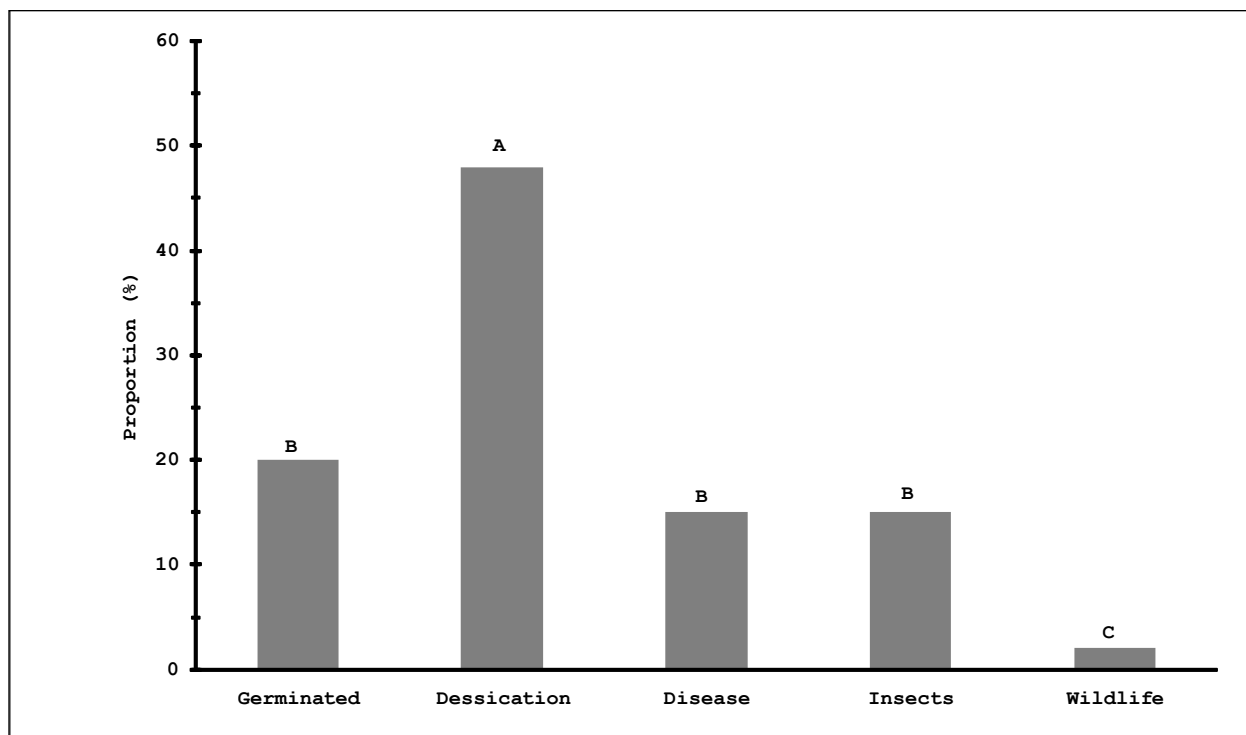


Figure 2.—Overwintering fate (germinated, desiccated, infected by disease, infested by insects, lost to wildlife) of red oak acorns placed in forest floor litter during the 2001-2002 winter at Clear Creek State Forest, Pennsylvania. Bars with different letters above them are statistically different at the 0.05 level.

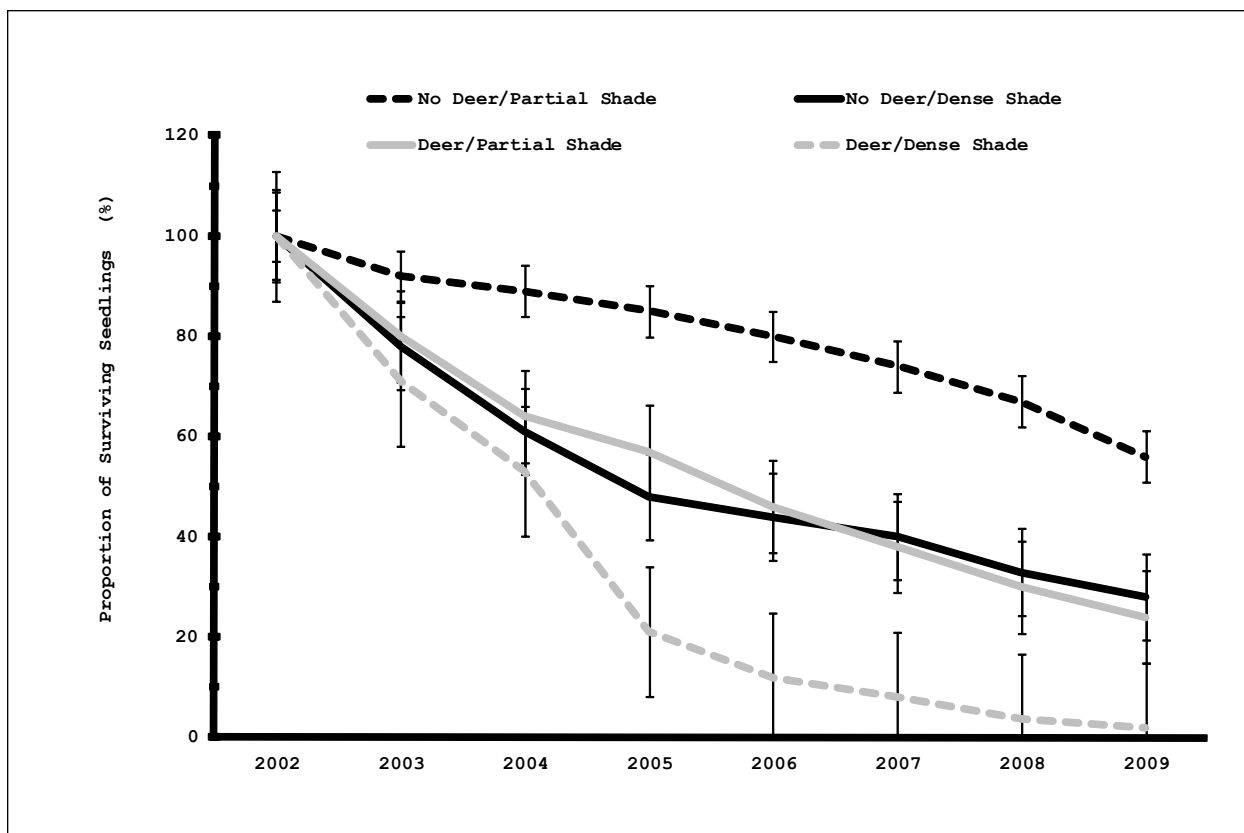


Figure 3.—Survival trends by treatment for red oak seedlings from 2002 to 2009 at Clear Creek State Forest, Pennsylvania. Vertical bars represent one standard error.

DISCUSSION

A critical part of sustainable oak management is the periodic establishment of new cohorts of oak seedlings. Without that occurrence, many oak stands will eventually convert to other forest types because post-harvest sprouting from stumps is usually insufficient to provide enough new stems to ensure a comparable oak stand in the future. Therefore, understanding the factors that limit acorn survival and the ensuing new seedlings is important in prescribing appropriate silvicultural treatments to maintain mixed oak forests. This study reiterates some well known facts about acorn production and survival and places that information in a context important to managers.

The estimate of a 169,000 to 247,000 acorns per acre is conservative. Acorns were tallied only once in each stand, not collected continuously in acorn traps. Counting acorns in quadrats misses those already removed by wildlife and those still in the trees. In a nearby state forest, an oak regeneration study that used seed traps estimated the 2001 mast crop to be 250,000 to 300,000 acorns per acre (pers. comm., Kurt Gottschalk, Research Forester). Furthermore, work scheduling dictated that the Edeburn, Lonestar, and Spring Creek stands were inventoried in mid-October before all the acorns had fallen. Thus, these three stands had fewer tallied acorns than did the other stands.

The small loss of acorns to insects and wildlife is considerably less than the 46 to 87 percent rate reported in other oak literature for Pennsylvania (Marquis and others 1976, Galford and others 1991, Steiner 1995). This

discrepancy is likely due to four factors: 1) the phenomenon of masting (Koenig and Knops 2002); 2) timing of acorn placement; 3) changes in hunting laws; and 4) the probability of consumption. In masting, several consecutive years of poor seed crops cause low populations of acorn insects and small mammals. Then, a mast year literally “floods the market” with more acorns than the baseline insect and wildlife populations can possibly use, ensuring that some of the acorns survive to germinate and become seedlings. This phenomenon was present in this study. The 2001 acorn crop was a bumper crop; the previous mast year occurred in 1991 and there were no or poor crops during the intervening years (pers. comm., Herb Landes, District Forester). Regarding timing, acorns were placed in the plots in early November. By that time, some small mammals such as chipmunks (*Tamias striatus*) and tree squirrels (*Sciurus* spp.) had nearly completed their seed caching and were becoming less active. Moreover, the Pennsylvania Game Commission had recently changed its deer hunting laws, resulting in larger doe harvests. Finally, the probability of consumption is that the chance that wildlife would eat any particular acorn decreases as the total number of acorns increases. With so many acorns on the ground, the probability that wildlife would consume those placed on the plots was small. If the 2001 acorn crop had not been so massive, if the study had started earlier in the fall, or if the deer hunting laws had not changed, the relative losses to insects and wildlife probably would have been much higher.

The finding that only one out of every five red oak acorns germinated even after a mast year was startling, especially given the modest losses to insects and wildlife. This study suggests that desiccation during the winter was the primary culprit. Red oak acorns must maintain at least 30-percent moisture content to remain viable (Olsen 1974, Sander 1990) and dissection of the ungerminated acorns in 2002 revealed that nearly half of them had simply dried out during the 2001-2002 winter. Weather during those months was cold and windy with little snowfall (NCDC 2003). The lack of snow cover allowed the wind to move the leaf litter and expose the acorns, presumably resulting in their desiccation and death. Conversely, buried acorns were protected from desiccation and germinated at greater than an 80 percent rate. This huge disparity in germination rates between buried and unburied acorns indicates that forestry operations such as a timely harvest or site scarification that incorporates freshly fallen acorns into the upper few inches of forest soil may be valuable treatments in protecting acorns and overcoming this obstacle in the oak regeneration process (Zaczek and Lhotka 2004).

Johnson and others (2002) describe oak forests as either intrinsic or recalcitrant accumulators. Intrinsic forests always have oak reproduction present as seedlings or seedling sprouts. Recalcitrant forests have ephemeral oak seedling cohorts that form immediately after bumper acorn crops and then disappear within a few years. At Clear Creek State Forest, browsing pressure of white-tail deer and the degree of dense understory shade determine whether an oak stand is intrinsic or recalcitrant. When both of these factors were present, only 2 percent of the red oak seedlings survived for 8 years. Surviving seedlings were small and nonvigorous, and had underdeveloped root systems. Excluding deer with a woven-wire fence or harvesting the midstory and some overstory trees increased 8-year survival rates to 28 and 24 percent, respectively. Oak seedling development was better than that observed in the control but still unsatisfactory for stand regeneration purposes. When deer were excluded and there was a partial timber harvest, more than half of the 2002 red oak seedling cohort was still alive in 2009. Equally important, these surviving seedlings were developing vigorous roots and stems so they would be able to successfully compete against reproduction of other tree species following a final overstory harvest. Clearly, ongoing forest management that controls deer populations and dense understory shade is better than no management for sustaining oak forests.

Continuous forest management is rarely done on private land, however, it is not even always feasible on public lands. This study offers some insight on how to handle neglected stands. The oak seedlings in the control treatment did not succumb all at once. Rather, the major die-off occurred in 2004 and 2005, suggesting that a management window does exist in chronically neglected stands for a couple of years immediately after a bumper acorn crop. If site preparation work (removing undesirable seed sources, controlling low shade, reducing deer impact) is not possible, then do no timber harvesting until after a bumper acorn crop has fallen. Because such crops are sporadic and unpredictable, advance planning of a timber harvest is not possible. The 2- to 3-year post-bumper-crop management window nevertheless allows sufficient time to complete a timber sale. Again, ongoing forest management is preferable to this reactionary approach, but this post-bumper-crop strategy is clearly superior to timber harvesting with no regard to the oak regeneration process.

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THE EFFECTS OF SHELTERWOOD HARVESTING ON OAK REGENERATION TWO YEARS AFTER HARVEST IN SOUTHERN OHIO

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Abstract.—This research examines the effects of two intensities of shelterwood harvesting (reduction of stocking levels to 50 and 70 percent of full stocking) on oak regeneration in southeastern Ohio 2 years after harvest. The main goal of this study is to develop an understanding of the relationship between residual stocking (harvesting intensity) and the successful release of oak regeneration. The silvicultural treatments were conducted during fall 2005 at Richland Furnace and Zaleski State Forests, which are located in the heavily forested, unglaciated Allegheny Plateau region of Ohio. Analysis of variance was used to compare stem densities of oaks and other woody vegetation among the two treatments and control. The results suggest that oak regeneration was successfully released following the implementation of shelterwood harvests at both residual percentages. Competing vegetation remained a problem, however; further management (such as prescribed fire) may be needed in these stands to increase the competitiveness of oak regeneration. We expect similar responses to shelterwood harvesting in other oak-hickory forests of the central hardwood region.

INTRODUCTION

Oak (*Quercus* spp.) has been the dominant component of the deciduous forests of the eastern United States for at least the past 5,000 years (Fralish 2004) and has been an important component for 10,000 years (Dolan and Parker 2004). During the past century, oak abundance has decreased while the abundance of red maple (*Acer rubrum*) and other hardwood species has increased (Griffith and others 1993, Van Lear 2004). This change in species composition is largely attributable to alterations of the local disturbance regimes and changes in land use. Periodic fires, once common across the forested landscape, favored the regeneration of oaks over red maple and other hardwoods, but the successful implementation of an aggressive national fire suppression policy has shifted forest stand composition toward fire-intolerant species (Abrams 1998). Forests in southern Ohio have not escaped the effects of this policy.

Most natural disturbances in forests reduce the amount of tree canopy and increase the available light for vegetative regeneration. Without canopy disturbance, shade-intolerant species cannot successfully regenerate. While oaks are intermediate in shade tolerance, prolonged shading can lead to seedling mortality (Crow 1988). When small canopy gap disturbances occur, oaks are often out-competed by more shade-tolerant species (Van Lear 2004). When major canopy disturbances occur, however, oaks are out-competed by faster-growing, more shade-intolerant species (Van Lear 2004). These patterns suggest that oaks require intermediate levels of disturbance. Studies by Sander (1979) and Loftis (1983) demonstrate that shelterwood

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cutting techniques, which produce intermediate levels of disturbance, often regenerate oaks on drier uplands. Therefore, the main objective of this study was to evaluate the effectiveness of two different intensities of shelterwood cutting on the establishment and growth of oak regeneration.

STUDY AREAS

Mixed oak hardwood forests located on upland sites in two of southeastern Ohio's state forests served as the locations for this study: (1) Richland Furnace State Forest, Jackson County (39°10'N, 82°36'W) ; and (2) Zaleski State Forest, Vinton County (39°15'N, 82°23'W). Both forests are in the unglaciated hill country of the Allegheny Plateau, where the topography is characterized by deeply dissected terrain (Kerr 1985). The topographic features create a gradient of moisture regimes and microclimates across the landscape (Kerr 1985). Both state forests contain large contiguous tracts of upland oak-dominated forests.

At Richland Furnace, 80 percent of the overstory basal area is composed of upland oak, primarily from the white oak group. Similarly, at Zaleski, 86 percent of the overstory basal area consists of oaks, with the majority of those species in the white oak group. White oak is the dominant tree species within both forests at 44 percent of the total trees per acre. Red maple is the second most common species in terms of trees per acre, 18 percent at Richland Furnace and 27 percent at Zaleski. The red oak group is the third most common tree species, representing 16 percent of the overstory stems at both study locations. Both upland oak stands selected for this study had stocking of approximately 100 percent, or stand density index values (SDI) of 215 (Williams 2003). The plots were located on medium sites (black oak site index 60-75 feet, base age 50 years) with slopes that ranged from 10-24 percent.

METHODS

Two different commercial harvest levels were implemented at each forest during fall 2005, reducing the stocking levels to 50 and 70 percent of full stocking (henceforth referred to as "50-percent treatment" and "70-percent treatment"), which represent the upper and lower bounds of previous management recommendations. These stocking levels are equivalent to SDI values of 110 and 150, respectively (Williams 2003).

The treatments were the first cut in a two-cut shelterwood method. A combined crown thinning and low thinning were implemented to reduce the stocking levels. Dominant and codominant oaks were the favored retention trees while all other species were selected for removal. Since this was a commercial harvesting operation, however, workers did little deliberate marking of noncommercial species. The harvesting in Richland Furnace was carried out via a cut-to-length logging system; Zaleski treatment areas were harvested using cable skidders.

At each state forest we set up three treatment areas: one 50-acre area reduced to 50-percent stocking, one 50-acre area reduced to 70-percent stocking, and one 25-acre control area. The primary reason for the larger size of the canopy reduction treatments compared to the control area is that this study is preliminary to a longer-term project. Eventually, 25 acres of the canopy reduction area at each of the two study locations will be treated with a growing-season prescribed fire and the other half of the areas will receive a dormant-season prescribed fire treatment.

VEGETATION PLOT ESTABLISHMENT AND MEASUREMENTS

Sixteen sample plots were established along transects within each canopy reduction area and eight sample plots were established in the control using a systematic sampling design. All transects were installed parallel to the slope wherever possible. During summer 2005, 40 vegetation plots were established before the silvicultural treatments were initiated to describe overstory and understory conditions. During summer 2007, two growing seasons after the silvicultural treatment, the plots were remeasured. Each sample plot was stratified into three subsample plots based on vegetation size category, and a tally of species was recorded: (1) sapling plot (0.05 acre)—all trees at least 4.5 feet tall and less than 4 inches diameter at breast height (d.b.h.), (2) large seedling plot (0.025 acre)—all woody stems 1 to 4.5 feet tall, and (3) small seedling plot (0.01 acre)—all woody stems less than 1 foot tall.

ANALYSIS

Although 38 species were identified within the vegetation plots, some similar species were grouped for the purpose of the analysis. Red maple (ACRU) was analyzed separately because of the large number of stems per acre, as well as because the competition between red maple and oak has been well documented (Abrams 1998, Fei and Steiner 2007, Nowacki and Abrams 2008). Three hickory species, shagbark, pignut, and mockernut, were placed into the same species group (CASP) because they belong to the same species guild (Sutherland and others 2000). Yellow-poplar was a group (LITU) by itself because few other shade-intolerant species were found in the understory. The non-commercial group (NONCOM) consisted of all woody shrub species. Blackgum was analyzed separately (NYSY) because evidence suggested that deer herbivory can significantly affect blackgum seedlings (Apsley and McCarthy 2004). The other group (OTHER) consisted of tree species with low stem densities. The white oak group (QUAL) consisted of both white oak and chestnut oak, which are both members of the white oak species guild and display similar seed dispersion and growth patterns (Sutherland and others 2000). The red oak group (QURU) consists of red oak, black oak, scarlet oak, and shingle oak. These species were a separate group because they belong to the same species guild and display similar life strategies (Sutherland and others 2000). Finally, sassafras (SAAL) was also analyzed separately due to its high density of regeneration in the preharvest plots.

The objective of the analysis was to determine whether the shelterwood treatments impacted the densities of woody species at each site. The regeneration data were analyzed using analysis of variance (ANOVA) to compare differences in stem densities among species groups within each size class among the three treatment types (50 percent, 70 percent, and control). Tukey's Studentized Range Test was used to indicate significant differences ($p < 0.05$) in mean stem densities among species and treatment groups for each size class. The two study areas were analyzed separately since the objective was not to directly compare the sites to one another. The results of the analysis were used to infer how the treatments impacted the growth of regeneration in terms of relative changes in stem density among species groups.

RESULTS

PREHARVEST DATA

The preharvest data, summarized in Table 1, were used to establish the initial conditions of the treatment plots at each site. The results of the analysis indicate that red maple was by far the most abundant species of regeneration at both sites for most size classes. The most notable exception was the dominance of sassafras

large seedlings at Zaleski compared to all other species. Substantial densities (>100 stems/acre) of small and large seedlings were present for oaks and all other groups at both sites, apart from the few NYSY and LITU large seedlings observed at Zaleski. At both sites, however, densities of oak saplings were much lower than those of other species, especially red maple and black gum.

Analysis of the preharvest data also indicated that mean stem densities were similar across all treatments for most species groups (Table 1). The largest differences occurred for non-commercial and other species, which were of less interest in this study. The most notable difference was that the Zaleski control plots had significantly higher densities of oak seedlings and sassafras saplings than did the 50- and 70-percent treatments. Aside from a few minor differences, the results demonstrate that the initial plot conditions were relatively uniform across treatments; as such, any observed post-harvest differences can be attributed at least partially to effects of treatment.

Table 1.—Summary of preharvest small seedling, large seedling, and sapling stems per acre, by species for Richland Furnace and Zaleski State Forests. Please see text for species' abbreviations. Means followed by the same uppercase letter indicate no significant difference within species group between treatments. Means followed by the same lowercase letter indicate no significant difference between species groups within each treatment (for all tests, $p = 0.05$).

Treatment	ACRU	CASP	LITU	NONCOM	NYSY	OTHER	QUAL	QURU	SAAL
Richland Small Seedling									
50	5,912 Ac	431 Aa	475 Aa	1,656 Ab	506 Aa	1,263 Aab	1,481 Aab	638 Aa	838 Ab
70	11,410 BC	233 Aa	460 Aa	727 Aa	113 Aa	213 Ba	3,513 Bb	1,300 Ba	1,100 Aa
Control	13,840 Be	213 Aa	700 Aab	2,020 Ab	100 Aa	175 Ba	2,850 ABb	738 Aa	1,100 Aa
Zaleski Small Seedling									
50	13,131 Ad	112 Aa	331 Aa	2,894 Ab	363 Aa	906 ABa	1,144 Aa	1,750 Aab	4,530 Ac
70	17,910 Ac	121 Aa	556 Aab	3,131 ABb	206 Ba	606 Aab	1,450 Aab	1,356 Aab	3,620 Ab
Control	8,700 Ac	50 Aa	25 Aa	2,487 Bb	13 Aa	1,975 Bab	2,863 Aab	1,113 Aab	2,900 Ab
Richland Large Seedling									
50	1,120 Ac	182 Aa	192 Aa	1,010 Bc	212 ABa	140 Aa	510 Aab	122 Aa	720 Abc
70	1,067 Ac	101 Aa	245 Aa	325 Aab	248 Ba	613 Ba	552 Abc	160 Aa	835 Ac
Control	1,244 Ab	125 Aa	190 Aa	350 Aa	85 Aa	180 Aa	370 Aa	155 Aa	1,104 Ab
Zaleski Large Seedling									
50	898 Aab	185 Ba	42 Ba	1,082 Ab	65 Aa	320 Aa	280 Aa	422 Aa	3,415 Bc
70	590 Aab	68 Aa	10 Aa	910 Ab	70 Aa	102 Ba	288 Aa	290 Aa	1,680 Ac
Control	720 Aab	170 Ba	0 Aa	1,140 Ab	40 Aa	320 Aa	1,130 Bb	670 Bab	1,795 Ac
Richland Sapling									
50	479 Ac	10 Aa	30 Aa	251 Ab	143 Aab	80 Aa	4 Aa	3 Aa	41 Aa
70	340 Ac	12 Aa	15 ABa	37 Ba	120 Ab	83 Ab	4 Aa	0 Aa	24 Aab
Control	53 Bb	8 Aa	0 Ba	8 Ba	55 Bb	55 Ab	3 Aa	3 Aa	18 Aa
Zaleski Sapling									
50	290 Ac	40 ABa	0 Aa	50 ABab	111 Ab	60 Aab	20 Aa	1 Ba	3 Ba
70	219 Ad	30 Aab	1 Aa	43 Aab	90 Ac	60 Abc	9 Aa	1 Ba	13 Ba
Control	175 Ac	65 Bab	0 Aa	103 Bb	23 Ba	215 Bc	13 Aa	23 Aa	70 Aab

TWO GROWING SEASONS AFTER HARVEST

Analysis of the post-harvest data illustrates the effects of the shelterwood treatments on stem densities for species groups compared to the control (Table 2). Comparison of post-harvest data with preharvest data shows a clear overall decrease in stem density across all small seedling groups for both treatments and the control. This change can be attributed to both seedling mortality and growth into the next class. In contrast with the preharvest data, there were significant differences in small seedling densities between treatment and control plots 2 years after harvest. At both sites, red maple, black gum, noncommercial, and other species were more abundant in the control plots than in the shelterwood treatments. The density of red oak small seedlings was higher for the 70-percent treatment at both sites, while white oaks were less dense. However, a higher density of white oaks in the control plots was also observed preharvest. Additionally, yellow-poplar small seedlings were more abundant in the 50-percent treatment compared to the control, while no changes were observed for blackgum or hickories.

Analysis of the large seedling data show somewhat similar trends. Red maple was the dominant species in all cases except for yellow-poplar in the 50-percent treatment at Richland. Significant differences in large red and white oak seedlings were not observed between treatments at either site, except for a greater number of red oaks in the Richland control. Higher densities of hickories and black gum were also observed in the shelterwood treatments.

The shelterwood treatments appeared to have an effect on sapling densities at both sites. Red maple remained the dominant species at both sites; densities were higher in the 50- and 70-percent treatments compared to the control. However, the density of oak saplings was also higher in the shelterwood treatments than in the control. White oaks were more abundant at both sites in the shelterwood treatments. The control plots at Richland Furnace and Zaleski had a relative absence of red oak saplings both preharvest and two growing seasons after harvest. No significant differences in sapling densities were observed for yellow-poplar, blackgum, or other species.

DISCUSSION

The results of this study demonstrate the effects of shelterwood cutting on woody regeneration in oak-hickory forests. The preharvest data record the overwhelming dominance of red maple regeneration compared to other species but also document the presence of red oak and white oak seedlings.

Overall densities of woody stems decreased 2 years post-harvest in both treatments and the control plots due to seedling mortality and growth into larger size classes. While the composition of the vegetation plots was relatively uniform within each study area during preharvest measurements, significant differences in stem density were observed 2 years after harvest. Red maple remained dominant in all size classes and for nearly all treatments. Additionally, red maple saplings were more abundant in the shelterwood treatments compared to the control. However, even though the treatments appeared to promote growth of red maple saplings, densities of oak saplings also increased significantly in the 50- and 70-percent treatment plots compared to the control.

These results suggest that the shelterwood cutting increased the survival of oak regeneration but did not eliminate competition from other species. Both the 50 and 70 percent of full stocking treatments allowed the oak seedlings to advance in size, which was a main objective of this study. However, the treatments were

Table 2.—Summary of 2 years post-harvest small seedling, large seedling, and sapling stems per acre, by species for Richland Furnace and Zaleski State Forests. Please see text for species' abbreviations. Means followed by the same uppercase letter indicate no significant difference within species group between treatments. Means followed by the same lowercase letter indicate no significant difference between species groups within each treatment (for all tests, $p=0.05$).

Treatment	ACRU	GASP	LITU	NONCOM	NYSY	OTHER	QUAL	QURU	SAAL
Richland Small Seedling									
50	1,213 Ab	194 Aa	4,931 Be	381 Aab	88 Aa	318 Aab	881 Aab	113 Aa	706 Aab
70	2,700 Ab	125 Aa	1,331 Ab	425 Aa	125 Aa	150 Aa	1,813 Ab	634 Ba	1,569 Bb
Control	12,412 Bc	538 Ba	700 Aa	2,087 Bab	38 Aa	300 Aa	3,413 Bb	938 Ba	1,175 ABa
Zaleski Small Seedling									
50	2,463 Ac	113 Aa	1,081 Aa	1,081 Aab	156 Aa	350 Aa	519 Ab	706 Aa	1,325 Aab
70	1,475 Ac	163 Aa	919 Ab	606 Aab	188 Aa	363 Aa	575 Aab	388 Ba	1,313 Ac
Control	4,125 Bc	100 Aa	25 Ba	1,875 Bb	100 Aa	613 Ba	3288 Ba	650 Aa	2,488 Bbc
Richland Large Seedling									
50	1,645 Ac	262 Aa	1,218 Abc	1,772 Ac	825 Aab	310 Aa	795 Aab	210 Aa	885 Aab
70	1,478 Ac	120 Ba	335 Bab	315 Bab	582 Ab	382 Aab	955 Ac	168 Aa	442 Bab
Control	1,325 Abc	100 Ba	165 Ba	685 Bb	30 Ba	225 Aab	530 Aab	300 Bab	1,000 Ab
Zaleski Large Seedling									
50	1,528 Ac	185 Aa	50 Aa	710 Ab	302 Aab	638 Ab	375 Ab	388 Aab	2,025 Ad
70	1,198 Bc	145 Aa	92 Aa	1,112 Ac	370 Aab	368 Bab	548 Ab	398 Aab	1,068 Bc
Control	310 Ca	115 Aa	0 Aa	885 Ab	50 Ba	110 Ca	655 Aa	440 Aab	930 Bb
Richland Sapling									
50	634 Bb	9 Aa	64 Aa	123 Ba	110 Aa	43 Aa	134 Ba	48 Ba	39 Aa
70	508 Bc	43 Ba	28 Aa	68 Abab	84 Aab	48 Aa	140 Bb	29 ABa	6 Aa
Control	88 Ab	3 Aa	15 Aa	8 Aa	60 Ab	83 Ab	0 Aa	0 Aa	65 Ab
Zaleski Sapling									
50	600 Cc	28 Ba	10 Aa	20 Aa	68 Ba	161 Ab	48 Ba	9 Aa	56 Aa
70	355 Bc	9 Aa	14 Aa	57 Aa	29 Aa	128 Ab	44 Ba	38 Aa	33 Aa
Control	143 Ab	40 Ba	0 Aa	260 Ba	45 Ac	60 Aa	3 Aa	33 Aa	45 Aa

also successful in generating substantial ingrowth of red maple and yellow-poplar. These species could retard the establishment and growth of oak regeneration in Zaleski and Richland State Forests. Consequently, other measures aimed at controlling competing vegetation may be necessary in conjunction with shelterwood cutting to secure the advance of oak regeneration. Oaks are highly fire-adapted species, and fire played an important role in the ecology of oak forests in the past, particularly in promoting the dominance of oak in the regeneration layers. Therefore, the use of prescribed fire could be justified to control the competitive vegetation after silvicultural harvesting (Artman and others 2001, Brose and VanLear 1998, Brose and others 1999).

Based on observations, the results from these silvicultural treatments should be typical for similar oak-hickory forests across the central hardwood region. However, it must be understood that before a particular silvicultural treatment is conducted, advance oak regeneration is vital to the overall success of the treatments. Managers should time their shelterwood cutting to coincide either with high levels of oak regeneration or with years of good acorn production. In some areas, shelterwood harvesting may not reduce competing vegetation enough to promote oak species, and the use of prescribed fire, in combination with a silvicultural prescription, should be considered.

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EFFECTS OF MIDSTORY REMOVAL ON UNDERPLANTED BLACK OAK AND WHITE OAK IN THE WESTERN CUMBERLAND PLATEAU

David L. Parrott, John M. Lhotka, and Jeffrey W. Stringer¹

Abstract.—Difficulties in successful oak regeneration have led to the examination of various techniques to increase oak recruitment. To ensure sufficient regeneration, oak seedlings can be underplanted and used in conjunction with intermediate treatments, such as midstory removal, that create a light environment favorable to oak advance reproduction. This study examines the 5-year response of underplanted white oak (*Quercus alba* L.) and black oak (*Quercus velutina* L.) seedlings to a midstory removal on intermediate sites within the western edge of the Cumberland Plateau. Three sites were established and split to provide an area where the midstory remained intact and an area where the midstory was removed. At each site, white and black oak 1-0 bareroot seedlings were underplanted. Midstory removal treatments removed 13 to 19 percent of the basal area in treated units. Seedling height, ground line diameter, and survival status were measured at the time of treatment implementation (spring 2004) and after five growing seasons. Fifth-year survival of white and black oaks under an intact midstory was 51 and 14 percent, respectively. Under removed midstories, white oak survival averaged 81 percent while black oak averaged 32 percent. Midstory removal increased diameter growth for both underplanted white and black oaks. Treatments did not affect seedling height growth of either species. The ground line diameter growth response is consistent with known patterns of oak carbon allocation favoring development of root systems prior to height growth, indicating increased aboveground growth potential.

INTRODUCTION

Periodic disturbances maintained oak-dominated forests of the eastern United States prior to European settlement (Abrams 1992, Brose and others 2001, Nowacki and Abrams 2008). Due to the alteration of historical disturbance regimes, dense midstory canopies of shade-tolerant species such as sugar maple (*Acer saccharum* Marsh.), red maple (*A. rubrum* L.), and American beech (*Fagus grandifolia* Ehrh.) have developed across the eastern hardwood region (Lorimer 1984, Goebel and Hix 1997, Fei and Steiner 2007, Oswalt and others 2008). Understory environmental conditions associated with dense midstory canopies has led to a decline in oak seedling development (Nowacki and others 1990, Lorimer and others 1994, Lockhart and others 2000).

Successful regeneration of oak in the eastern United States has been difficult due to the absence of large advance reproduction (Smith 1993, Lorimer and others 1994, Brose and others 1999). Underplanting can enhance the abundance of oak reproduction, but underplanted seedling development is fundamentally linked

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to the understory microenvironment. Lorimer and others (1994) found light intensities as low as 1 percent full light under developed midstory canopies. Although oaks have intermediate shade tolerance, extended periods without adequate light can result in mortality and loss of vigor (Dey and Parker 1997).

Light levels greater than 20 percent full light may increase growth under forest canopies for some oak species (Gottschalk 1994, Guo and others 2001, Dillaway and Stringer 2006). Midstory removal can create a favorable light environment and enhance recruitment of desired oaks into the overstory (Dey and others 2008). Underplanting and midstory removal have been suggested as an approach to increase advance reproduction in stands lacking sufficient natural seedling densities. Previous studies have focused on highly productive sites with cherrybark oak (*Quercus pagoda* Raf.) (Lockhart and others 2000, Lhotka and Lowenstein 2009) and northern red oak (*Q. rubra* L.) (Loftis 1990, Miller and others 2004). Understanding of how white (*Q. alba* L.) and black oaks (*Q. velutina* L.) respond to these treatments on intermediate-quality sites is limited.

Dillaway and others (2007) initiated a study to evaluate the effects of midstory removal on underplanted white and black oak in 2004. The objective of this paper is to quantify these effects five growing seasons following this treatment.

STUDY AREA

Study sites were located on Berea College Forest in Madison County, KY, and are part of the western rim of the Cumberland Plateau. Three stands of intermediate quality were used: Horse Cove, Water Plant, and Pigg House. Horse Cove is a southeast-facing site with a white oak site index (base age 50 yr) of 22 m, and, at the time of treatment, had a mean dominant tree age of 90 years and 28.0 m²/ha basal area. The Water Plant site has an east aspect and site index of 23 m; it had a mean dominant tree age at time of treatment of 94 years and basal area of 23.7 m²/ha. The third site, Pigg House, faces west, has a site index of 24 m, and had a basal area of 26.0 m²/ha; dominant trees averaged 116 years old at the time of treatment. At all sites, mixed oaks and American beech dominated the overstories while red maple and sugar maple were primary components of the midstory.

METHODS

Within each site, a 0.2-ha area was divided into two 0.1-ha experimental units. Experimental units were randomly assigned one of two treatments: midstory removal or undisturbed control. The midstory removal treatment targeted removal of approximately 20 percent of the total stand basal area. Removals using cut stump treatment with glyphosate started with trees 2.5 cm diameter at breast height (d.b.h.) and progressively moved through increasing size classes until the targeted 20 percent of total basal area was removed or until midstory was completely removed. Treatment retained all dominant and co-dominant trees.

Removed basal area and diameter distribution of removed stems varied among sites due to different stand structures. Removed stems were classified as small (d.b.h. 2.5 to 12.7 cm) and large (d.b.h. 12.7 to 25.4 cm). At Horse Cove, removal of 1.3 m²/ha basal area of small stems and 4.1 m²/ha of large stems amounted to a 19-percent removal of the total basal area. Pigg House had 2.2 m²/ha of small stems and 1.0 m²/ha of large stems removed, resulting in a removal of 17 percent of the total basal area. The Water Plant site had 14 percent of the total basal area removed; small stems accounted for 1.3 m²/ha and large stems accounted for 4.1 m²/ha.

Within each experimental unit, black and white oak 1-0 bareroot seedlings were planted in March 2004. Black and white oak seedling sample sizes varied among experimental units, ranging from 50 to 65 per unit. Initial seedling ground line diameters (gld) and heights were recorded at planting and seedlings were tagged to facilitate long-term monitoring (Dillaway and Stringer 2007).

Seedling height, gld, and survival were reevaluated following five growing seasons (August 2008). Seedling heights were measured with a meter stick held perpendicular to the ground. Tallest shoot height was rounded to the nearest 0.1 cm. Glds were measured at the convergence of the stem and ground using the average of two perpendicular measurements made with digital calipers. Seedlings possessing visible living tissue were considered alive. Average height, gld, and percent survival were calculated for each experimental unit by species. Two-way ANOVA (SAS Institute Inc., Cary, NC) was used to assess the effects of midstory removal treatment and species on 5-year height and diameter growth using site as a blocking variable and an alpha = 0.05. Residual analysis confirmed statistical assumptions of normality and homogeneity of variance were met. Survival data were analyzed using a generalized linear model assuming binomial distribution with a logit link function (SAS Institute Inc.).

RESULTS

Initial seedling gld and height were similar between treatments for both black and white oaks ($p > 0.05$) (Table 1). Black oak seedlings averaged 31.3 cm in height and mean gld was 6.2 mm. White oak heights averaged 24.6 cm, and gld averaged 5.9 mm.

Midstory treatment increased gld growth of black oak ($p = 0.0005$) and white oak ($p = 0.0041$). After five growing seasons, black oak gld growth averaged 0.4 mm in the control and 2.2 mm in the midstory removal treatment while white oak gld growth was 0.6 mm in the control and 1.9 mm under midstory removal. Gld growth was not different between species within treatments ($p < 0.05$). Midstory removal had no effect on height growth of black oaks ($p = 0.1287$) or white oaks ($p = 0.5855$). Height growth for black oaks averaged 1.8 cm in the control and 13.2 cm in the removal treatment (Table 2). White oaks had an average height growth of 7.4 cm in the control and 11.1 cm in the removal treatment. Height growth did not differ between species.

Table 1.—Initial seedling height and diameter (mean $\pm$ standard error) of black and white oak underplanted in the western Cumberland Plateau.

Dependent variable	Black oak	White oak
Seedling diameter (mm)		
Control	6.1 $\pm$ 0.1 ^{a*}	6.1 $\pm$ 0.5 ^a
Removal	6.2 $\pm$ 0.1 ^a	5.7 $\pm$ 0.2 ^a
Seedling height (mm)		
Control	30.8 $\pm$ 1.6 ^a	24.9 $\pm$ 1.0 ^b
Removal	31.8 $\pm$ 0.8 ^a	24.3 $\pm$ 0.4 ^b

* Numbers followed by the same superscript letters indicate no significant difference ($\alpha = 0.05$) between treatments and species.

Table 2.—Seedling measurements (mean $\pm$ standard error) of white and black oaks underplanted in the western Cumberland Plateau after five growing seasons.

Dependent variable	Black oak	White oak
Survival (%)		
Control	14 $\pm$ 1 ^{a*}	51 $\pm$ 15 ^c
Removal	32 $\pm$ 1 ^b	81 $\pm$ 6 ^d
Diameter growth (mm)		
Control	0.4 $\pm$ 0.3 ^a	0.6 $\pm$ 0.2 ^a
Removal	2.2 $\pm$ 0.2 ^b	1.9 $\pm$ 0.2 ^b
Height growth (cm)		
Control	1.8 $\pm$ 4.7 ^a	7.4 $\pm$ 3.3 ^a
Removal	13.2 $\pm$ 8.9 ^a	11.1 $\pm$ 3.1 ^a

* Numbers followed by the same superscript letters indicate no significant difference ($\alpha = 0.05$) between treatments and species.

Black oak survival was increased by treatments ($p < .001$); survival averaged 32 percent under midstory removal treatments and 14 percent in the control. Survival of white oak was also higher in the midstory treatments ($p < .001$), averaging 81-percent survival compared to 51 percent in the control. White oak survival was greater ($p < 0.05$) than black oak in both treatments.

DISCUSSION

The midstory removal treatment applied in this study increased gld growth and survival of underplanted white oak and black oak seedlings. Seedling gld growth was 450 and 217 percent higher in the midstory removal treatment than the controls for black oak and white oak, respectively. White and black oak survival in the midstory removal treatment was approximately 30 and 18 percent higher, respectively. These results follow the trend seen in previous studies that focused on northern red oak (Loftis 1990, Miller and others 2004) and cherrybark oak (Lockhart and others 2000, Lhotka and Loewenstein 2009) seedlings under midstory and shelterwood treatments. It is reasonable to suggest that the improved gld under the midstory removal is resulting in increased root biomass and potentially improved carbohydrate stores in these seedlings (Dillaway and others 2007). Diameter is also a key indicator of seedling ability to support growth after release (Sander and others 1984).

Increased height growth has been observed under shelterwood and midstory removal treatments in previous studies that looked at artificial and natural regeneration of northern red oak (Loftis 1990, Miller and others 2004) and cherrybark oak (Lockhart and others 2000, Lhotka and Loewenstein 2009). In our study, 5-year height growth of white and black oak was not increased by midstory treatment. This outcome could be attributed to factors such as transplant shock generally associated with post-planting moisture and/or nutrient stress (Pope 1993) or a lag time in seedling response to midstory removal as seen by Lockhart and others (2000). Observed growth rates for white and black oak may also reflect carbon allocation patterns associated with oak species, which possess a more conservative growth strategy (Johnson and others 2002), or favored root growth over height growth on intermediate-quality sites where soil moisture may be limiting (Kolb and others 1990).

Research involving northern red oak (Loftis 1990) and cherrybark oak (Lockhart and others 2000) on highly productive sites suggests that midstory removal should precede overstory removal by 7 to 10 years so that oak reproduction can reach a competitive size prior to release. The minimum height for successful release of oak is generally considered to be 1.37 m (Sander 1972). If our observed growth rates continue, average seedling height after 10 growing seasons would be approximately 58 cm for black oaks and 46 cm for white oaks. These data suggest that underplanted white and black oak seedlings on intermediate-quality sites similar to our experimental area may require more time to reach competitive heights following midstory removal than other oak species found on more productive sites. Perhaps seedling height growth rates in our study may have improved if a small percent of the overstory basal area were removed in conjunction with the midstory removal to increase light availability (Loftis 1990).

Increases in seedling gld growth and survival for black oak and white oak indicate that the midstory removal treatment had a positive effect on the underplanted seedlings through five growing seasons. Increased seedling diameter under midstory removal treatment suggests seedlings will have a greater height growth potential in future growing seasons. Continued monitoring of seedlings will help provide insight into the long-term effectiveness of this treatment in improving underplanted black and white oak growth and survival on intermediate-quality sites.

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EARLY SURVIVAL AND GROWTH OF PLANTED SHORTLEAF PINE SEEDLINGS AS A FUNCTION OF INITIAL SIZE AND OVERSTORY STOCKING

John M. Kabrick, Daniel C. Dey, Stephen R. Shifley, and Jason L. Villwock¹

Abstract.—Shortleaf pine was once abundant throughout the Missouri Ozarks and there is renewed interest in its restoration. Past research suggested that the greatest survival and growth of shortleaf pine seedlings occurred where there was little competition for sunlight. This study, in the oak and oak-pine forests of the Sinkin Experimental Forest in southeastern Missouri, quantified the effects of residual overstory density on the early survival and growth of planted shortleaf pine seedlings. We established 48 one-acre experimental units and thinned each from below to establish a uniform stocking level in the range of 0 to 80 percent. In the center of each experimental unit we established 30 1-0 shortleaf pine seedlings on a 12- by 24-foot spacing. One growing season after planting we used regression to analyze seedling (1) survival, (2) basal diameter growth, and (3) shoot growth as functions of initial seedling size and overstory stocking. Increasing overstory stocking nominally decreased seedling survival, but the effect was small and not significant. However, each 20-percent increase in overstory stocking significantly ($P < 0.001$) decreased diameter growth by about 0.01 inches and shoot growth by 0.7 inches. Seedlings with larger initial size had a significantly greater ($P < 0.02$) survival rate.

INTRODUCTION

Shortleaf pines (*Pinus echinata* Mill.) were once dominant on about 4.2 million acres of the Missouri Ozarks and were prominent on an additional 2.4 million acres (Liming 1946). They were most abundant on rolling to steep terrain on soils derived from sandstone or cherty dolomite that were excessively well drained and acidic (Fletcher and McDermott 1957).

Today, shortleaf pine in its native range often accounts for less than 10 percent of stocking (Kabrick and others 2002), having been displaced by black oaks (*Quercus velutina* Lam.) and scarlet oaks (*Q. coccinea* Muenchh.) following extensive timber harvesting and subsequent land clearing and frequent burning during the early 1900s. These two species of oaks are particularly susceptible to oak decline as they mature. Consequently, hundreds of thousands of acres of declining oak forests throughout the Missouri Ozarks are now found on sites where shortleaf pine or pine-oak mixtures were formerly prominent.

There is renewed interest in restoring shortleaf pine throughout its native range in the Ozark Highlands. Restoring shortleaf pine on former pine and oak-pine sites is viewed as a long-term strategy for mitigating

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chronic oak decline (Law and others 2004). There also is an increasing interest in restoring the native oak-pine woodland communities where they once were abundant.

In the Missouri Ozarks, methods for establishing and recruiting shortleaf pine in plantations and natural stands were evaluated from the 1930s through the 1960s. U.S. Forest Service scientists conducted considerable research in the former Clark National Forest, in the Mark Twain National Forest, and on the Sinkin Experimental Forest in southeastern Missouri. This research included studies of natural and artificial regeneration (Phares and Liming 1960, Seidel and Rogers 1965), an examination of the effects of stand density on growth and tree quality (Brinkman and Liming 1961, Brinkman and Rogers 1965), and the development of yield and volume tables for shortleaf pine stands (Brinkman and Rogers 1965, Brinkman 1967). Most of the findings were summarized in two pivotal management publications, *Timber Management Guide for Shortleaf Pine and Oak-Pine Types in Missouri* (Brinkman and Rogers 1967) and *Managing Shortleaf Pine in Missouri* (Brinkman and Smith 1968). These guides emphasized establishing and managing even-aged shortleaf pine-dominated stands. Consequently, guidelines recommended controlling or removing all undesirable hardwoods, particularly in the overstory, using a combination of prescribed fire, herbicides, and mechanical removal with a bulldozer (Brinkman and Smith 1968).

However, foresters are increasingly seeking or are required to select methods for regenerating or re-establishing shortleaf pine other than the ones recommended by Brinkman and Rogers (1967) and Brinkman and Smith (1968). These alternative methods often include leaving moderate to high densities of residual oaks and pines in the overstory of harvested stands either for aesthetic reasons or to meet other natural community or habitat goals.

Past research (Brinkman and Liming 1961) indicates that leaving a residual overstory is likely to reduce the survival or growth of shortleaf pine seedlings. However, there are few quantitative estimates of the magnitude of these survival and growth reductions. The objective of this study was to quantify the survival and growth reductions of shortleaf pine seedlings due to overstory stocking compared to stands having no residual overstory. Findings from this study will allow foresters to anticipate the survival and growth of shortleaf pine seedlings under various levels of overstory stocking to meet desired stocking goals for pine reproduction.

METHODS

STUDY AREA

The study was conducted in the Sinkin Experimental Forest in southeastern Missouri on sites suited for managing shortleaf pine or pine-oak mixes. Study sites were located on summit and shoulder slope positions on soils derived from sandstones (Roubidoux formation) or cherty dolomites (Gasconade formation) and were deep (> 4.9 ft), were moderately-well to excessively-well drained, and had low cation supply. Soil series included Nixa (Loamy-skeletal, siliceous, active, mesic, Glossic Fragiudults) and Coulstone and Clarksville (each Loamy-skeletal, siliceous, semiactive, mesic, Typic Paleudults).

Before the study began, forests were mature and fully stocked. Species in the overstory by proportion of basal area were as follows: black oak (34 percent), white oak (*Q. alba* L., 27 percent), shortleaf pine (17 percent), scarlet oak (6 percent), post oak (*Q. stellata* Wangenh., 5 percent), hickories (*Carya* spp. Nutt., 5 percent), and other hardwoods (6 percent).

EXPERIMENTAL UNITS AND TREATMENTS

We established 48 one-acre experimental units. In the center of each experimental unit we placed a circular 0.2-acre plot for the inventory of all trees ≥ 2 inches in diameter at breast height (d.b.h.) and a circular 0.001-acre plot for the inventory of all trees < 2 inches d.b.h. After pretreatment inventories were completed, the density of each of the 48 experimental units was adjusted to a single stocking level (*sensu* Gingrich 1967, Rogers 1983) ranging from 0 to 80 percent. To achieve the target stocking levels, stems were removed “from below,” creating a shelterwood structure. To facilitate marking to a specified stocking level, a marking guide was created that expressed the target stocking level in units of basal area that could be verified with a 10-BAF prism. Once the experimental units were marked, the merchantable timber was harvested commercially and all other stems marked for removal were felled.

After harvesting, trees ≥ 2 inches d.b.h. were re-inventoried on all experimental units. In the 0.2-acre circular vegetation plots 30 1-0 bareroot shortleaf pine seedlings were hand-planted on a 12-foot by 24-foot spacing during the first week of April. Seedlings were produced by the George O. White State Tree Nursery in Licking, MO. Planting within the 0.2-acre plot ensured a buffer of at least 52 feet from the experimental unit boundary to the nearest planted seedling. After planting, a numbered metal tag attached to a wire was placed next to each seedling so that they each could be relocated for subsequent determination of survival and growth. The initial basal diameter (measured one-half inch above the root collar) and shoot length of each seedling were recorded after planting. The 1,440 seedlings that were planted initially had an average basal diameter of 0.13 inches and an average shoot length of 8.5 inches. One growing season later, the shoot length and basal diameter of all living seedlings were re-measured. We also recorded the condition of each seedling and noted browse or other forms of animal and insect damage.

ANALYSIS

We used logistic regression (LOGISTIC procedure, SAS version 9.1, SAS Institute, Inc., Cary, NC) to analyze the survival probability and the probability of browse occurring as a function of initial basal diameter (and initial height for browse) and overstory stocking. We then used linear regression (REG procedure, SAS version 9.1) to examine the change in basal diameter or shoot length as a function of overstory stocking and the initial basal diameter or initial shoot length. These models included both browsed and non-browsed seedlings. The significant parameters of these models were used to estimate survival probabilities and growth rates for the range of initial basal diameters and shoot lengths of seedlings planted in our study.

RESULTS

The first-year survival rate was 76 percent and the survival probability largely was a function of initial basal diameter (Table 1, Fig. 1). Seedlings with a basal diameter > 0.4 inches had survival probabilities that exceeded 0.90, and those seedlings were 1.2 times more likely to survive than those that had basal diameters < 0.2 inches. On average, seedlings on plots beneath greater residual overstory stocking had slightly higher survival rates, but the effect was small in magnitude and not significant (Table 1).

Increasing the residual stocking of the overstory significantly decreased both the basal diameter growth and the shoot growth of underplanted shortleaf pine seedlings (Table 1, Figs. 2 and 3). Each additional 20-percent increase in overstory stocking decreased the shortleaf pine basal diameter growth by 0.01 inches and the shoot growth by 0.7 inches compared to seedlings grown under no overstory canopy. Seedlings grown under stocking levels greater than 50 percent had decreased basal diameters because of shoot dieback and resprouting. Initial seedling size did not significantly alter the growth response to overstory density (Table 1).

Table 1.—Model parameters^a for estimating survival probability, basal diameter growth (in inches), and height growth (in inches).

Parameter	df	Estimate	Standard error	P-value
Survival probability				
Full model (Hosmer and Lemeshow goodness of fit = 0.10)				
Intercept	1	0.410	0.279	0.142
Initial basal diameter	1	4.851	2.032	0.017
Percent stocking	1	0.003	0.002	0.184
Partial model ^b (Hosmer and Lemeshow goodness of fit = 0.57)				
Intercept	1	0.536	0.262	0.040
Initial basal diameter	1	4.854	2.028	0.017
Diameter growth				
Full model (Adjusted R-square = 0.26)				
Intercept	1	0.063	0.048	0.192
Initial basal diameter	1	-0.080	0.371	0.830
Percent stocking	1	-0.0006	0.001	<0.001
Partial model ^b (Adjusted R-square = 0.28)				
Intercept	1	0.053	0.006	<0.001
Percent stocking	1	-0.0006	0.001	<0.001
Shoot growth				
Full model (Adjusted R-square = 0.38)				
Intercept	1	4.759	1.647	0.006
Initial shoot length	1	-0.076	0.185	0.683
Percent stocking	1	-0.033	0.006	<0.001
Partial model ^b (Adjusted R-square = 0.39)				
Intercept	1	4.093	0.283	<0.001
Percent Stocking	1	-0.033	0.006	<0.001

^aModel form: $Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2$, where Y is the logit of survival probability, diameter growth, or shoot growth, β_0 is the intercept, β_1 is the parameter for initial diameter or shoot length, and β_2 is the parameter for percent stocking.

^bPartial models contain the intercept and significant parameters of the full model and are to be applied by foresters for predicting survival probability, diameter growth, or shoot growth of planted shortleaf pine seedlings.

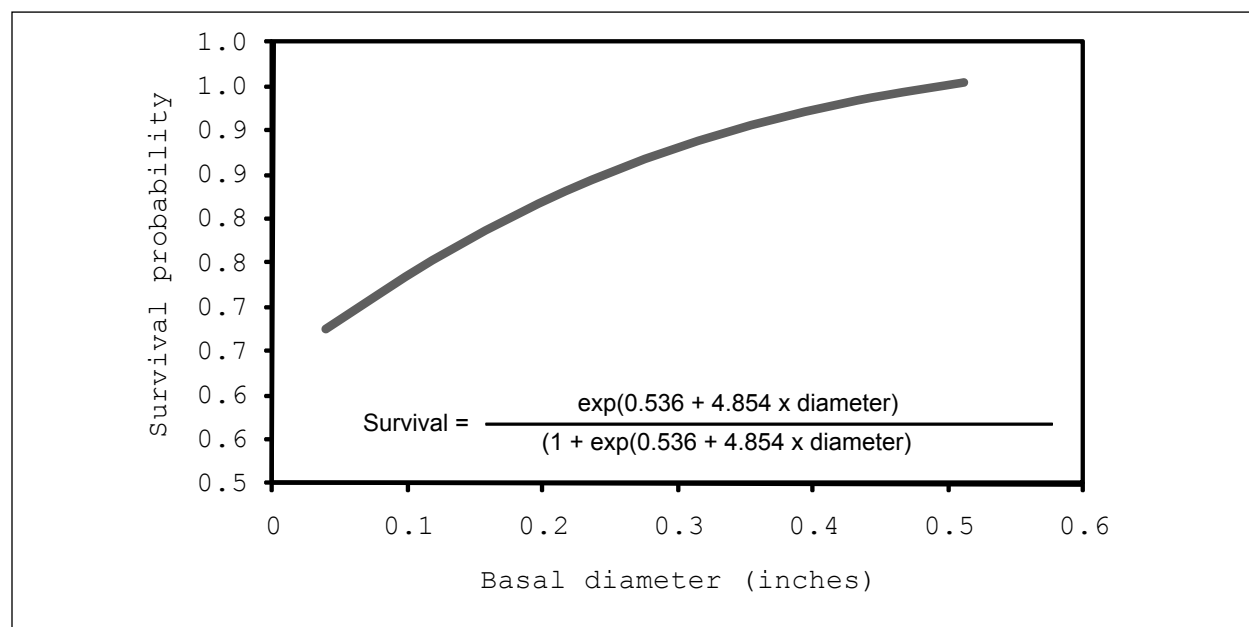


Figure 1.—First-year survival probability of shortleaf pine seedlings as a function of initial basal diameter and overstory stocking.

About 11 percent of the seedlings exhibited some degree of browse damage by animals. The browsing partially removed some of the leader or the ends of the branches. The shoot length of seedlings that were browsed was, on average, about 2 inches shorter than those that were not browsed. Browsed seedlings had significantly ($P < 0.01$) reduced shoot growth compared to those that were not browsed, but the diameter growth of browsed seedlings was unaffected. According to the analysis, browsing was not related to the initial basal diameter or shoot length, or the amount of harvesting (i.e., the residual overstory stocking).

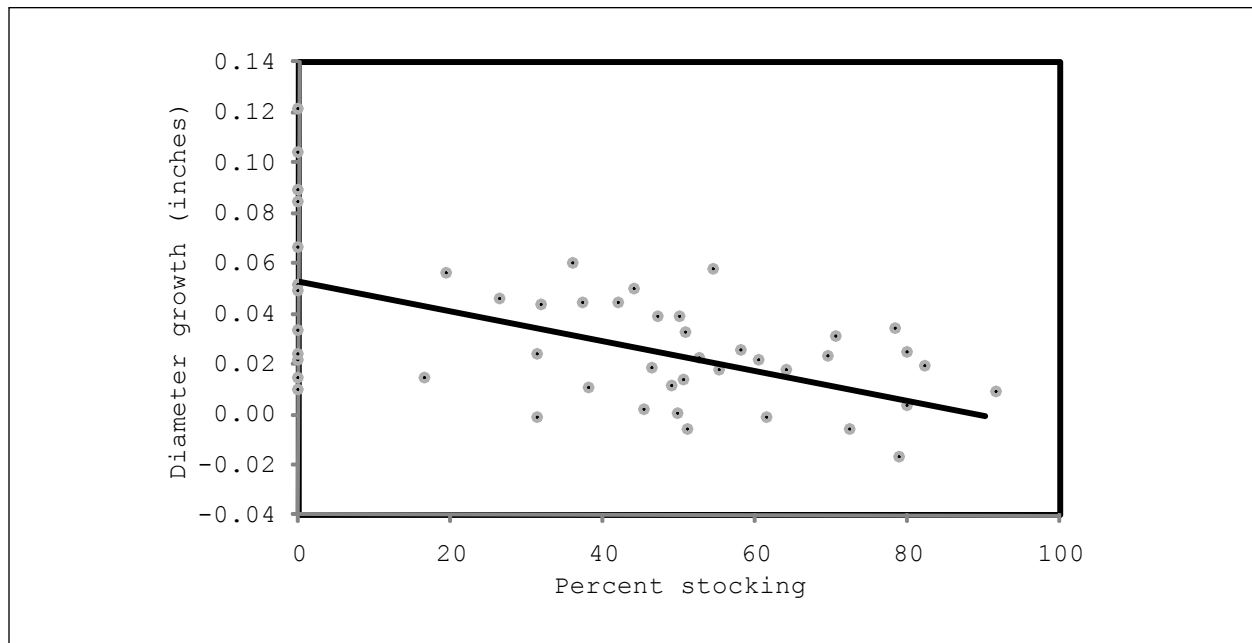


Figure 2.—First-year basal diameter growth as a function of overstory stocking.

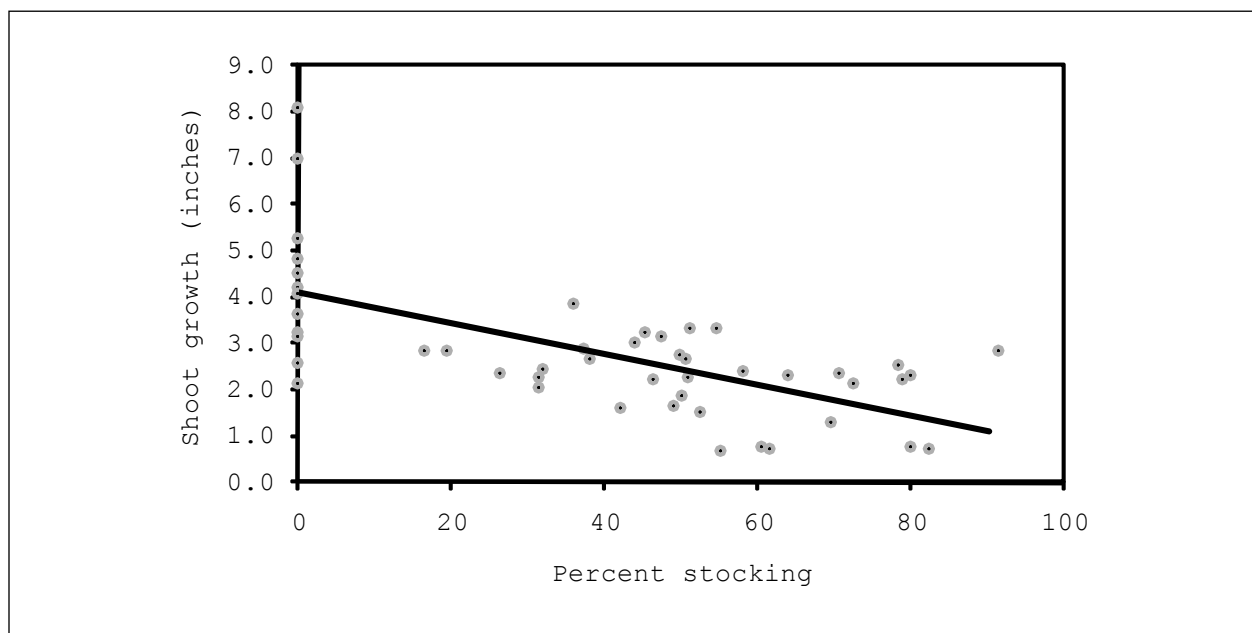


Figure 3.—First-year shoot growth as a function of overstory stocking.

DISCUSSION

The first-year survival rates of shortleaf pine seedlings were consistent with survival rates reported for bareroot seedlings planted elsewhere in the Missouri Ozarks (Gwaze and others 2006) and in the southern United States (Barnett and Brissette 2004). Seedlings with larger basal diameters had significantly greater survival than smaller ones, much like has been reported for oak seedlings (Johnson and others 2009). Larger shortleaf pine seedlings generally have a more balanced root:shoot ratio (Barnett and Brissette 2007), which was reported to increase shortleaf pine seedling survival (Mexal 1992).

We found no significant relationship between overstory stocking and seedling survival. Because of shortleaf pine's shade intolerance, we hypothesized that the increased light levels resulting from a lower overstory stocking (Fig. 4) would increase the seedling survival probability. Shelton (1995) noted that overstory density did not appear to be limiting to the establishment of natural shortleaf pine seedlings and suggested that newly-established pine seedlings may be moderately shade-tolerant and become less tolerant with age. Moreover, the canopy growing space requirement of a 1-0 pine seedling is less than a square foot and initially can be accommodated by a small canopy gap. This adaptability may explain why shortleaf pine seedlings can persist under a partial or full overstory for a short time (Shelton and Cain 2000). Blizzard and others (2007a) reported that overstory density had little effect on the establishment of natural shortleaf pine seedlings elsewhere in the Missouri Ozarks. As seedlings grow, we anticipate a significant effect from overstory density on growth and competitive status (see below), which will eventually manifest itself in reduced long-term pine seedling survival with increasing overstory density, other factors being equal.

Maintaining a residual overstory had an immediate negative effect on basal diameter growth and on shoot growth. For an average-size seedling, both diameter growth and shoot growth were reduced by 8 percent. Our

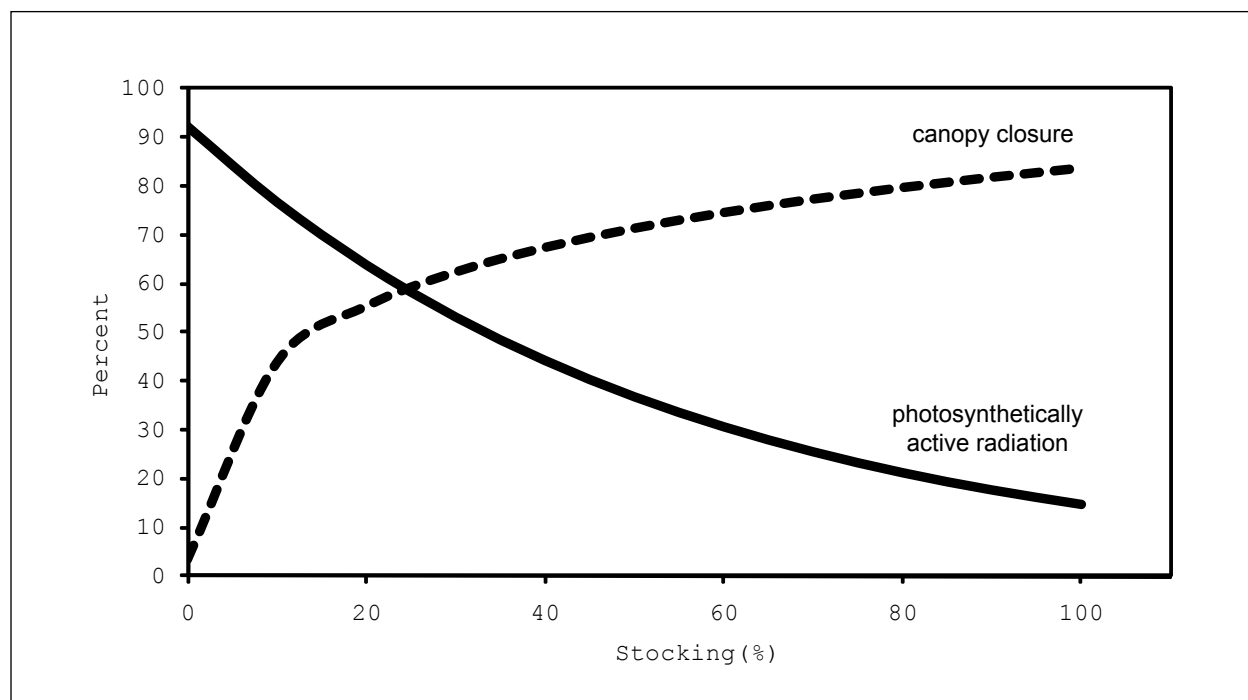


Figure 4.—Relationship between overstory stocking, canopy closure (dashed line), and photosynthetically active radiation (solid line) expressed as percent of full sunlight, and canopy closure for the study area. Adapted from Blizzard and others (in press).

findings were consistent with those of Liming (1946), who reported that the height growth rate of shortleaf pine increased significantly when the overstory density was reduced to or below 27 square feet per acre, which is about 20-percent stocking in our study region. Shelton (1995) also reported growth reductions in 1-year-old shortleaf pine seedlings with increasing overstory density in the Ouachita Mountains of western Arkansas and southeastern Oklahoma although direct comparisons cannot be made to our data because of differences in the respective studies. These shortleaf pine seedling growth reductions are likely to persist and eventually favor the hardwoods as Blizzard and others (2007a) noted that increasing the residual overstory density decreased the probability of producing dominant or codominant pine reproduction 5 years after harvesting. These growth reductions are attributed to the reduction in light levels in plots retaining a residual overstory (Tappe and others 1993). Blizzard and others (in press) demonstrated that even minor increases in residual stocking such as from 0 to 30 percent decreases the photosynthetically-active radiation from 92 to 50 percent of full sunlight in our study area (Fig. 4).

Collectively, our findings and those of others examining the growth of shortleaf pine seedlings are compatible with many of the methods recommended for regenerating shortleaf pine in the Missouri Ozarks. Because of the noted growth reductions under residual overstories, shortleaf pine guidelines recommended managing pines in even-aged stands using clearcutting with hardwood competition control as the preferred regeneration method (Brinkman and Rogers 1967, Brinkman and Smith 1968). These recommendations were intended to minimize the rotation length by maximizing the survival and growth rate of the desired pine reproduction and thereby maximizing the economic return (Brinkman and Rogers 1967).

However, maximizing the economic return is seldom the primary management goal for much of the publicly and privately owned land in the Ozark Highlands (Butler 2008) and there are many situations in which foresters elect to retain some residual overstory when managing or regenerating shortleaf pine or pine-oak mixes. For example, retaining some pines and oaks in the overstory may help promote a healthy ground flora community comprising legumes, forbs, and grasses that otherwise would be lost to competition by hardwood seedlings and seedling sprouts should the entire overstory be removed (Nelson 2005). Public land management agencies are increasingly retaining partial overstories during harvesting for aesthetic reasons (Blizzard and others 2007b). Uneven-age management of shortleaf-loblolly pine mixtures has been practiced successfully for many decades in the southern United States (Shelton and Cain 2000), suggesting that with attentive management, pines can successfully be regenerated and recruited under partial overstories. Although significant, the small growth reductions in shortleaf pine seedlings caused by retaining a residual overstory may be overcome by other management interventions to control hardwood competition. Forest managers are increasingly interested in the application of prescribed fire and periodic release of understory trees to favor shortleaf pine seedlings when hardwoods appear to be gaining dominance (Blizzard and others 2007a). The application of these management interventions will be examined in future years during this study.

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

DYNAMICS OF PLANTED CHERRYBARK OAK SEEDLINGS AND YELLOW-POPLAR FROM SEED FOLLOWING A REGENERATION CLEARCUT

Wayne K. Clatterbuck¹

Abstract.—A bottomland loblolly pine (*Pinus taeda*) stand originating from an early 1940s planting on a minor stream bottom of the Coastal Plain in west Tennessee was harvested in 1992 and allowed to regenerate to hardwoods. Although the pines had been planted, a few naturally regenerated yellow-poplar (*Liriodendron tulipifera*) and sweetgum (*Liquidambar styraciflua*) were present in the overstory at harvest. Cherrybark oak (*Quercus pagoda*) seedlings were planted the spring after the harvest to supplement a non-existent oak component in the regenerating stand. Advance reproduction of any species was not present in the closed canopy stand (stem exclusion stage) prior to harvest. This study follows the growth and development of the planted cherrybark oaks with the natural reproduction present on a productive site for 17 years following the harvest. Most of the planted oaks are in intermediate or suppressed crown positions compared to yellow-poplar after 17 years. Although competing yellow-poplars adjacent to planted oaks were cut within a 3-foot radius after 7 growing seasons, yellow-poplars continue to outgrow the oaks, affecting oak growth from increasing distances. This study suggests that on better sites, yellow-poplar from seed grows faster than planted cherrybark oak such that supplemental plantings of cherrybark oak are not successful. Either intensive site preparation is required to control the establishment and growth of yellow-poplar from seed or intermediate operations are necessary on a 3- to 5-year interval to ensure oaks are not overshadowed by faster growing yellow-poplar. Competition control is essential if oaks are to become an overstory component of the new stand.

INTRODUCTION

Oak (*Quercus* sp.) enrichment plantings have been advocated as a substitute for gaining oak advance reproduction when advance reproduction is not present (Wishard and others 1999, Johnson and others 2002, Spetich and others 2004, Morrissey and others 2007). Most oak regeneration recommendations suggest that adequate advance oak reproduction must be present in sufficient size and number before a regeneration harvest takes place (Sander 1972, Loftis 1990, Johnson and others 2002). If not present, oaks will not be a significant component of the new stand. Developing natural oak advance reproduction of adequate size and number prior to the harvest may take 6 to 12 years depending on seed availability, germination, weather, animal predation, and seedling growth and development. One alternative to securing advance reproduction of oak is substituting enrichment plantings after the harvest.

Oak seedlings grow notoriously slowly after outplanting. Faster-growing species often outgrow oak seedlings on the more productive sites. Research has investigated how to increase the growth of planted oaks through

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nursery protocols by planting high-quality, large seedlings capable of growing in height after outplanting (Clark and others 2000, Kormanik and others 2003) or through oak tree improvement programs (Jacobs 2003). However, equally important and receiving less research attention is controlling competing and faster-growing vegetation. Even with the best oak seedling growth, some species, especially yellow-poplar (*Liriodendron tulipifera*) in Tennessee, will outgrow oak on the better sites without treatments to deter growth of yellow-poplar and enhance oak growth.

Given the dominance of other species, how did current oak forests develop on the better sites when faster-growing species were present? The contemporary thought is that frequent small-scale disturbances, such as repeated use of fire and frequent, light partial harvesting through subsistence activities by landowners for fuel and building supplies, promoted conditions that deterred the growth of yellow-poplar and enhanced the growth of oak. Those anthropogenic management conditions do not exist today. Foresters are continually challenged to create conditions to promote oak regeneration, growth, and development through cost-effective management activities.

The purpose of this study is to document the growth of cherrybark oak (*Quercus pagoda*) planted post-harvest in the presence of a faster-growing species such as yellow-poplar on a high-productivity site. This research will demonstrate a need for rigorous control of competing vegetation to ensure successful oak enrichment planting.

STUDY AREA

The study was conducted in Henderson County, TN, about 4 miles east of Wildersville, TN, on the Natchez Trace State Forest (NTSF) managed by the Tennessee Department of Agriculture, Forestry Division (TDA-FD). NTSF occupies a portion of the Red Hills Belt of the East Gulf Coastal Plain section of the Coastal Plain physiographic province (Fenneman 1938). The study site is within Landtype 23, Narrow Moist Bottoms, as defined by the site classification of Smalley (1991). Soils are Aquic Udifluvents (Collins and Iuka series) formed from alluvium washed from upland soils and formed in loess and sandy, loamy, and clayey Coastal Plain sediments. Soils in these minor bottoms have a variety of textures because of mixing during transport, active erosion, and differential rates of deposition. The site is moderately well-drained, with short-duration flooding (flowing water) and a water table from 2- to 5-feet deep.

Annual precipitation is 51 inches, usually evenly distributed across all seasons. Average site index (base age 50) is estimated to be 100 feet for loblolly pine and cherrybark oak and 105 feet for yellow-poplar (Smalley 1991).

The study area was an agricultural field prior to government acquisition by the Resettlement Administration in the late 1930s and was planted with loblolly pine (*Pinus taeda*) during the early 1940s by TDA-FD to control erosion. The resulting stand was characterized as a bottomland pine stand with planted loblolly pine and volunteer yellow-poplar and sweetgum (*Liquidambar styraciflua*) (Table 1), with basal area bordering overstocked conditions of 140 ft²/acre. A regeneration harvest was conducted during winter/spring 1992. The stand was in the stem exclusion stage. A formal regeneration survey was not conducted because leaves were not present during the dormant season when the harvest and planting occurred. However, a survey was conducted on twelve 0.004-acre plots to document remaining stems greater than 3 feet in height. Little advance reproduction more than 3 feet tall was present for commercial species (Table 2).

Table 1.—Stand composition, average number of trees per acre, and mean d.b.h. of trees greater than 5 inches present in 1991 before the harvest for the cherrybark oak enrichment planting study at Natchez Trace State Forest, TN. Data are from timber sale inventory conducted by TDA-FD.

Species	Average number of trees/acre	Mean d.b.h. in inches(range)
Loblolly pine	81	16.8 (12 - 24)
Yellow-poplar	7	14.1 (8 - 18)
Red maple	7	6.4 (5 - 9)
Sweetgum	4	7.7 (6 - 12)
Red oak	3	9.3 (8 - 14)

Table 2.—Advance reproduction greater than 3 feet tall present after harvest (March 1992) for the cherrybark oak enrichment planting study at Natchez Trace State Forest, TN. Inventory based on twelve 0.004-acre plots located on 3 acres with each plot 60 feet apart on a transect.

Species	Number of stems per acre > 3 feet tall
Red maple	42
Sweetgum	21
Blackgum	21
Loblolly pine	0
Red oak	0
Yellow-poplar	0

Cherrybark oak 1-0 bare-root seedlings were planted in April 1992 at varied spacings of 8 to 12 feet to avoid logging slash, stumps, skid trails, and other obstacles from the harvest. The intent was to establish oaks within the regeneration pool and follow their growth in relation to other species. Seedlings greater than $\frac{3}{8}$ inch at the root collar and with tops greater than 24 inches with at least five first-order lateral roots were planted. No site preparation was required prior to planting because the harvest was fairly clean. The management prescription was to allow natural hardwood reproduction to occur, effectively changing the species composition from pine to hardwoods. Oaks were not present in the advance reproduction. Thus, enrichment planting with cherrybark oak was conducted to (1) provide that oak would be a component in the future stand; and (2) conduct research comparing growth of sheltered and unsheltered seedlings. A 4-foot, brown Tubex® tree shelter (Tubex, UK) was placed on every second seedling, creating 70 pairs of seedlings (with and without shelters).

The sheltered seedling study was a companion study to one reported by Clatterbuck (1999). After 7 growing seasons when there was no difference in height growth between sheltered and unsheltered cherrybark seedlings, the shelters were removed and the shelter study was discontinued. A release cleaning through a crown touching technique was conducted in 1999 if the crown of a planted tree was influenced by an adjacent tree (Miller and others 2007). Competing trees were manually cut, usually within a 3-foot radius of the planted cherrybark oak, and no herbicide was used to control sprouting from the cut trees. The remaining seedlings (both sheltered and unsheltered) are the basis of the present study.

METHODS

After 17 growing seasons, in winter 2008, 117 of the planted cherrybark oak trees were located. Diameters at 4.5 feet (d.b.h.) and crown class of planted cherrybark oak trees were recorded. Distance to the nearest competing tree that was greater than 2 inches d.b.h. were measured and crown class and species were recorded. Bivariate analyses of cherrybark oak diameter and distance to competing trees were explored using JMP 8.0 (SAS Institute Inc., Cary, NC, 2008). A stand inventory was conducted to characterize the stand by establishing a transect and locating ten 0.05-acre plots at 100-foot intervals.

RESULTS

After the 2008 growing season (17 years), the fully stocked stand is dominated by yellow-poplar with a few sweetgum in the upper canopy (Table 3). Planted cherrybark oaks are subordinate to the yellow-poplar. Red maple (*Acer rubrum*), blackgum (*Nyssa sylvatica*), and dogwood (*Cornus florida*) are also present in the subcanopy. Yellow-poplar averages 7.0 inches in diameter with a range of 4.9 to 10.3 inches. The planted cherrybark oak average 1.7 inches with the majority of the stems in the one-inch diameter class (Table 4).

A positive relationship ($r^2 = 0.80$; $p < 0.0001$) exists between the diameter of the planted cherrybark oak (transformed log) and distance to the nearest yellow-poplar greater than 2 inches (Fig. 1).

Table 3.—Stand composition, average number of trees per acre, and mean d.b.h. of trees greater than 2 inches present after the 2008 growing season for the cherrybark oak enrichment planting study at Natchez Trace State Forest, TN. Inventory sample based on ten 0.02-acre plots on 3 acres with each plot 100 feet apart on a transect. Planted cherrybark oak trees were not included in this inventory.

Species	Average number of trees/acre	Mean d.b.h. in inches(range)
Yellow-poplar	255	7.0 (2 - 10)
Sweetgum	75	4.3 (2 - 8)
Red maple	55	5.6 (2 - 12)
Blackgum	40	4.4 (2 - 6)
Dogwood	15	3.1 (2 - 4)
Loblolly pine	5	4.8 (*)
Total	445	

*only one tree in sample

Table 4.—Number of planted cherrybark oak (n = 117) by 1-inch diameter class after 17 growing seasons for the cherrybark oak enrichment planting study at Natchez Trace State Forest, TN.

Diameter size class (inches)	Number of planted cherrybark oak
1-inch (0.5 to 1.4)	63
2-inch (1.5 to 2.4)	35
3-inch (2.5 to 3.4)	14
4-inch (3.5 to 4.4)	4
5-inch & greater (> 4.5)	1

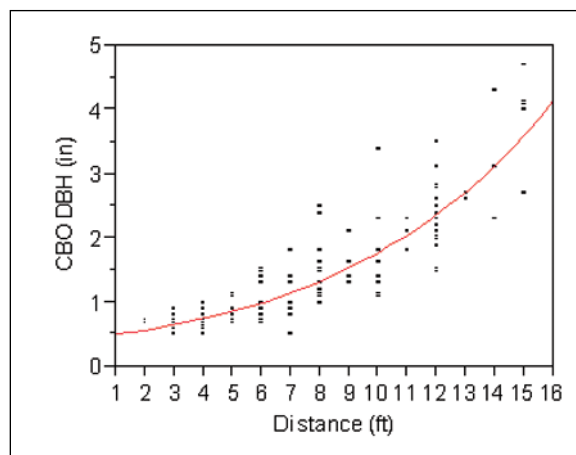


Figure 1.—Logarithmic relationship ($r^2 = 0.80$) of planted cherrybark oak diameter (CBO DBH) to distance of nearest yellow-poplar stem greater than 2 inches in diameter (n = 117) after 17 growing seasons for the cherrybark oak enrichment planting study at Natchez Trace State Forest, TN. Fitted equation: $\text{Log [CBO DBH (in)]} = -0.856 + 0.143 [\text{Distance (ft)}]$.

DISCUSSION

Although yellow-poplar seedlings were not present in the advance reproduction at time of harvest, viable seed was present in the soil duff. Seed available from adjacent trees also germinated and regenerated the area after the harvest. Yellow-poplar seed is known to stay viable in the forest duff for 4 to 7 years (Clark and Boyce 1964, Beck 1990) and to grow rapidly following disturbance from fire or soil scarification during harvesting operations (Beck and Della-Bianca 1981). Yellow-poplar presently composes nearly 90 percent of the overstory after 17 growing seasons and averages 7 inches in diameter. The present stand is fully stocked, bordering on overstocked conditions.

On these better sites, forest managers at TDF-FD desire a more mixed composition with oaks as part of the composition. Oak does not need to dominate the overstory but should compose a portion of the overstory. Since oaks were not present in the advance reproduction, oaks were planted to supplement the composition of the natural regeneration. The planted cherrybark oaks are suitable for these sites (Smalley 1991) but were influenced by the faster-growing and more numerous yellow-poplar. Planted oaks were relegated to subordinates of yellow-poplar and averaged 1.7 inches d.b.h. in 17 growing seasons. A distance relationship between cherrybark oak and competing yellow-poplar was apparent in this study, with larger diameter oaks occurring at increasing distances from yellow-poplars. However, only 19 of 117 oaks (16 percent) surviving the 17 growing seasons were 3 inches in d.b.h. or greater in contrast with the larger yellow-poplar. Even though cherrybark oak seedlings were graded such that the best seedlings with adequate root systems were planted, the planted oaks could not keep up with the faster-growing yellow-poplar.

Although competition adjacent (3-foot radius) to the planted cherrybark oaks was cut after 7 growing seasons, the faster growth of yellow-poplar outside the 3-foot radius still affected the growth of the cherrybark oak. As the yellow-poplar grew, they progressively influenced the slower growing oaks from greater distances. Although spacing was wider, similar relationships were found in a sycamore and cherrybark oak plantation (Clatterbuck and others 1987).

This study did not have a control to monitor the growth of planted cherrybark oak without any competition from adjacent stems. However, a cherrybark oak plantation on a similar site at NTSF where competition was controlled averaged 5 inches in d.b.h. after 15 growing seasons (Clatterbuck 2002).

If advance oak reproduction is absent and oaks are desired in the new stand, yellow-poplar regeneration from seed must be controlled. Even with the use of high-quality oak seedlings that are capable of growth once outplanted, natural yellow-poplar reproduction will usually overwhelm the oaks, relegating oaks to the subordinate canopy. Several options could be available to control the yellow-poplar and promote the growth of oaks, even though most of these options remain untested. These alternatives include:

1. Conduct post-harvest burning to consume the viable yellow-poplar seed and provide less competition to the oaks once outplanted. Adjacent remaining yellow-poplar seed trees should be cut so the seed from the current year would not disseminate into the harvested opening.
2. Use herbicides on an annual or biennial basis to control yellow-poplar seedlings
3. Use a pre-emergent herbicide to control the viable yellow-poplar seed in the forest duff during site preparation prior to planting.
4. Perform a crop tree release at intervals of 3 to 5 years to allow the planted oaks to grow and develop and control yellow-poplar at progressively greater distances.

Site productivity affects the development of cherrybark oak and yellow-poplar. On the highly productive site in this study, yellow-poplar grows faster than cherrybark oak. With decreases in site productivity, however, yellow-poplar growth may not be as great (O'Hara 1986). Cherrybark oak has a greater probability of growing, surviving, and forming stand mixtures with yellow-poplar on lower-productivity sites.

SUMMARY

Yellow-poplar from seed has the ability to outgrow and overwhelm planted cherrybark oak seedlings on highly productive sites that support the growth of both species. Most research has explored methods to increase the growth of oak through tree improvement programs and nursery protocols to produce seedlings capable of greater growth during the first growing season. However, control of faster-growing species/competition is also necessary through site preparation or intermediate operations to allow oaks to grow and develop with little hindrance. This study demonstrates that even with larger planted seedlings where advance reproduction of any species is not present, some type of control of yellow-poplar is essential for oak to become a viable component of the future stand

ACKNOWLEDGMENTS

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THE INFLUENCE OF CULTURAL TREATMENTS ON THE LONG-TERM SURVIVAL AND GROWTH OF PLANTED *QUERCUS RUBRA*

James J. Zaczek and Kim C. Steiner¹

Abstract.—A northern red oak (*Quercus rubra* L.) plantation testing 20 nursery stock and planting methods was used to evaluate treatments 3, 6, 10, and 17 years after planting. Survival over all treatments was 92 percent at age 3 and declined to 74 percent, 56 percent, and 39 percent at ages 6, 10, and 17, respectively. At age 17, survival was highest for containerized stock (73 percent) and was independent of nursery undercutting (root-pruning), top-clipping at planting, producing nursery transplant stock, and tree shelters. Surviving trees tended to be initially taller (mean of 44 cm vs. 33 cm for unclipped trees) and had larger mean stem caliper (8.3 mm vs. 6.7 mm) at the time of planting compared to those that died. Trees shorter than mean height (1.3 m) tended to succumb to effects of browsing in early years after fence removal, and later to intraspecific competition after canopy closure. Height at age 10 and diameter at age 17 were greatest for containerized stock that was not top-clipped at planting time, but many of the treatments were not statistically distinguished from each other. Top-clipping did not promote multiple stems.

INTRODUCTION

Typically larger and better-developed seedlings are considered to be higher in quality and are recommended for planting (Johnson and others 1986, Dey and Parker 1997, Ward and others 2000) but do not always exhibit greater growth or survival after planting (Myers and others 1989). Furthermore, older and larger stock can be more costly or difficult to lift, package, ship, and plant (Bullard and others 1992, Countryman and others 1996).

Varying nursery cultural practices are promoted for growing high-quality northern red oak planting stock, including undercutting, top-clipping, containerizing stock, and transplanting in the nursery, but the effects of these treatments differ. Undercutting (sometimes referred to as “root-pruning”) in nursery beds can benefit outplanting survival and stimulate lateral root development (Schultz and Thompson 1991, 1996) but can lead to developing smaller seedlings (Zaczek and others 1993). Top-clipping can increase survival (Limstrom 1963, Spetich and others 2002) but also can reduce seedling height and can retard the growth of new roots after planting (Farmer 1975, Larson 1975). Tree shelters have been shown to increase oak survival or height in some comparisons (Stange and Shea 1998, Ward and others 2000, Ponder 2003) but were detrimental or neutral in others (Clatterbuck 1999, Ward and others 2000, Ponder 2003).

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This study was uniquely designed not only to test a wide variety of combined nursery and planting treatments simultaneously but also to use the same seed source to minimize genetic variability among treatments. When a common seed source is used, survival and growth differences expressed among treatments are more reliably attributed to the nursery cultural and planting treatments rather than to potential genetic differences known to exist among varying seed sources. Ultimately, our goal was to demonstrate results to help nursery growers produce higher quality northern red oak stock and to aid land managers in their stock choice and planting regime decisions to improve establishment in the short term, and survival and growth in stands over the long term.

METHODS

A northern red oak plantation was established in April 1988 within a clearcut mixed-oak stand in Huntingdon County, PA, to evaluate performance relative to type of planting stock and cultural factors. The planting site was located in a former mixed hardwood stand that supported several oak species, including northern red oak. It had been clearcut in late 1987 and the slash was removed.

Twenty treatments, each with at least 33 replications per treatment, were originally planted (Table 1) in a completely random design at 1.2-m x 1.2-m spacing. Nursery treatments consisted of various planting stock types: 2-year-old containerized (2-CON, 7.9-liter pots); 2-1 and 1-1 bareroot nursery transplants; and 2-0 and 1-0 bareroot. Some treatments were combined factorially with top-clipping (TC to 15 cm tall vs. UC or unclipped) at planting time and undercutting in the nursery bed. Undercutting or root-pruning (RP) vs. no undercutting or unpruned (UP) was performed on seedlings in the nursery bed to a depth of 20 cm in midsummer of each year. Treatments were sowed in the nursery over 3 years to produce stock that was planted at the field site during the same year, 1988. The same seed source, a bulk seedlot collected in similar parental proportions over years from four seed parents, was used to produce the seedlings of all treatments except the

Table 1.—Summary, abbreviation, description of nursery stock treatments, and the number per treatment compared for outplanting performance.

Treatment Abbrev.	Description	Number per treatment
2-CON/TC	2-yr containerized, top-clipped	35
2-CON/UC	2-yr containerized, unclipped top	35
2-1/TC	2-1 transplants, top-clipped	35
2-1/UC	2-1 transplants, unclipped top	35
1-1/TC	1-1 transplants, top-clipped	35
1-1/UC	1-1 transplants, unclipped top	34
2-0/RP/TC	2-0 seedlings, root-pruned (undercut), top-clipped	33
2-0/RP/UC	2-0 seedlings, root-pruned (undercut), unclipped top	34
2-0/UP/TC	2-0 seedlings, unpruned roots, top-clipped	33
2-0/UP/UC	2-0 seedlings, unpruned roots, unclipped top	33
1-0/RP/TC	1-0 seedlings, root-pruned (undercut), top-clipped	33
1-0/RP/UC	1-0 seedlings, root-pruned (undercut), unclipped top	33
1-0/UP/TC	1-0 seedlings, unpruned roots, top-clipped	33
1-0/UP/UC	1-0 seedlings, unpruned, unclipped top	34
1-0/ALA/TC	1-0 seedlings, Alabama raised, top-clipped	34
1-0/ALA/UC	1-0 seedlings, Alabama raised, unclipped top	33
D-S/TBX	Direct-seeded acorns, Tubex® installed	33
D-S	Direct-seeded acorns, no Tubex®	34
1-0/NURS/TBX	1-0 nursery-run seedlings, Tubex® installed	29
1-0/NURS	1-0 nursery-run seedlings, no Tubex®	35

direct-seeded and the 1-0 nursery-run (1-0 NURS) treatments. All seedling planting stock was grown at Penn Nursery (Pennsylvania Bureau of Forestry) except for the 1-0/ALA, which was grown at International Paper Company's Selma, AL, nursery under a regime designed to produce larger 1-0 stock. Direct-seeded (D-S) acorns were sown directly at the site at the time of the field planting establishment in 1988.

For the first three growing seasons, the planting was enclosed by a six-strand electric fence, which was generally effective at excluding white-tailed deer (*Odocoileus virginianus* Boddaert). The site was maintained weed-free with directed applications of glyphosate herbicide while shielding seedlings from the spray. After the second growing season, two treatments of tree shelters using Tubex[®] (Tubex Limited, Aberdare, UK) (TBX) were established by installing shelters over seedlings that grew from treatments of D-S acorns (n = 33) and 1-0 nursery-run stock (n = 29) (see Zaczek and others 1993). Survival of tree-sheltered seedlings reflects those trees remaining alive after year 2.

After the third growing season, the fence was removed and herbicide discontinued. Results from this and two other replicated sites were previously reported after the third growing season (Zaczek and others 1993) and from this site only after 6 years (Zaczek and others 1997). Further site and treatment details may be found in these previous papers.

Height (dm) and diameter at breast height (d.b.h. in cm) were recorded for each living tree in late August 1997, essentially 10 growing seasons after planting. We also recorded d.b.h. for each living tree in late July 2004, 17 growing seasons after planting.

Our analysis was designed to answer several other questions that might be asked by a field forester over the long term: Does undercutting improve survival and growth for all types of nursery stock? Does top-clipping increase survival or growth?

Survival comparisons among contrasting treatments of interest at age 10 and 17 were made using chi-square analysis at alpha = 0.05 (Preacher 2001). Testing for treatment differences in height and diameter among treatments and treatment groups was performed with analysis of variance at alpha = 0.05 using SPSS (SPSS Inc., Chicago, IL). Pearson correlation coefficients were generated for initial seedling heights and subsequent measurements of heights and diameter for those trees alive at age 17.

RESULTS

TENTH-YEAR SURVIVAL

Ten years after outplanting, survival was 56.0 percent over all treatments and ranged widely from 17.2 percent to 94.3 percent among treatments (Fig. 1). Chi-square tests among treatment groups indicated that survival was not significantly different for TC vs. UC, 1-0 ALA vs. 1-0 (Pennsylvania grown), D-S vs. 1-0, and TBX vs. unsheltered. Survival was greater for undercut (RP) vs. UP seedlings, 2-yr containerized vs. 2-0 bareroot seedlings, 2-0 vs. 1-0 bareroot stock, and 1-0 vs. 1-0 nursery run seedlings (Table 2). Further analysis within the undercut and not undercut treatment groups determined that survival of 2-0/RP (86.6 percent, n = 67) differed from 2-0/UP (68.2 percent, n = 66) seedlings (p = 0.011). However, survival was similar (p = 0.438) for undercut 1-0/RP (47.0 percent, n = 66) and 1-0/UP (40.3 percent, n = 67) treatments. Transplant stock (2-1 and 1-1) had lower survival than less intensively cultured 2-0 stock. Overall, trees that died between ages 6 and 10 were (p<0.001) shorter at age 6 (mean 10.9 dm) than the height at age 6 (mean 28.9 dm) of the trees alive at age 10.

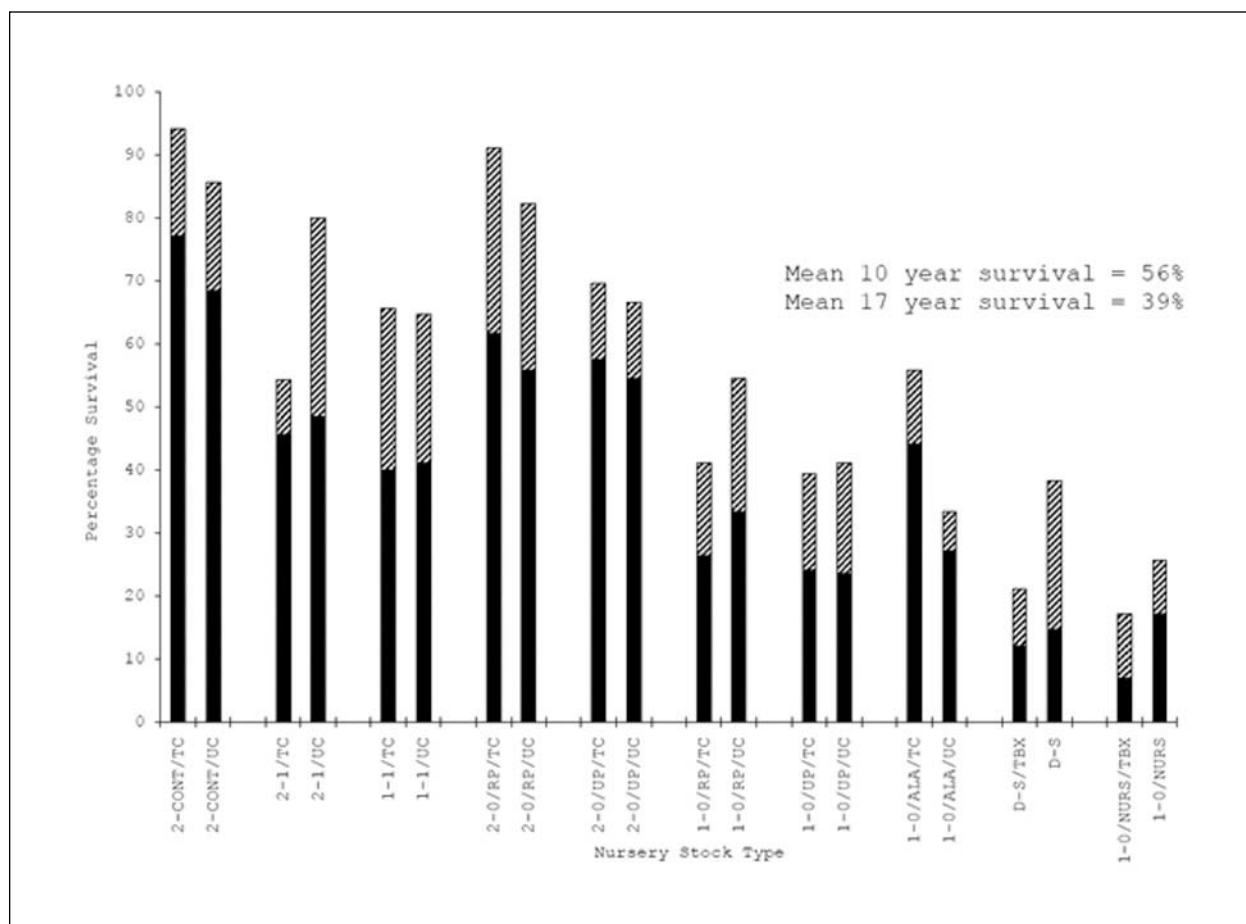


Figure 1.—Percentage survival of northern red oak nursery stock 10 years (top of hatched bar) and 17 years (top of solid bar) after planting by treatment.

SURVIVAL AFTER 17 YEARS

Seventeen years after planting, plantation mean survival was further reduced to 39.4 percent ($n = 265$) and ranged from 6.9 percent to 77.1 percent among all treatments (Fig. 1). At this time, survival differed among treatment group comparisons: 2-CON vs. 2-0 and 2-0 vs. 1-0 (Table 2). Survival differed between the 2-CON (72.9 percent, $n = 70$) and 2-0/UP (56.1 percent, $n = 66$) treatment comparisons ($p = 0.041$) but not the 2-CON and 2-0/RP (59.7 percent, $n = 67$) comparisons ($p = 0.103$). Overall, trees that died between the ages of 10 and 17 were significantly ($p < 0.001$) shorter at age 10 (mean of 37.6 dm) than those that were alive at age 17 (mean height at age 10 of 53.8 dm).

TENTH-YEAR HEIGHT

Mean height at age 10 for surviving trees was 53.8 dm. Although there were some significant differences ($p = 0.001$) in tree height among treatments (Fig. 2), many treatments were not statistically distinguishable from each other. Trees grown from 2-year-old containerized stock had the greatest overall mean height at 59.4 dm, but could not be statistically distinguished from 15 other treatments (Fig. 2). The direct-seeded tree shelter treatment produced the shortest trees (42.0 dm), but this result was not statistically different from 10 other treatments. When we consider only those 16 treatments grown from the same seed source, height at age 10 differed among this subset of treatments ($p = 0.027$).

Table 2.—Chi-square analyses of survival at age 10 and 17 amongst specific treatments or treatment groups. For treatment descriptions refer to Table 1. All treatments and treatment groups utilized a common known seedlot except for 1-0 nursery-run, direct-seeded, and the tree shelter (which enclosed seedlings grown from directed seeded acorns) treatment.

Treatment or Treatment Group Comparison	Survival % Age 10	n	Chi-square p value	Survival % Age 17	n	Chi-square p value
Top-clipped (TC) Uncropped Top (UC)	63.8 63.8	271 271	1.000	47.6 44.3	271 271	0.438
Roots Pruned (RP) Unpruned Roots (UP)	66.9 54.1	133 133	0.033	45.1 39.9	133 133	0.385
2-yr Container (CON) 2-0 Bareroot	90.0 77.4	70 133	0.028	72.9 57.9	70 133	0.036
2-0 Bareroot 1-0 Bareroot	77.4 44.0	133 200	<0.001	57.9 30.0	133 200	<0.001
1-0 Pennsylvania-grown 1-0 Alabama-grown (ALA)	43.6 44.8	133 67	0.874	27.1 35.8	133 67	0.202
1-0 Bareroot 1-0 Nursery-run (NURS)	44.0 25.7	200 35	0.004	30.0 17.1	200 35	0.118
Direct-Seeded (DS) 1-0 Nursery-run (NURS)	38.2 25.7	34 35	0.265	14.7 17.1	34 35	0.783
Tree shelter (TBX) No Tree shelter	19.4 31.9	62 69	0.102	9.7 15.9	62 69	0.145

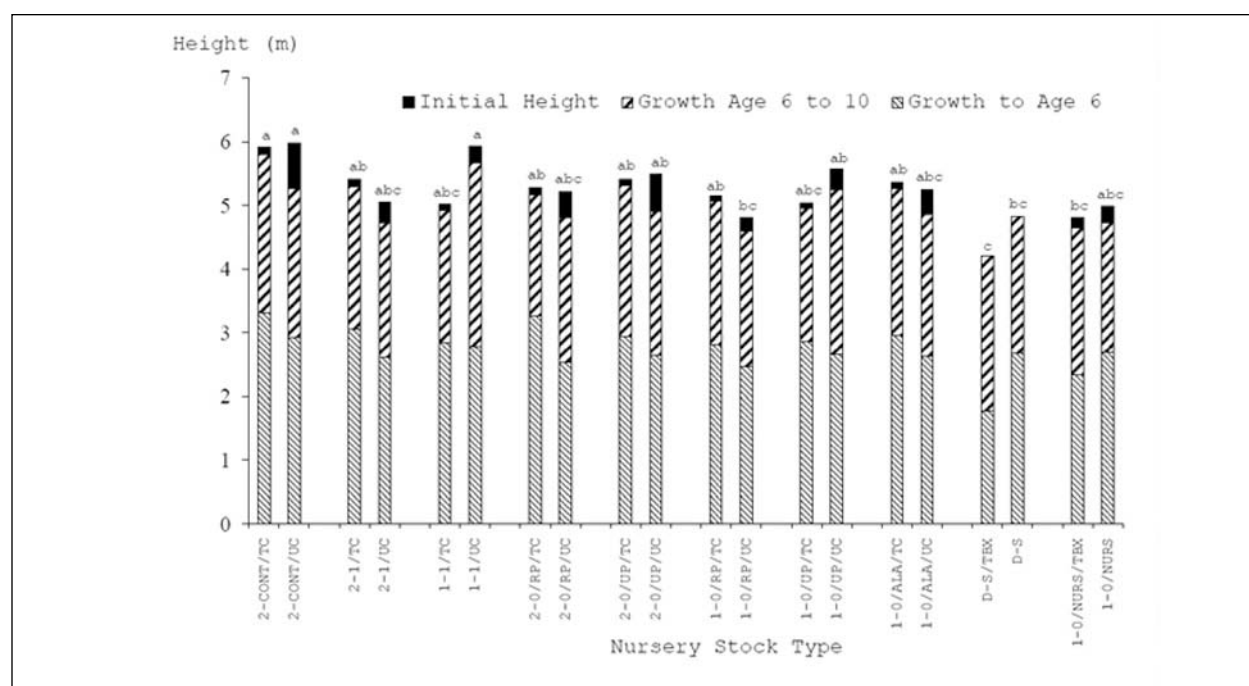


Figure 2.—Mean initial height (dm) of surviving northern red oak trees at planting time, growth (dm) until age 6, growth from age 6 to 10, and total height at age 10 (top of the bars) by treatment. Treatments sharing the same letter are not significantly different at $p = 0.05$ (Duncan's Multiple Range Test).

DIAMETER AFTER 17 YEARS

Mean DBH among all live trees at age 17 was 7.51 cm and ranged from a high of 9.97 cm for the 2-CON/UC treatment to 3.18 cm for the D-S/TBX treatment (Fig. 3). As was the case in the height analysis, there were significant differences among treatments ($p < 0.001$). However, the treatment with the greatest mean d.b.h. (2-0 CON/UC) was not statistically different from eight other treatments while the treatment with the smallest mean diameter (DS/TBX) was significantly smaller than 17 of the treatments (Fig. 3).

INITIAL SEEDLING SIZE

For trees living through age 17, significant ($p \leq 0.002$) Pearson correlations were found when testing all height and diameter measurements (Table 3). In general, correlations became weaker as the time span between measurements increased.

STEM FORM

At age 17, 14 trees out of the 265 living individuals had two stems and no trees had greater than two stems. Of the multi-stemmed trees, only five trees had both stems living, three trees of which developed from top-clipped stock and two from unclipped stock.

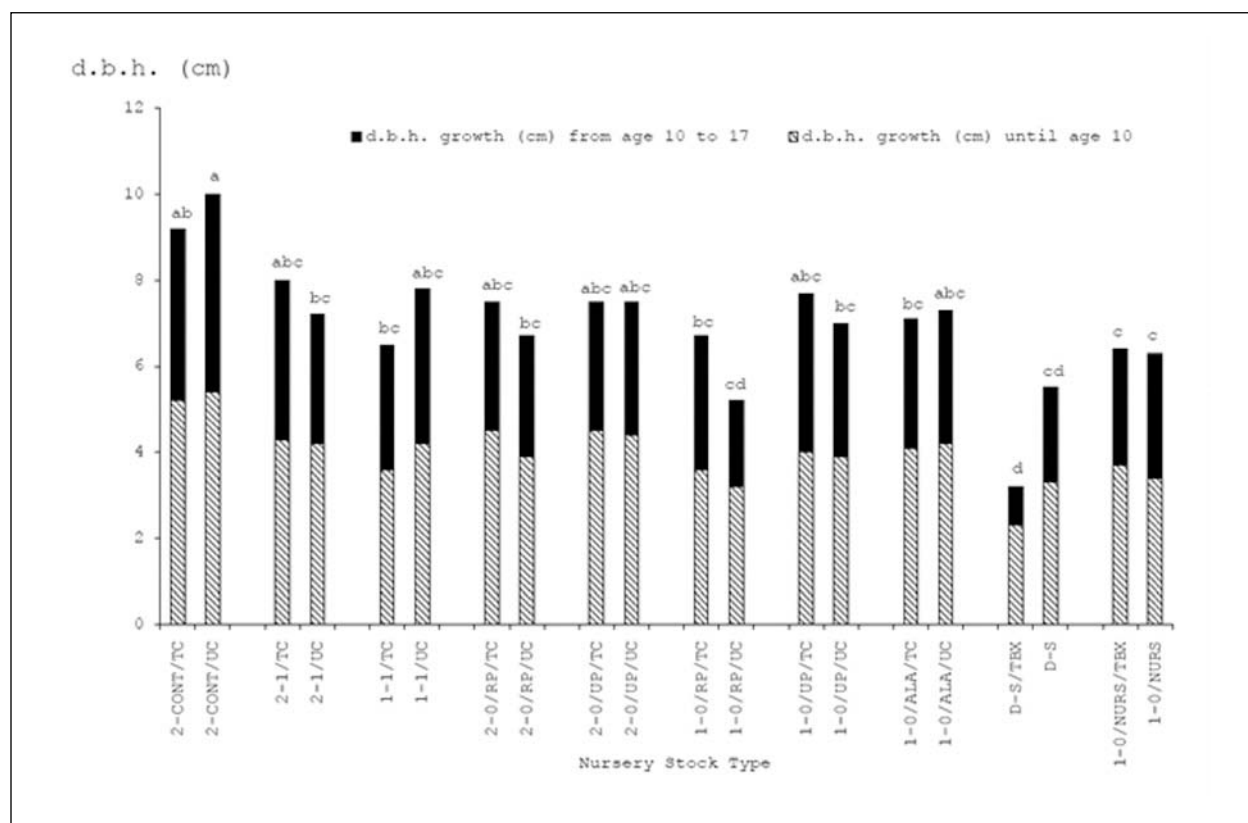


Figure 3.—Mean d.b.h. growth (cm) to age 10 (top of the solid bar) and d.b.h. growth from age 10 to 17 (hollow bar) of surviving northern red oak trees by treatment. Treatments sharing the same letter are not significantly different at $p = 0.05$ (Duncan's Multiple Range Test).

Table 3.—Pearson correlation coefficient matrix of initial stem caliper (Cal 0) and height at various ages (Ht X, where X = 0, 1, 2, 3, 5, 6, 10 years) and d.b.h at age 17 of trees that survived through age 17. All relationships are significant at $p \leq 0.002$.

	Cal 0	Ht 0	Ht 1	Ht 2	Ht 3	Ht 5	Ht 6	Ht 10	d.b.h. 17
Cal 0	1.000	0.361	0.582	0.537	0.495	0.432	0.425	0.280	0.433
Ht 0		1.000	0.529	0.299	0.309	0.264	0.239	0.179	0.257
Ht 1			1.000	0.713	0.619	0.514	0.512	0.352	0.466
Ht 2				1.000	0.849	0.670	0.625	0.475	0.552
Ht 3					1.000	0.793	0.723	0.562	0.598
Ht 5						1.000	0.819	0.701	0.604
Ht 6							1.000	0.735	0.679
Ht 10								1.000	0.649
d.b.h. 17									1.000

DISCUSSION

Although a minority of the trees that were originally planted managed to survive to age 17, this high mortality rate was attributed to competition among the planted trees (the plantation was never mechanically thinned) rather than to “failure” of the plantation establishment. Although there were some differences in tenth-year height and seventeenth-year diameter among treatments, differences in survival among the treatments were paramount. Survival tended to be greater for treatments with larger seedlings, either taller in height (excluding top-clipped seedlings) or larger in diameter at planting time, or for those 2-0 seedlings that had been undercut in the nursery.

Survival at age 10 or at age 17 was not significantly influenced by top-clipping, rearing in another nursery, or tree shelters. Other reports have found top-clipping to be beneficial (Limstrom 1963, Spetich and others 2002) or to have little or no benefit (Crunkilton and others 1989, Gordon and others 1992), and the latter has consistently been the case in all previous reports of this study (Zaczek and others 1993, Zaczek and others 1997). Our experiences with top-clipping were that, after planting, clipped seedlings broke bud several weeks later and the resulting leaves were much larger in area than unclipped seedlings. Even though top-clipping has been shown to somewhat retard the growth of new roots after planting (Farmer 1975, Larson 1975), clipping may allow seedlings to first establish roots before transpirational demands on the seedling begin. Clipped seedlings recovered in height to become similar in height to unclipped counterparts by the second year after planting and this practice had no apparent negative effect on height or survival (Zaczek and others 1993). A potential concern of top-clipping stock at planting is that multiple-stemmed trees may develop, leading to decreased quality. However, of the living top-clipped trees after 17 years, only 2.3 percent had living multiple stems.

Root pruning in the nursery and using more intensively cultured stock (2-CON vs. 2-0; 2-0 vs. 1-0; and 1-0 vs. 1-0 NURS) resulted in greater survival at age 10 vs. contrasting treatments. Others have found root pruning to be beneficial to survival and to result in the development of more lateral roots (Schultz and Thompson 1991, 1996). In the current study, however, survival to age 17 was not consistently benefited by root pruning; a positive effect was seen for older 2-0 stock but not for 1-0 stock. Undercutting tends to produce slightly more compact seedlings and is likely warranted and beneficial for nurseries that produce moderately larger and older stock (Zaczek and others 1993). These more compact undercut seedlings, although neutral in survival and growth benefit after 17 years, are easier for nurseries to handle and ship.

By age 17, the 2-CON stock had significantly greater survival than the contrasting 2-0/UP but not the 2-0 RP bareroot stock. Height at age 10 and d.b.h. at age 17 were similar for 2-CON and 2-0 treatments. The greater inputs (pots, potting media, shipping) and efforts (transporting to field sites, digging large holes, and increased planting time) associated with large containerized stock reduces the feasibility of using this stock type. Older bareroot transplant stock (2-1) did not perform as well over time as less intensively cultured 2-0 material. This outcome is apparently due to transplant shock within the nursery, leaving 2-1 and 1-1 transplants 16 percent and 34 percent smaller in initial stem diameter, respectively, and 46 percent and 60 percent smaller in initial height, respectively, than 2-0/UP/UC seedlings.

Tree shelters were not beneficial to survival or growth in this study. Others have found that shelters can enhance survival or height growth of oaks (Stange and Shea 1998, Ward and others 2000, Ponder 2003) but in some research, shelters were neutral or detrimental to survival or height (Clatterbuck 1999, Ward and others 2000, Ponder 2003). The poor response of shelters in the current study may have been due to our deliberate approach to refrain from any maintenance to the shelters once they were installed. Shelters toppled over because the stakes that held them broke or became rotten. Rodents chewed their fasteners. Black bears (*Ursus americanus* Pallas) damaged the shelters, causing the seedlings inside to either die or resprout from the root collar. The need for repeated shelter maintenance—replacing stakes and fasteners—has been noted by others (Kittredge and others 1992, Minter and others 1992). Additionally, periodic cicada (*Magicalcada septemdecim* L.) and gypsy moth (*Lymantria dispar* L.) appeared to thrive in the shelters (Zaczek and others 1997). The added costs of shelters and the lack of a survival or growth benefit fail to justify their use in applications similar to those in our study.

Although there were significant differences among treatments for height at age 10 and diameter at age 17, most treatments were similar in mean growth. This result largely reflects the stock types; a greater proportion of smaller trees tend to have greater mortality of their smallest trees over time. This mortality pattern, in turn, removed the small trees from the calculations of mean height and tended to bias upward the heights of those treatments. Consequently, in cases like this, where survival varied greatly among treatments, choosing planting stock based on height or diameter performance in later years does not appear to be the most appropriate approach. Rather, survival differences are the superior indicators to base stock choice decisions. Ultimately, the size of a tree is critically important and drives survival and future growth. All treatments produced at least some seedlings that were larger than the overall average and thus survived to age 17. Treatments that have greater proportions of seedlings that are larger tend to have higher percentage survival. Nearly all trees that were below average in height during the course of the study failed to survive. For individual trees, survival was tied closely to size and competitive position relative to their neighbors.

The decline in survival in later years is in part due to intraspecific competition among relatively closely-spaced trees, which tended to result in higher mortality for those trees that were shorter than the overall average. To help illustrate this result, we calculated mortality between consecutive measurement periods (years 3 to 6, 6 to 10, and 10 to 17) by size class of the preceding period (Fig. 4). The vast majority of mortality occurred in the smaller size classes of the trees in an age cohort. Averaging over the three comparisons, 94.4 percent of the trees that died in the following measurement period were below average in height in the preceding measurement period. Between years 3 and 6, canopy closure was not complete and although the influence of intraspecific competition among the planted oaks was likely beginning to occur, a significant proportion of mortality was attributed to white-tailed deer browsing of seedlings shorter than 130 cm (Zaczek and others

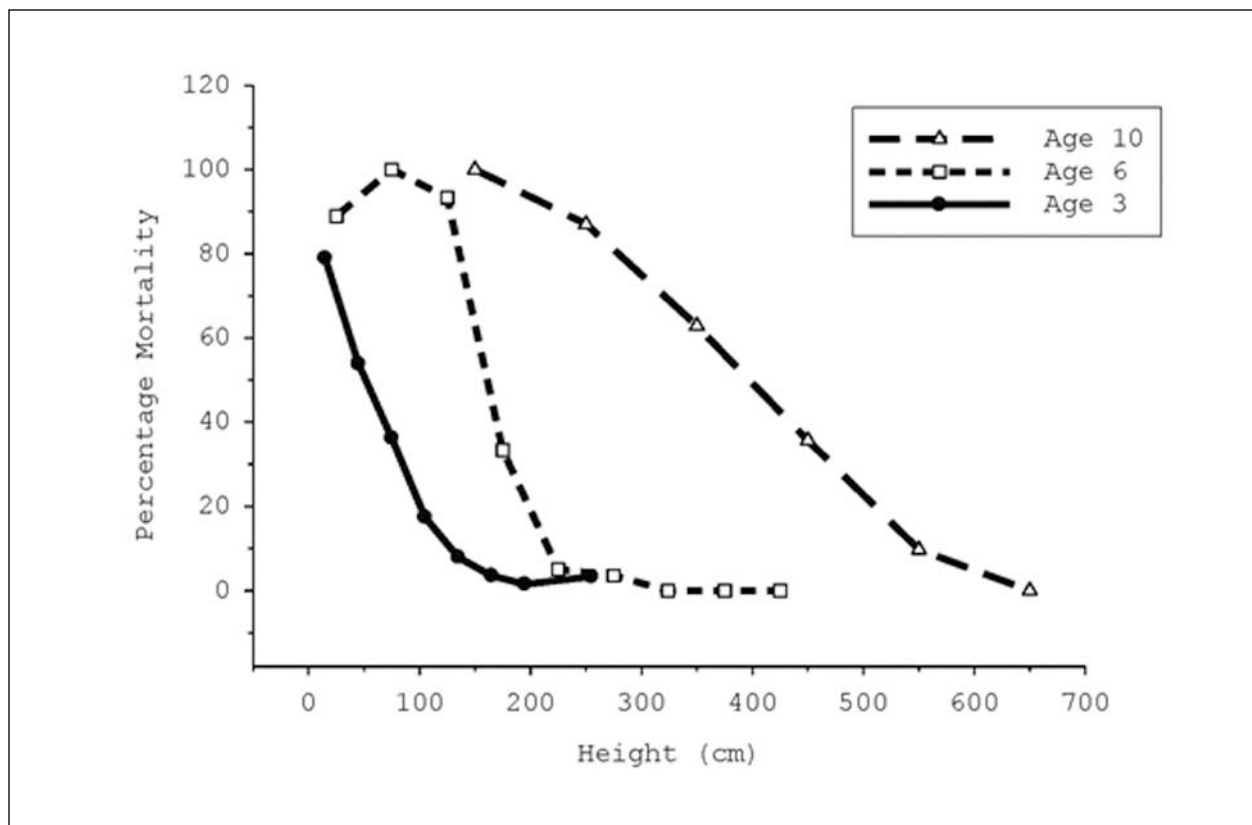


Figure 4.—Percent mortality between measurement periods (years 3 to 6, 6 to 10, 10 to 17) by size class at the beginning of the period. Mean heights: age 3 = 134 cm, age 6 = 245 cm, and age 10 = 538 cm.

1997). Browsing has been shown to negatively affect survival and growth of seedling oak (Buckley and others 1998, Jacobs and others 2004). In later years, with the onset of canopy closure and the development of some understory vegetation, tree-to-tree (intraspecific) competition was most likely the primary influence on survival.

Trees were planted at 1.2 m x 1.2 m or 6,726 stems ha⁻¹ in this study. The site in later years had a nearly complete canopy coverage of northern red oak with very few other woody plant species growing within the oak overstory. This result was influenced by herbicide treatments in the first 3 years, which kept all other vegetation from becoming established. After 17 years, the understory vegetation consisted of a moderate cover of ferns and some grass.

Oak seedling density of this magnitude or greater is common in developing stands in this part of the Appalachians. In a location near this study, tree seedling densities early in development of a mixed-oak shelterwood (age 1) were 23,756 stems ha⁻¹ or at age 3, 43,318 stems ha⁻¹ with the overwood intact (Zaczek 2002) or later (age 6) at 40,382 stems ha⁻¹ with the overwood removed (Zaczek and Lhotka 2004). Seedling densities ranged from 1,557 to 12,231 stems ha⁻¹ for 10-year-old developing mixed-species stands in the Central Hardwoods (Myers and Buchman 1988) and 5,541 stems ha⁻¹ 14 years after harvest in a bottomland hardwood forest (Bowling and Kellison 1983).

It is uncertain whether the competitive effects that are experienced by northern red oak individuals in a single-species planting such as this are more severe or less so than what might be experienced in developing natural stands or plantings of mixed species. Individuals of a species can be more adversely affected by intraspecific competition than by interspecific competition (Kimmins 1997) because of the requirements for using the same resources. However, northern red oak has been estimated to be a strong competitor among 14 co-occurring species in a study of New England forests (Canham and others 2006). In that study, the strong competitive effect on growth that sugar maple (*Acer saccharum* Marsh.) and white ash (*Fraxinus americana* L.) exerted on northern red oak was similar to the strength of the competitive effect that northern red oak exerted on itself. Consequently, a northern red oak planting at this density can approximate the competitive forces that occur in stands that naturally develop after disturbance.

Our results demonstrate that smaller than average planted seedlings and developing oak trees typically persist for only a few years and are subject to self-thinning when in a subordinate competitive position. However, all treatments produced at least some trees that survived through age 17. For the highest survival in planting oak, it is crucial to utilize larger and more intensively cultured oak planting stock that can initiate and maintain rapid growth to grow beyond the reach of deer early on and then to develop a favorable competitive position relative to neighboring trees.

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

ONE-YEAR RESPONSE OF AMERICAN SYCAMORE (*PLATANUS OCCIDENTALIS* L.) AND BLACK LOCUST (*ROBINIA PSEUDOACACIA* L.) TO GRANULAR FERTILIZER APPLICATIONS ON A RECLAIMED SURFACE MINE IN EASTERN KENTUCKY

Josh Brinks, John Lhotka, and Chris Barton¹

Abstract.—The establishment of intensively managed woody energy crops on reclaimed surface mine lands provides an opportunity to diversify domestic biomass sources, while increasing the productivity and economic value of underutilized land. Our objective is to test the effect of fertilization on the growth and biomass accumulation of American sycamore (*Platanus occidentalis* L.) and black locust (*Robinia pseudoacacia* L.) planted on a reclaimed surface mine. In 2008, replicated plantings of sycamore and black locust were established on Big Elk Mine in eastern Kentucky. The treatments tested include a onetime granular fertilizer application of 37 kg N, 30 kg P, and 16 kg K ha⁻¹ and an unfertilized control. Following one growing season, American sycamore exhibited significantly ($p < 0.05$) greater stem biomass in the fertilizer treatment (48.98 g) than in the control (23.52 g). Stem biomass of black locust was also greater ($p < 0.01$) in the fertilizer treatment (32.36 g) than in the control (13.20 g). Mean diameter and height growth of black locust was significantly ($p < 0.05$) greater in fertilizer treatments (6.52 mm, 20.37 cm, respectively) compared to the control (2.27 mm; 9.38 cm). American sycamore also displayed significant ($p < 0.05$) differences in diameter and height growth between fertilizer (7.86 mm; 30.63 cm) and control (4.42 mm; 15.97 cm) treatments. Mean black locust survival was 79.25 percent in control treatments and 88.07 percent in fertilizer treatments. American sycamore survival did not differ between control (96.43 percent) and fertilizer (93.81 percent) treatments. Our findings indicate that fertilizer applications to 1-year-old seedlings on reclaimed mines in Appalachia increase height, diameter, and biomass accumulation.

INTRODUCTION

Biofuel plantations provide an opportunity to diversify and increase domestic energy feedstocks (Johnson and others 2007). Currently, woody biomass is the leading domestic renewable energy supply and the U.S. Department of Energy forecasts an annual biomass supply of 1 billion tons by 2050 (English and Ewing 2002). Demand for woody biomass will be met through fuel reduction treatments, industrial waste residues, and reforestation projects (Perlack and others 2005). Woody crops intended for co-firing with coal for electricity generation will prolong coal supplies, reduce NO_x and SO_x emissions, and contribute to national woody feedstock supplies (Boylan 1996).

Feedstock plantations on reclaimed surface mines facilitate the restoration of previously forested lands. More than 90 percent of mined lands in Kentucky were forested prior to mining activities, but 98 percent of

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reclaimed areas are restored as grasslands (Cole and others 2001). This considerable shift in land cover may be offset by planting woody biofuel plantations (Casselman and others 2006, Sperow 2006). In addition to the restitution of forested landscapes, woody crops sequester atmospheric carbon and can counteract anthropogenic emissions (Dickmann 2006). Sperow (2006) found that carbon sequestration was greatest in forests as opposed to cropland or grasslands on degraded lands in Indiana, Michigan, Ohio, Kentucky, West Virginia, Pennsylvania, and Maryland.

Nutrient amendments are shown to improve feedstock yields and enhance reforestation efforts in non-mined lands (Coyle and Coleman 2005, Cobb and others 2008). Fertilizer treatments increase leaf area which enhances carbon assimilation and whole plant growth (Samuelson and others 2001). Maximizing yields will be essential as rising global populations place increasing pressures on forest resources and energy supplies. Despite elevated biomass and carbon sequestration, fertilizer applications may negatively impact on-site carbon budgets and release N_2O , which has a greater global warming potential than CO_2 (Crutzen and others 2007).

Numerous studies have quantified the impact of nutrient applications on plantation growth and survival (Coleman and others 2004, Coyle and Coleman 2005); however, more research is needed to better understand the effects of fertilizer treatments on reclaimed surface mines (Coleman and others 2004, Coyle and Coleman 2005). Moreover, an accurate assessment of the atmospheric carbon sequestration potential of reclaimed mine lands and their suitability as biofuel feedstock production sites are needed on all landscapes (Dickmann and others 1985, Rae and others 2004, Coyle and Coleman 2005, Casselman and others 2006, Cobb and others 2008). Furthermore, past studies questioned the impacts of increased resource availability on relationships among diameter, height, and biomass, but recent findings suggest nutrient amendments only accelerate ontogenetic stages and do not alter allometric relationships (Coyle and Coleman 2005). This study may further our understanding of these relationships.

The objective of this study is to test the effect of annual fertilizer application on biomass production and survival for black locust (*Robinia pseudoacacia* L.) and American sycamore (*Platanus occidentalis* L.) planted on a reclaimed surface coal mine. This paper presents the 1-year response of tree height growth, diameter growth, survival, and aboveground leafless biomass accumulation in fertilizer and control treatments.

STUDY SITE

The study is located on a reclaimed surface mine in the Cumberland Plateau at Big Elk Mine near Hazard, KY, in Knott County. The region is characterized by warm, humid summers and cool winters. The yearly rainfall average is 127 cm. The average July high temperature is 30 °C and the average January low is -5 °C. Upland oak-hickory hardwood forests dominated the site prior to being cleared for mining. The soil profile is undeveloped, compacted, and derived from shale and sandstone. Large rocks pervade the soil matrix (Conrad 2002).

The study site was previously reclaimed using smooth grading. The site was ripped parallel to future planting with a single-blade ripping shank on a D-11 bulldozer in February 2008 to alleviate compaction. Six 0.209-ha experimental plots were established following ripping. In March 2008, half of each plot was planted with American sycamore, while the other half was planted with black locust. 1-0 bare root seedlings were used. Boulders and rough topography prevented uniform seedling plantings. Mean seedling density across all

plots was 1,842 stems ha⁻¹. Following surface ripping and planting, the vegetation community was populated by old-field and other early successional plants reported at other reclaimed surface mines in the region (Bell and Ungar 1981, Holl and Cairns 1994, Holl 2002). Dominant genera are *Rubus*, *Lespedeza*, *Ambrosia*, *Oxalis*, *Cirsium*, *Vernonia*, *Leucanthemum*, *Rudbeckia*, *Festuca*, and *Coronilla*.

METHODS

FIELD METHODOLOGY

Height and basal diameter were tracked in two centrally located 0.019-ha measurement subplots containing tagged seedlings within each experimental unit (Fig. 1). Ten destructive samples (n = 60) were harvested from four 0.008-ha destructive subplots within each experimental unit. Control (C) and fertilization (F) treatments were replicated three times and randomly assigned to the six 0.209-ha experimental units. At least three border rows were present between all experimental units and subplots. All fertilized plots received 36 kg of nitrogen, 30 kg of phosphorus, and 16 kg of potassium ha⁻¹ on June 3, 2008. Glyphosate was applied to all treatments at a rate of 0.871 l ha⁻¹ with backpack sprayers during June 2008.

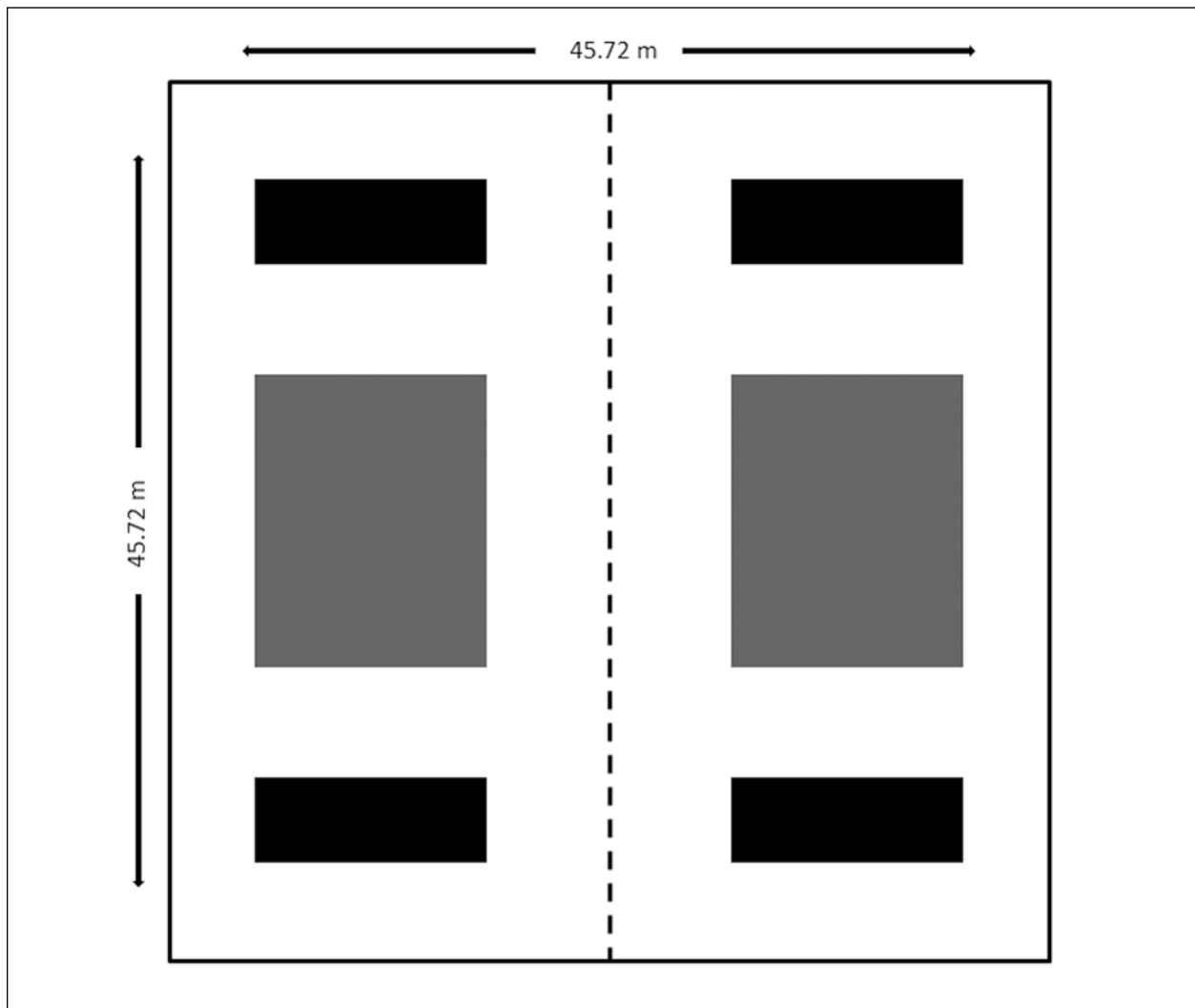


Figure 1.—Layout of individual 0.209-ha experimental unit. Dashed line represents species divide. Light-gray areas are seedling-measurement subplots for tagged and tracked trees. Black areas are destructive-harvest subplots.

FIELD AND LABORATORY MEASUREMENTS

We recorded initial basal diameters and heights in measurement subplots after planting in May 2008. Basal diameter was taken to the nearest tenth of a mm with digital calipers at two locations, 90° apart from one another, and then averaged. Measurements taken following the first growing season were recorded in January 2009. We severed five trees from each treatment and species combination at the base with loppers and harvested them in January 2009. Diameters and heights were recorded before specimens were oven-dried for 48 hours at 60 °C, and then weighed to the nearest hundredth of a gram.

STATISTICAL METHODS

Harvested trees were used to develop regression equations for predicting aboveground leafless biomass (SAS 9.1; SAS Institute Inc., Cary, NC). Stem biomass was predicted using a linearized version of a power function with seedling height and basal diameter as independent variables. Equations were then used to predict aboveground leafless biomass of trees being tracked in measurement subplots. Plot means were used in one-tailed t-tests to evaluate the effect of fertilization on height growth, diameter growth, and biomass accumulation for each species following one growing season. For analysis of biomass accumulation, we used average stem biomass as a dependent variable. Stem averages were used because area-based calculations would have differed due to varying plot densities, not disparate growth patterns. A generalized linear model was used for the ANOVA that tested for differences in seedling survival among treatment and species. The generalized linear model utilized a binomial distribution and a logit link function to account for the binomial nature of survival data (Littell and others 2002).

RESULTS

GROWTH MEASUREMENTS

Both species responded positively to granular fertilizer treatments (Table 1). Mean black locust basal diameter growth was greater in fertilizer treatments (6.52 mm) than controls (2.27 mm). American sycamore exhibited similar results (7.86 mm [F]; 4.42 mm [C]). Mean height growth of black locust was greatest in fertilizer (20.37cm) treatments compared to controls (9.38 cm). American sycamore also displayed considerable ($p < 0.05$) height differences between fertilizer (30.63 cm) and control (15.97 cm) treatments. Fertilizer applications had no significant effect on survival. However, overall survival was significantly higher for American sycamore compared to black locust. Survival of American sycamore was greater than 90 percent in both treatments following one growing season, and ranged from 79-88 percent for black locust.

Table 1.—Mean* seedling survival, growth, and stem biomass accumulation by treatment and species one growing season after planting on a reclaimed surface mine in Knott County, KY.

	Survival (%)	Diameter growth (mm)	Height growth (cm)	Stem biomass (g)
Black locust				
Control	79.25Aa	2.27a	9.38a	13.20a
Fertilizer	88.07Aa	6.52b	20.37b	32.36b
American sycamore				
Control	96.43Bb	4.42a	15.97a	23.52a
Fertilizer	93.81Bb	7.86b	30.63b	48.98b

*Uppercase letters indicate significant ($p < 0.05$) differences across treatment and species, and lowercase letters indicate significant ($p < 0.05$) differences across intra-species treatments .

REGRESSION EQUATIONS AND BIOMASS PREDICTIONS

Stem biomass was predicted from seedling diameter and height using a linearized version of a power function. No significant interactions between treatments or species with diameter or height were present. Regression equations were as follows:

$$\begin{array}{ll}\text{Black locust:} & \ln(\text{Weight}) = -4.15 + 0.74 \cdot \ln(\text{Height}) + 1.86 \cdot \ln(\text{Diameter}) \\ \text{Sycamore:} & \ln(\text{Weight}) = -4.48 + 0.74 \cdot \ln(\text{Height}) + 1.86 \cdot \ln(\text{Diameter})\end{array}$$

where weight is listed in grams stem⁻¹, height in centimeters, and basal diameter in millimeters (n = 60; r² = 0.97).

Stem biomass means for fertilizer treatments were more than double that of controls in both species (Table 1). American sycamore exhibited greater stem biomass (48.98 g [F]; 23.52 g [C]; p < 0.05) than black locust (32.36 g [F]; 13.20 g [C]; p < 0.01) during the first growing season.

DISCUSSION

SURVIVAL

The survival trend between species may be a function of ungulate browsing. The study site (Big Elk Mine) is home to the highest densities of elk throughout all of the Eastern Kentucky elk reintroduction zone (Dahl 2008), and numerous black locust trees were heavily browsed. Furthermore, despite herbicide treatments, weedy competition from *Lespedeza* spp. and *Ambrosia* spp. may have reduced seedling survival (Fortson and others 1996) and could have compounded the impacts of browsing on black locust. Fertilizer amendments may also have exacerbated weedy competition at our site.

These findings fall within ranges established by studies of sycamore in western Kentucky (Witter 1980) that reported 90-percent first-year survival in control plots and 80 percent in fertilized plots, and on a reclaimed agricultural field in South Carolina (Davis and Trettin 2006) that reported an 84-percent survival rate. Therefore, despite a heavily disturbed landscape, survival at our site was similar to other studies.

GROWTH AND BIOMASS

Response to fertilizer treatments at our site was similar to other studies. Casselman and others (2006) reported similar heights for hybrid poplars, but lower heights and diameters for mixed planted hardwoods with nutrient amendments on reclaimed surface mines in Virginia, West Virginia, and Ohio. Control mean total heights were 50.3 cm, 60.2 cm, and 65.4 cm at three reclaimed mines. Their respective heights in fertilized plots were 50.8 cm, 57.6 cm, and 126.6 cm. These findings are similar to those in our study.

Studies established on higher-quality sites in the eastern United States outperformed our plantings in total height and diameter after one growing season. A reclaimed agricultural field in South Carolina planted with American sycamore reported heights nearly three times greater than our control and fertilizer treatments (Coleman and others 2004). Van den Driessche and others (2003) witnessed negligible impacts of fertilizer treatments in aspen (*Populus tremuloides*) plantations, but heights under control, fertilizer, and irrigation treatments following 2 years were roughly 27 cm higher than our best-performing species and treatment combination (fertilized American sycamore). Growth response at our site did not compare favorably to plantations established on reclaimed agricultural fields and bottomlands. Impeded growth on reclaimed

surface mines is commonly reported on sites composed primarily of shale and sandstone overburden (Fortson and others 1996, Kost and others 1998). Despite ripping prior to planting, soil compaction may still be high relative to non-mined forested lands (Conrad 2002). Moreover, the positive impacts of surface ripping have been touted by numerous studies (Cleveland and Kjelgren 1994, Ashby 1997), but they have been questioned in some research (Kost and others 1998). Excessive soil compaction hinders root growth, nutrient uptake, and soil drainage (Andersen and others 1989). Additionally, large rocks permeate the soil matrix and inhibit root growth (Fortson and others 1996).

Aboveground leafless biomass displayed considerable gains for both species in response to fertilizer applications. Black locust increased per-stem biomass production by 145 percent while American sycamore leafless biomass rose by 108 percent. Our findings are similar to hybrid poplar plantations established on reclaimed surface mines in Ohio, Virginia, and West Virginia that exhibited a per-stem biomass of approximately 22 g stem⁻¹ in control and 50 g stem⁻¹ in fertilized plots (Casselman and others 2006). Both species at our site exceed per-stem biomass accumulation found in mixed hardwood plantations in the same study.

Establishment of woody bioenergy plantations on reclaimed mined lands may help transform underutilized, low-value lands into a biomass source that helps diversify the domestic feedstock supply. Our results indicate that the application of granular fertilizer may increase biomass accumulation of black locust and American sycamore planted on reclaimed surface mines. Increased height growth associated with fertilization may also benefit ongoing reforestation efforts in the Appalachian region by accelerating the development of young seedlings in restoration projects (Angel and others 2005, Groninger and others 2007). While our initial results appear promising, it remains to be seen whether long-term biomass production will reflect our preliminary results and whether annual fertilization can make the production of bioenergy feedstocks on reclaimed surface mines more economically feasible. Continued monitoring of the study will help address these questions.

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

EFFECT OF MECHANICAL SITE PREPARATION TREATMENTS ON OAK SURVIVAL IN A RETIRED FIELD AFFORESTATION EFFORT – FIRST-YEAR RESULTS

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Abstract.—Mechanical site preparation is frequently proposed to alleviate poor soil conditions when afforesting retired agricultural fields. Without management of soil problems, oak seedlings planted in these areas may exhibit poor survival. While mechanical site preparation methods currently employed in hardwood afforestation are proven, there is a substantial void in research comparing subsoiling, bedding, and combination plowing treatments. In this study, 1,440 bare-root Nuttall oak (*Quercus texana* Buckley) seedlings were planted in February 2008 on three Mississippi sites. All sites had comparable soils and received above-average precipitation throughout the first growing season. Four site preparation treatments were replicated at each site, with 360 seedlings planted in each. Mechanical treatments were implemented using 10-foot row centers, with treatments as follows: control, subsoiling, bedding, and combination plowing. These treatments were applied to evaluate survival. Oak seedlings exhibited excellent survival in all treatments, ranging from 96.7 to 100 percent. No significant difference in overall survival was observed among sites or site preparation treatments. The only observed difference occurred among treatments on one site.

INTRODUCTION

Oaks (*Quercus* spp.) are an important component of southern U.S. forests. Attempts at artificial regeneration frequently fail. For instance, survival of planted seedlings in retired fields is often unsatisfactory, resulting in a low percentage of oaks in established stands (McGee and Loftis 1986, Schoenholtz and others 2001, Stanturf and others 2001, Schweitzer 2004, Stanturf and others 2004). The preponderance of these failed afforestation efforts suggests a need for greater understanding and proper use of techniques in plantation establishment.

Several factors can decrease seedling growth and survival, including: soil conditions, planting techniques, seedling quality, and competing vegetation. With the exception of poor soil conditions, these problems can be alleviated at or after planting time through proper use of high-quality seedlings, as well as by applying proper silvicultural practices to enhance growth and survival. Many retired fields have compacted soils due to past land-use practices (Allen and others 2001, Gardiner and others 2002, Stanturf and others 2004). Seedling growth and survival can potentially be improved through the use of mechanical site preparation to correct compaction problems.

Subsoiling, bedding, and combination plowing are mechanical site preparation treatments which could ameliorate compaction commonly found in these areas. Subsoiling, also known as ripping, is generally

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performed by pulling a 3- to 6-inch wide shank through the soil behind a tractor. Typical cutting depth is 16 inches to 24 inches using traditional straight or parabolic shanks with or without winged tips. If winged tips are used, planting should not occur for up to 1 year (Stanturf and others 2004). This method exposes less mineral soil than other ameliorative methods of mechanical site preparation. Mineral soil is required by most pioneer hardwood species for establishment; thus, subsoiling should result in lower numbers of germinants undesirable for management objectives. Of the various mechanical treatments available, subsoiling is the least expensive and most easily performed due to availability of contractors and the simplicity of the treatment.

The objective of subsoiling is to fracture restrictive layers in the soil, such as those common to retired agricultural fields. Compacted layers are known to restrict plant development and their removal increases tree survival and growth (Russell and others 1997, Ezell and Shankle 2004, Stanturf and others 2004). Subsoiling improves seedling performance by enhancing soil moisture availability, planting depth uniformity, root development, and soil exploitation (Morris and Lowery 1988, Miller 1993, Russell and others 1997).

Significant increases in pine (*Pinus* spp.) survival and growth have been noted when seedlings were planted in subsoil-treated areas (Berry 1986, Morris and Lowery 1988, Fallis and Duzan 1994, Gwaze and others 2006). These benefits should not change when applied to planting oak seedlings. Rathfon and others (1995) reported that subsoiling improved rooting depth in the first growing season for northern red oak (*Q. rubra* L.). Improvements in root growth would be helpful under conditions of low moisture and low fertility.

Ezell and Shankle (2004) stated that subsoiling significantly increased height and groundline diameter growth during the first growing season for Shumard oak (*Q. shumardii* Buckley), water oak (*Q. nigra* L.), willow oak (*Q. phellos* L.), and green ash (*Fraxinus pennsylvanica* Marsh.) seedlings. Improved black walnut (*Juglans nigra* L.) survival has been observed on compacted mine soil after subsoiling (Ashby 1996). Twelfth-year survival of black walnut in subsoiled plots was 33 percent greater than in untreated plots (41 percent and 74 percent, respectively), and average height growth was 7.2 feet and 14.7 feet, respectively. An unspecified red oak species exhibited similar growth and survival responses. Self and others (2010) reported that when oaks were planted under intense herbaceous competition, greater survival was observed in areas receiving only a subsoiling treatment, compared to areas receiving both subsoiling and chemical site preparation. Lower survival in treatment areas receiving both subsoiling and chemical applications was thought to result from greatly increased broadleaf weed regeneration in chemically treated areas compared to areas receiving only the subsoil treatment. Moree and others (2010) observed that Nuttall oak (*Q. texana* Buckley) seedlings planted in subsoiled plots exhibited 11.2 percent greater survival than seedlings planted in untreated plots. The authors indicated that seedlings planted in subsoil trenches were better equipped to tolerate vegetative competition than flat-planted seedlings.

Bedding, also known as hipping or mounding, is typically performed utilizing a moldboard plow, offset disk, levee plow, or furrow plow (Kabrick and others 2005). Soil is turned inward, creating a planting site 3- to 6-feet wide and 6 inches to 2 feet high above the soil surface (Williams 1988, Kabrick and others 2005). Bedding is typically used in the establishment of conifer seedlings on poorly drained soils (Londo and Mroz 2001, Xu and others 2002, Kabrick and others 2005). Increased pine survival has been documented in bedded areas that were inundated and/or saturated regularly (McKee and Wilhite 1986, Morris and Lowery 1988, Xu and others 2002). Greater survival probably resulted from raising the elevation of the rooting zone. Many oak species do not tolerate poorly drained soils and could benefit from the increased aeration found on slightly higher elevations (Hodges and Switzer 1979, Hodges 1997).

Bedding can increase growth of hardwood seedlings through improved soil aeration and drainage, concentrated organic matter and nutrients, and competition control (Fisher and Binkley 2000, Kabrick and others 2005). Patterson and Adams (2003) stated that fifth-year Nuttall oak height growth and survival were greater on bedded sites than on unbedded sites. This response was stronger on heavier clay soils, indicating that the seedlings were probably benefiting from better aeration in the elevated rooting zone. Overall survival observed across bedded sites was approximately 10 percent greater than on unbedded sites.

Not all bedding research has generated positive results. Patterson and Adams (2003) did not note any significant differences in green ash growth or survival between bedded and unbedded sites. Kabrick and others (2005) observed no bedding effects on growth or survival in pin oak (*Q. palustris* L.) and swamp white oak (*Q. bicolor* Willd.) seedlings. Seifert and others (1984) reported a similar lack of effect with swamp chestnut oak (*Q. michauxii*) on a poorly drained upland flat site.

Combination plowing is accomplished by combining subsoiling and bedding treatments into one mechanical treatment. Typically, a subsoil shank or coulter is pulled in front of a bedding plow. Problems with soil compaction, poor drainage, and vegetative competition are improved, enhancing growth and survival of planted seedlings. This technique has been used in pine plantation management and can provide satisfactory results (Will and others 2002, Lincoln and others 2006, Morris and others 2006). Corresponding work has not been performed in hardwood plantings.

STUDY AREAS

This study was located on three sites. The Mississippi Department of Wildlife, Fisheries, and Parks (MDWFP) owns two of the sites: Copiah County Wildlife Management Area (WMA) and Malmaison WMA. The third site is located near Arkabutla Lake on land owned by the U. S. Army Corps of Engineers.

COPIAH COUNTY WMA

The Copiah County WMA study area is located approximately 16 miles west of Hazlehurst, MS (90.6719°W, 31.8189°N) in Copiah County and was retired from row crop production in the late 1980s. It was maintained as an opening for wildlife through mowing and disking from the time of production retirement until the initiation of this study. The study area encompasses approximately 5.25 acres. The soil series is Oaklimer silt loam (coarse-silty, mixed, thermic Fluvaquentic Dystrochrept), and is moderately well drained and moderately permeable. Average annual precipitation is 56.33 inches. Soil tests indicate that the site has silt loam texture with an average pH of 5.2.

The woody competition component prior to mechanical treatment primarily comprised sweetgum (*Liquidambar styraciflua* L.), American elm (*Ulmus americana* L.), sycamore (*Platanus occidentalis* L.), and common persimmon (*Diospyros virginiana* L.). Estimated woody coverage was less than 1 percent. Dominant herbaceous species onsite were crimson clover (*Trifolium incarnatum* L.), ladino clover (*T. repens* L.), and ryegrass (*Lolium multiflorum* Lam.). Cumulative herbaceous coverage by these species was 100 percent. Other herbaceous species occurred in small quantities.

MALMAISON WMA

The Malmaison WMA study area is located approximately 14 miles north of Greenwood, MS (90.0531°W, 33.6876°N) in Grenada County. The site was formerly used in row-crop production and retired from

agricultural production in the late 1990s. It was also maintained as an opening for wildlife through mowing and disking from the time of retirement until this study began. The study area encompasses approximately 5.25 acres. The soil series is Collins silt loam (coarse-silty, mixed, acid, thermic Aquic Cumulic Haplothent) with slopes between 0 and 3 percent, and is moderately well drained and moderately permeable. Average annual precipitation is 52.30 inches. Soil tests indicate that the site has silt loam texture with an average pH of 6.3.

Onsite woody competition prior to mechanical treatment was composed mainly of sweetgum, black willow (*Salix nigra* Marsh.), sycamore, green ash, and eastern cottonwood (*Populus deltoides* Bartr. ex. Marsh.). Estimated woody species coverage was less than 1 percent. Dominant herbaceous species onsite were ryegrass, bermudagrass (*Cynodon dactylon* L.), Brazil vervain (*Verbena brasiliensis* Vellozo.), and Carolina horsenettle (*Solanum carolinense* L.). Cumulative herbaceous coverage by these species was 100 percent. Other herbaceous species occurred in minor quantities.

ARKABUTLA LAKE

The Arkabutla Lake study area is located approximately 5 miles northwest of Coldwater, MS (90.079°W, 33.744°N) in DeSoto County. The site was in soybean production until September 2007. The study area encompasses approximately 5.25 acres. Soil series are Memphis silt loam (fine-silty, mixed, active, thermic Typic Hapludalfs) and Loring silt loam (fine-silty, mixed, active, thermic Oxyaquic Fragiudalfs). Both are well drained gray-brown Podzolic soils with slopes between 2 and 5 percent. Average annual precipitation is 47.69 inches. Soil tests indicate that the site has silt loam texture with an average pH of 6.2. Dominant herbaceous species on site are Brazil vervain, poorjoe (*Diodia teres* Walt.), and thorny amaranth (*Amaranthus spinosus* L.). Cumulative herbaceous coverage by these species was approximately 5 percent at project initiation. Other herbaceous species occurred in very small quantities.

METHODS

EXPERIMENTAL DESIGN AND PLOT ESTABLISHMENT

The study was replicated at all three sites. A randomized complete block design was utilized, and each site received its own unique installment of randomized treatments. Each of the three replicates at each site contained all possible site preparation treatments. Each replication consisted of four planting rows approximately 390 feet by 10 feet and contained 40 seedlings. Each planting row constituted a plot, and received a randomly selected site preparation treatment. All plot and tree locations were determined using a 300-foot surveyor's tape and a handheld compass. Plot ends were marked with 4-foot pieces of 3/8-inch rebar. Individual tree planting locations were marked with 15-inch pin flags.

SITE PREPARATION TREATMENTS

Four mechanical site preparation treatments were used in this study: control (no mechanical treatment), subsoiling, bedding, and combination plowing. Mechanical treatments were applied on 10-foot centers using an agricultural tractor during the first week of November 2007. Subsoiling was performed using Case International's eco-til™ 2500 subsoiler system, with trenches cut to a depth of 15 inches using a 3/4-inch straight shank with an 8 inch tiger no-till point. Bedding was performed using a furrow plow with blades set to pull a soil bed approximately 3 feet wide and between 12 and 16 inches deep. The same equipment was used for combination plowing, which consisted of pulling a soil bed over the top of subsoiled trenches.

A 2-ounce application of Oust® XP (E.I. du Pont de Nemours and Co., Wilmington, DE) was applied during March 2008 for herbaceous weed control during the growing season. The Oust® XP treatment was applied in 5-foot bands over the top of seedlings prior to budbreak. A Solo® backpack sprayer (Solo, Newport News, VA) was used for herbicide application at 10 gallons per acre. No surfactant was added in the application.

SEEDLING ESTABLISHMENT

Nuttall oak was chosen for this study. It is a commonly planted species in retired field afforestation, is commercially important, and is a good mast producer for wildlife. Seedlings were purchased from Joshua Timberlands Elberta Nursery in Elberta, AL, and were lifted mid-January 2008. Seedling specifications required 1-0 bare-root seedlings with stems between 18 and 24 inches, and required root systems to be between 8 and 10 inches long with a minimum of 8 first-order lateral roots. Mississippi State University personnel planted 1,440 seedlings (480 seedlings per site) during February 2008. Seedlings were planted at root collar depth using planting shovels. A spacing of 10 feet was chosen to give a seedling density of 435 seedlings per acre.

SURVIVAL AND PRECIPITATION OBSERVATIONS

Survival checks were recorded monthly from May 2008 through October 2008. All seedlings were inspected each month. Seedling survival was based on visual evaluation. The cambium was examined on seedlings which appeared dead to confirm survival status.

Onsite precipitation totals were recorded in 0.01-inch intervals using a RainWise® Model 111 rain gauge (Rainwise, Inc., Bar Harbor, ME) and a HOBO® Event Rainfall Logger. The HOBO® Event Rainfall Logger can record up to 80 inches of rainfall, and data were off-loaded monthly. This precipitation measuring system meets National Weather Service specifications for statistical accuracy. Precipitation totals were recorded for the entire first year and are considered as explanatory variables.

DATA ANALYSIS

Survival averages were calculated using Statistical Analysis System (SAS) software version 9.1 (SAS Institute, Cary, NC). All survival data for Nuttall oak at each site were analyzed for interaction among sites and mechanical site preparation treatments. Seedlings were pooled by site to test for overall seedling survival among sites. Seedlings were also pooled by site preparation treatment to test for overall seedling survival among site preparation treatments. PROC GLM was used to test for significance among site and treatment data. Data were analyzed using least square means (LSMEANS). Means separation was performed using Duncan's New Multiple Range test. Differences were considered significant at $\alpha = 0.05$.

RESULTS

No statistical differences were observed in overall survival of oak seedlings among sites or site preparation treatments (Tables 1 and 2, respectively). No statistical difference in survival was noted among site preparation treatments at the Arkabutla Lake or Malmaison WMA sites (Table 3). Seedlings planted in bedding treatment areas at the Copiah County WMA site exhibited significantly lower survival (96.7 percent) than seedlings in the subsoiling or combination plowing treatments (99.1 percent, and 99.2 percent, respectively). This difference was also present in the comparison among sites for bedding; the Arkabutla Lake and Malmaison WMA sites both exhibited 99.2-percent survival. No survival differences among sites were found among the control, subsoiling, and combination plowing treatments.

Table 1.—First-year Nuttall oak survival for three Mississippi sites.

Study site	Survival (%)
Arkabutla Lake	99.4 a*
Copiah County WMA	98.3 a
Malmaison WMA	99.6 a

*values within a column followed by different letters are significantly different at $\alpha = 0.05$ according to Duncan's New Multiple Range Test.

Table 2.—First-year Nuttall oak survival by site preparation type for three Mississippi sites.

Site preparation	Survival (%)
Control	99.4 a*
Subsoiling	99.2 a
Bedding	98.3 a
Combination plowing	99.5 a

*values within a column followed by different letters are significantly different at $\alpha = 0.05$ according to Duncan's New Multiple Range Test.

Table 3.—First-year Nuttall oak survival by site preparation type at three Mississippi sites.

Site preparation	Arkabutla	Copiah County WMA	Malmaison WMA
	Survival (%)		
Control	100.0 A*a†	98.3 ABa	100.0 Aa
Subsoiling	99.2 Aa	99.1 Aa	99.2 Aa
Bedding	99.2 Aa	96.7 Bb	99.2 Aa
Combination plowing	99.2 Aa	99.2 Aa	100.0 Aa

*Values followed by different uppercase letters within a column are significantly different at $\alpha = 0.05$ according to Duncan's New Multiple Range Test.

†Values followed by different lowercase letters within a row are significantly different at $\alpha = 0.05$ according to Duncan's New Multiple Range Test.

DISCUSSION

Generally, greater survival would be expected in areas receiving more intense mechanical site preparation. Several factors may have been involved in the lack of statistical differences in the survival observed in this study. These factors include good seedlings, proper planting, the Oust® XP application, and above-average rainfall. Of particular interest is that all three sites received rainfall well above average for the year. Rainfall for the Arkabutla Lake, Copiah County WMA, and Malmaison WMA sites was 73.23 inches, 69.24 inches, and 60.85 inches, respectively. These values represent 25.54 inches, 12.91 inches, and 8.55 inches more than the average annual expectation, respectively. This additional rainfall was distributed relatively evenly across the year.

A difference in survival was observed among sites for the bedding treatment and among mechanical treatments at the Copiah County WMA site. While the slightly lower survival in bedded areas was significant, this difference was very small (2.5 percent) and was not noted on the other two sites. Even if similar results had been found on the other two sites, the difference was so slight that management strategy would not be impacted. Additionally, the Copiah County WMA site exhibited slightly lower survival for most of the treatments compared to the other two sites. Thus, the lower survival observed in this particular treatment was possibly a result of site differences.

The results of this study concur with those of previous studies (Haynes and others 1995, Moree and others 2010). When high-quality seedlings are planted properly, receive proper growing-season herbaceous weed control, and are supplied with good growing conditions, good survival can be achieved. Also, while

mechanical treatment may result in growth differences, it may not be necessary for initial survival if rainfall is adequate. While hardwood plantings are never guaranteed, land managers can control most of the inputs into an afforestation effort. If these inputs are correct and nature cooperates, excellent survival is possible.

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CONTRASTING THE EFFECTS OF ORGANIC MATTER REMOVAL AND SOIL COMPACTION ON ROOT BIOMASS OF 9-YEAR-OLD RED OAK, WHITE OAK, AND SHORLEAF PINE IN A MISSOURI OZARK FOREST

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Abstract.—Nine-year old artificially regenerated red oak (*Quercus rubra* L.), white oak (*Q. alba* L.), and shortleaf pine (*Pinus echinata* Mill.) trees were excavated from plot borders of a U.S. Forest Service long-term soil productivity study in the Carr Creek State Forest near Ellington, MO, to quantify treatment effects on above- and belowground tree biomass. The study consists of factorial combinations of soil compaction and organic matter removal treatments that are replicated three times. Seventy-two trees were removed from treatments containing two levels each of soil compaction (SC) and organic matter removal (OMR) with weed control (WC) and without weed control (NWC). Except for red oak, neither SC nor OMR alone affected root or shoot biomass production (weights) for trees in the study. However, biomass for both root and shoot were affected by interactions between SC and OMR with and without weed control. Regardless of the SC or OMR treatments, root biomass was higher with WC than NWC. Only the root:shoot ratio of red oak was affected by treatments, where it was higher for trees in the severe SC treatment than for trees in the no soil compaction treatment. Overall, measurements of above- and belowground biomass on plot border trees indicate that after nine growing seasons, site productivity has been affected more by the WC treatment than by SC or OMR.

INTRODUCTION

Roots utilize space in the soil. The more space roots control, the more potential resources they control. Coarse root (>2 mm in diameter) production reflects the vigor of the tree's growth (Theodorou and Bowen 1993) and it increases with increasing resource availability (Albaugh and others 1998). However, soil disturbance can influence the behavior of tree roots, thereby impacting forest productivity. Reductions in soil porosity through compaction and losses in nutrients through site organic removal are considered potentially detrimental consequences of forest operations (Kozlowski 1999, Kranabetter and others 2006). Kranabetter and others (2006) go on to say, "Quantifying the effects of these disturbances on soil properties and tree growth is an ongoing objective of the long-term soil productivity (LTSP) studies" (Powers 2006). While soil disturbance treatments will likely affect specific chemical and physical properties that can be characterized by measurement of soil properties, tree responses to disturbance treatments will be based on impacts seen in aboveground biomass rather than in belowground biomass.

Early LTSP treatment responses were noted by differences in tree growth associated with differences in soil compaction (SC) and understory control (WC) and/or their interactions; organic matter removal (OMR)

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resulted in fewer responses (Fleming and others 2006, Page-Dumroese and others 2006). Moderate soil disturbance has been found to create better growing conditions for some, but not all, tree species on medium- or coarse-textured soils (Brais 2001, Gomez and others 2002, Kabzems and Haeussler 2005, Ponder 2007). The main objective of this study was to assess the root and shoot biomass of trees planted in the Missouri LTSP study in plots treated with two levels of soil compaction (none and severe), two levels of organic matter removal (bole only and whole tree plus forest floor removed), and two levels of understory control or weed control (with and without weeds) using trees excavated from plot borders.

METHODS AND MATERIALS

SITE AND STUDY DESCRIPTION

This work was conducted using trees in the LTSP Study, on the Missouri Department of Conservation's Carr Creek State Forest in Shannon County. The study is located on upper slopes (slopes range from 20 to 28 percent) of two north-facing parallel ridges (Ponder and Mikkelsen 1995). Soils are predominately Clarksville (loamy-skeletal, mixed, mesic Typic Paleudults). The soil is primarily derived from Ordovician and Cambrian dolomite with some Precambrian igneous rock (Missouri Geological Survey 1979). Mean annual precipitation and temperature are 112 cm and 13.3 °C, respectively. Moisture drains easily through the soil to subsurface channels. The pre-harvest forest was occupied by a mature, second-growth oak-hickory forest type. Site index ranged from 74 to 80 based on *Quercus velutina* Lam. at 50 years (Hahn 1991). The Missouri LTSP study has three levels of OMR—bole only (BO), whole tree (WT), and whole tree plus forest floor (WTFF)—and three levels of SC: none (NC), medium (MC), and severe (SSC). The factorial design was replicated three times. One-year-old red oak (*Quercus rubra* L.), white oak (*Q. alba* L.), and shortleaf pine (*Pinus echinata* Mill.) were planted 3.66 m apart in rows 3.66 m apart at a ratio of three oaks of each species to one shortleaf pine in spring 1994. For the first 2 years, a manually operated backpack sprayer was used to spray all plots annually in late spring with a mixture of glyphosate [N-(phosphonomethyl) glycine] and simazine (6-chloro-*N,N'*-diethyl-1,3,5-triazine-2,4-diamine) at the recommended rate of 3 and 3.6 kg a.i. ha⁻¹, respectively, to control weeds and to enhance the establishment of 1-year-old planted tree seedlings. Beginning in year 3, the understory (weeds) in half of all plots was controlled while the other half of the plot had no weed control.

For this experiment, fewer treatments were used from the original nine in the LTSP study. Treatments were as follows:

- two levels of OMR: merchantable boles only (BO) and WTFF
- two levels of SC: no soil compaction (NSC) and SSC
- two levels of WC: half of each plot treated to control weeds (WC) and the other half untreated so weeds were free to grow (NWC)

Plots are approximately 0.4 ha in size and randomly selected trees in the border rows from 12 of the 27 plots were used.

Excavating Trees

Trees selected for excavation were located within the treated buffers surrounding the core measurement plots. Heights and diameters were determined and trees were excavated using a backhoe in the fall after several killing frosts and after their ninth growing season. Intact trees (above- and belowground parts) were excavated to recover maximum root (root crowns) volume (3 trees x 2 subplots x 12 plots = 72 trees, or 24 each of red

oak, white oak, and shortleaf pine) with minimal damage to remaining trees. Trees were excavated within a 1.5- x 1.5-m square about 1 m deep. The goal was to maximize the removal of roots in the 2.25-sq m area. For most trees, the backhoe disturbed the soil within the measured area sufficiently to allow trees to be lifted from the soil matrix by two individuals. For the larger trees, soil was removed around trees before the backhoe bucket was able to successfully remove the tree. Before roots and shoots were put into separate bags, the excavated trees, including root ball (soil and root) and shoot were cut into lengths when necessary; put in large, clear, thick-walled plastic bags; labeled according to date, plot, subplot, tree number, and name; and closed with plastic ties for transport to the U.S. Forest Service Laboratory at Lincoln University in Jefferson City, MO. Excavations and areas of disturbed soil were filled and/or leveled to initial ground height.

Preparation for Drying and Weighing Roots and Shoots

At the laboratory, bags containing shoot parts were opened and positioned on benches in the greenhouse for air drying. Roots were washed in a gentle stream of tap water from a garden hose after roots were soaked in garbage containers containing tap water overnight. After washing, roots were placed in newly labeled plastic bags and positioned on benches in the greenhouse for air drying. Roots and shoots were allowed to air dry in the greenhouse for approximately 6 months (last week in December 2003 to third week of June 2004). Greenhouse temperature was kept at 71 °C for five days during the third week in June 2004 before root and shoot weights were determined using an Ohaus digital scale (Ohaus Corporation, Pine Brook, NJ).

Statistical Analysis

Root and shoot biomass (weights) and growth measurements were analyzed as a 2 x 2 factorial split-plot for weed control treatments replicated three times using PROC GLM procedure (SAS 9.1, SAS Institute, Cary, NC) and root and shoot means are reported. Statistical significance was set at $\alpha = 0.05$.

RESULTS

MAIN TREATMENT EFFECTS

After 9 years, controlling the understory (weeds) on plots of planted trees contributed significantly to increases in shoot diameter for all three species excavated in the border of plots in this study compared to border trees on plots not treated to control weeds. Root and shoot weights were higher for trees in WC plots by 80 and 82 percent, by 68 and 71 percent, and by 64 and 43 percent, respectively for red oak, white oak, and shortleaf pine compared to these species in the NWC treatment (Table 1). But shoot height increases attributed to weed control for trees in WC treatment were insignificant compared to NWC treatment (Table 2). Also, neither shoot height nor diameter was affected by OMR or SC. Except for the shoot dry weight of red oak, however, there were significant interactions between OMR and WC and/or SC, thus yielding two- or three-way interactions.

Treatment Interactions for Organic Matter Removal

Shoot dry weight was not affected by OMR alone for any tree species, but there was a significant three-way interaction (WC x OMR x SC, $P=0.0184$) for white oak (Table 1). Shoot dry weight was higher with WC than without, regardless of the level of OMR and SC. Total root biomass was higher for red oaks in the BO treatment than for red oaks in the WTFF treatment, but the two-way interaction (WC x OMR, $P=0.0172$) showed that the mean root dry weight was higher for the BO with WC than with the WTFF treatment with WC. Neither OMR alone nor interactions with WC or SC affected shoot or root weight of shortleaf pine (Table 2).

Table 1.—Dry weights of stem and root biomass and root:shoot ratio (mean $\pm$ standard deviation) for excavated red oak, white oak, and shortleaf pine planted in a Missouri clear-cut with organic matter removal, soil compaction, and understory control treatments.

Treatment	Biomass		Root/shoot ratio
	Stem (kg)	Root (kg)	
Red oak			
Organic matter removal (OMR)			
Bole only (BO)	6.4(1.4)a*	4.3(4.4)a	0.8(0.2)a
WT + forest floor (WTFF)	4.4(2.4)a	2.6(2.6)b	0.9(1.8)a
Soil compaction (C)			
No soil compaction (NSC)	6.5(1.7)a	4.5(4.5)a	0.8(0.3)a
Severe soil compaction (SSC)	4.2(2.1)a	2.4(2.4)b	1.0(0.6)b
Weed control (WC)			
Weed control (WC)	8.0(2.1)a	6.3(5.5)a	0.8(0.2)a
No weed control(NWC)	1.4(0.2)b	1.3(1.2)b	0.9(1.3)a
<i>p-values</i> for treatment interactions [‡]			
OMR x C	0.8054	0.1158	0.4814
WC x OMR	0.8884	0.0253	0.4243
WC x SC	0.3165	0.0013	0.5751
WC x OMR x SC	0.0675	0.4171	0.6159
White oak			
Organic matter removal (OMR)			
Bole only (BO)	3.7(0.9)a	3.8(1.1)a	1.4(0.4)a
WT + forest floor (WTFF)	3.7(1.1)a	3.3(0.9)a	1.0(0.3)a
Soil compaction (C)			
No soil compaction (NSC)	3.5(0.8)a	3.3(0.8)a	1.4(0.2)a
Severe soil compaction (SSC)	4.0(1.3)a	3.7(1.3)a	1.0(0.4)a
Weed control (WC)			
Weed control (WC)	6.0(1.4)a	5.4(1.4)a	0.9(0.5)a
No weed control (NWC)	1.6(0.6)b	1.7(0.7)b	1.4(0.3)a
<i>p-values</i> for treatment interactions [‡]			
OMR x C	0.4564	0.4070	0.4701
WC x OMR	0.1333	0.3009	0.6441
WC x SC	0.3954	0.3545	0.3512
WC x OMR x SC	0.0002	0.0160	0.0881
Shortleaf pine			
Organic matter removal (OMR)			
Bole only (BO)	15.3(5.9)a	5.5(1.3)a	0.4(0.8)a
WT + forest floor (WTFF)	17.5(6.3)a	6.9(1.7)a	0.6(0.8)a
Soil compaction (C)			
No soil compaction (NSC)	19.0(6.0)a	6.3(1.4)a	0.4(0.3)a
Severe soil compaction (SSC)	13.9(6.4)a	6.0(0.9)a	0.5(1.2)a
Weed control (WC)			
Weed control (WC)	26.0(8.5)a	9.0(1.7)a	0.4(0.3)a
No weed control (NWC)	9.8(5.0)b	3.3(1.1)b	0.5(0.8)a
<i>p-values</i> for treatment interactions [‡]			
OMR x C	0.5633	0.2317	0.2252
WC x OMR	0.5407	0.1398	0.3040
WC x SC	0.0158	0.0014	0.9309
WC x OMR x SC	0.3225	0.4029	0.5488

*Within each column, mean values for treatment differences for a treatment category with the same letter are not significantly differently at $\alpha = 0.05$ according to Tukey test.

[‡]*p-values* for treatment interactions greater than 0.05 are not significantly different.

Table 2.—Dry weights of stem and root biomass (mean $\pm$ standard deviation) for excavated red oak, white oak, and shortleaf pine planted in a Missouri clearcut with organic matter removal, soil compaction, and understory control treatments.

Treatment	Red oak		White oak		Shortleaf pine	
	Stem	Root	Stem	Root	Stem	Root
	kg					
Organic matter removal (OMR)						
Bole only	6.4(4)a*	4.3(2)a	3.7(2)a	3.8(1)a	13.6(5)a	5.5(1)a
WT + forest floor	4.4(2)a	2.6(1)b	3.7(1)a	3.3(1)a	14.0(5)a	6.9(1)a
Soil compaction (C)						
No compaction	6.5(3)a	4.5(2)a	3.5(1)a	3.8(1)a	16.5(6)a	6.3(1)a
Severe compaction	4.2(1)a	2.4(1)b	4.0(1)a	3.3(1)a	11.1(4)a	6.0(1)a
Weed control (WC)						
Weed control (WC)	8.0(5)a	6.3(2)a	5.7(2)a	5.4(1)a	17.1(5)a	9.0(2)a
None (NWC)	1.4(1)b	1.3(1)b	1.6(1)b	1.7(1)b	9.8(5)b	3.3(1)b
<i>p</i> -values for treatment interactions [‡]						
OMR x C	0.5143	0.6621	0.4035	0.1437	0.2348	0.2562
WC x OMR	0.9283	0.2892	0.4997	0.3009	0.9688	0.1398
WC x C	0.5302	0.0652	0.6662	0.3545	0.0769	0.0014
WC x OMR x C	0.3819	0.8167	0.1094	0.0160	0.9180	0.4029

*Within each column, mean values for treatment differences for treatment categories with the same letter are not significantly different at $\alpha = 0.05$ according to Tukey test.

[‡]*p*-values for treatment interactions greater than 0.05 are not significantly different.

Treatment Interactions for Soil Compaction

While shoot dry weights for both red oak and shortleaf pine tended to be higher for trees in the NC treatment than in the SSC treatment, soil compaction alone did not significantly affect shoot dry weights for any of the three species (Table 2). There was a significant interaction for shortleaf pine between WC and SC ($P=0.0074$). With NC and WC, shortleaf pine shoot dry weights were nearly two times higher than that for shoots in the SSC treatment with or without WC. The root weights of red oak and shortleaf pine were significantly affected by the interaction between SC and WC ($P=0.0007$ and 0.0017 , respectively). Both had higher total root biomass in the NC + WC treatment than in the SSC + NWC treatment. But with NWC, root weights were similar regardless of SC treatment for red oak and heavier for shortleaf pine roots in the SSC + NWC treatments than in the NC + NWC. Root weights of white oak were affected by a three-way interaction (WC x OMR x SC, $P=0.0160$). Regardless of the compaction and organic matter removal treatment, root weights were higher with WC than with NWC.

Among tree species, the root:shoot ratio determinations between treatments were significant ($P=0.05$) only for SC for red oak (Table 1). The allocation of aboveground and belowground biomass among tree species responded differently to OMR and SC. With no soil compaction, red oak in the BO treatment had a distribution of 64 to 36 percent aboveground to belowground biomass (root:shoot ratio = 0.55) compared to 53 and 47 percent for aboveground and belowground biomass with severe compaction (root:shoot ratio = 0.90). In the WFFF treatment with no compaction, the percent biomass distribution aboveground and belowground for red oak was 53/47 (root:shoot ratio = 0.89) and was 77/23 (root:shoot ratio = 0.30) with severe compaction.

DISCUSSION

Soil bulk density measurements reported for the LTSP study 5 years after establishment showed that soil compaction had effectively increased bulk density of severe compaction treatments by 22, 29, and 26 percent for the 10-, 20-, and 30-cm depth increments over no compaction treatments (Ponder and others 1995). However, the duration of effects of soil compaction on bulk density increases for the fine-textured cherty soil on this site is not known (Ponder 2004). Research has shown that substantial increases in bulk density of fine-textured soils may (Powers and others 2005) or may not (Sanchez and others 2006) affect tree growth.

For the present study, except for the root dry weight of red oak and significant interactions between treatments, there were no main treatment differences for either OMR or SC (Tables 1 and 2). Fleming and others (2006) conducted a regional comparison of tree growth at age 5 for all the replicates of the LTSP study at the various locations, including the Missouri site, and reported that growth for trees in the Missouri study plots was not affected by OMR, but SC increased shortleaf pine diameter. By age 9, differences between NC and SSC for the diameter at breast height of shortleaf pine remained significant based on measurement of core plot trees (Ponder 2007). However, this treatment difference for shortleaf pine growth in the larger LTSP study is not reflected in the biomass of excavated border trees. In the present study, one reason for some of the lack of biomass difference of excavated trees in OMR and SC treatments was likely due to restricting excavated trees to border rows; as border trees, some trees apparently did not meet all criteria of core plot trees. Sample trees in the border rows were preselected, taking precautions to minimize erosion and not cause damage to core plot trees from excavating equipment. However, in another LTSP study in North Carolina (Ludovici 2008), 18 planted loblolly pine trees were excavated in the border rows of plots. While tree growth measurements for excavated trees were not significantly different for OMR or SC treatments, growth measurements of excavated trees were consistent with plot tree measurements, which also showed no growth differences between treatments (Sanchez and others 2006).

Soil moisture can often be limited by the presence of understory competition (Gautam 1998). Soil moisture data collected 3 years before trees were excavated showed that soil moisture stress was highest in the NWC plots (Ponder 2004). Soil moisture stress was also higher for NC with NWC than for SSC with NWC treatments. Soil compaction can adversely impact root penetration and create water stress in trees. Blouin and others (2008) in a greenhouse study using lodgepole pine (*Pinus contorta* Dougl. Ex. Loud. var. *latifolia* Engelm.) seedlings reported that compaction influenced seedling growth and biomass at dry water content, but not at wet water contents. In another greenhouse study, Siegel-Issem and others (2005) grew ponderosa pine (*Pinus ponderosa* var. *scopulorum* Dougl. Ex. Laws), shortleaf pine (*Pinus echinata* Mill.), and loblolly pine (*Pinus taeda* L.) seedlings in PVC cylinders and reported that root growth decreased with compaction, with water regulating the effect of compaction on all species. In the present study, the greater root biomass for all three tree species in the WC treatment compared to the NWC treatment can likely be attributed to higher soil moisture and nutrient availability.

The root:shoot ratio was not altered significantly by weed control treatments for any species, despite the difference in growth patterns for above- and belowground biomass between the two treatments (Table 1), suggesting a functional equilibrium between root and shoot (Drew and Ledig 1980, Clinton 1990). Coyle and Coleman (2005) concluded that biomass accumulation does not favor shoots in developing stands, even with favorable soil resources. Other studies show that resource availability in developing stands has little or no effect on relative belowground biomass accumulation (Drew and Ledig 1980, Gebauer and others 1996,

Coleman and others 1998). In simply reporting total root biomass in the present study, the author did not account for the more responsive measurement of root activity—biomass allocation—as in the results of a similar study reported by Ludovici (2008).

CONCLUSIONS

Root and shoot biomass (weights) were affected by interactions between soil disturbance treatments consisting of two levels of OMR and SC with and without WC. All significant interactions involved weed control. Results from this work demonstrate that controlling understory vegetation in forest stands can increase root and shoot growth. Except for red oak, neither OMR nor SC alone affected root or shoot biomass production for trees in the study. These findings suggest that growth rates of regenerating red oak stands could be reduced when significant amounts of biomass are removed and when soils are compacted during harvesting operations. Also, these results suggest that because white oak appears less sensitive to disturbance caused by organic matter removal and soil compaction compared to red oak, regenerating severely disturbed sites to white oak could be advantageous.

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ADAPTATION AND VALIDATION OF THE REGEN EXPERT SYSTEM FOR THE CENTRAL APPALACHIANS

Lance A. Vickers, Thomas R. Fox, David L. Loftis, and David A. Boucugnani¹

Abstract.—REGEN is an expert system that predicts future species composition at the onset of stem exclusion using preharvest stand conditions. To extend coverage into hardwood stands of the Central Appalachians, we developed REGEN knowledge bases for four site qualities (xeric, subxeric, submesic, mesic) based on relevant literature and expert opinion. Data were collected from 48 paired stands to calibrate our preliminary REGEN knowledge bases. Data from 17 additional paired stands were used to independently validate the final performance of the model. Predictions for species groups were within 4 percentage points of measured values on average, and model error was typically no more than approximately 20 percentage points for any species group. These independent validation results confirmed the potential of REGEN to predict species composition of stands regenerated using the clearcut method in the Appalachians of Virginia and West Virginia.

INTRODUCTION

The mixed hardwood forests of the Appalachians are diverse and complex ecosystems (Braun 1950, Miller and Kochenderfer 1998, Yarnell 1998). Recent inventories have shown that many of these forests across the Appalachian Region are maturing (Oswalt and Turner 2009). Given the many species assemblages that occur throughout the Appalachians (Eyre 1980), several silvicultural systems can potentially be used to regenerate these maturing forests. Prior to implementing the silvicultural action aimed at fostering regeneration, however, an estimate of regeneration potential should be obtained to ensure landowner objectives can be met. Several approaches have been used to model regeneration in the historically oak-dominated ecosystems of the eastern United States (Rogers and Johnson 1998). Probabilistic models and management guidelines have been developed to evaluate regeneration potential in the Missouri Ozarks (Sander and others 1984), the Alleghenies (Marquis and others 1992), the Southern Bottomlands (Belli and others 1999), and the Central Appalachians (Steiner and others 2008). In some cases, multi-species, regional predictive models have been developed (e.g., Waldrop and others 1986, Dey 1991), but such models are not available for all areas.

REGEN, a multi-species, regional predictive model, has been developed for the Southern Appalachians (Loftis 1989). The REGEN model is an expert system designed to predict the future species composition of dominant and codominant stems at the onset of stem exclusion using a numerical ranking system of species' competitiveness in conjunction with preharvest stand characteristics. To generate predictions in REGEN, a fixed-radius plot survey of advance reproduction is required. Advance reproduction is categorized into five pre-defined size classes: germinant (newly germinated seedlings), small seedlings (<2 feet tall), medium

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seedlings (2-4 feet tall), large seedlings (>4 feet tall), and potential stump-sprouts (trees >4 feet tall and ≥ 2 inches diameter at breast height [d.b.h.]).

Each size class of advance reproduction for each individual species in REGEN is assigned a competitive rank, ranging from one to eight. Stems that are given a rank of one are considered to be the most competitive, and stems given a rank of eight are considered to be the least competitive. During the prediction process, a predetermined number of the highest-ranked stems of existing advance reproduction at each plot are selected as “winners.” These winning stems are expected to be in a dominant or codominant crown position when the stand reaches stem exclusion following a regeneration harvest. Constant or logistic probabilistic parameters are used to establish stump-sprouts, root-suckers, and new seedlings that germinate following harvest. A REGEN Knowledge Base (RKB) contains the competitive rankings and probabilistic parameters used to process input data. These RKBs are modular, which allows REGEN to be adapted to different scenarios by creating custom RKBs. Boucugnani (2005) describes the development of the computer adaptation of REGEN and provides documentation for the underlying framework of the model in greater detail.

Efforts are currently under way to expand REGEN applicability throughout the Appalachians. The focus of this article is our assessment of REGEN’s potential to predict future species composition in the Appalachian hardwood forests of Virginia and West Virginia following clearcutting.

STUDY AREA

This study was conducted on the Appalachian Plateau and Ridge and Valley Physiographic Provinces (Fenneman 1938) of Virginia and West Virginia (Fig. 1). Sample sites were located within four counties in Virginia (Craig, Giles, Montgomery, Wise) and six counties in West Virginia (Fayette, Greenbrier, Nicholas, Randolph, Tucker, Webster).

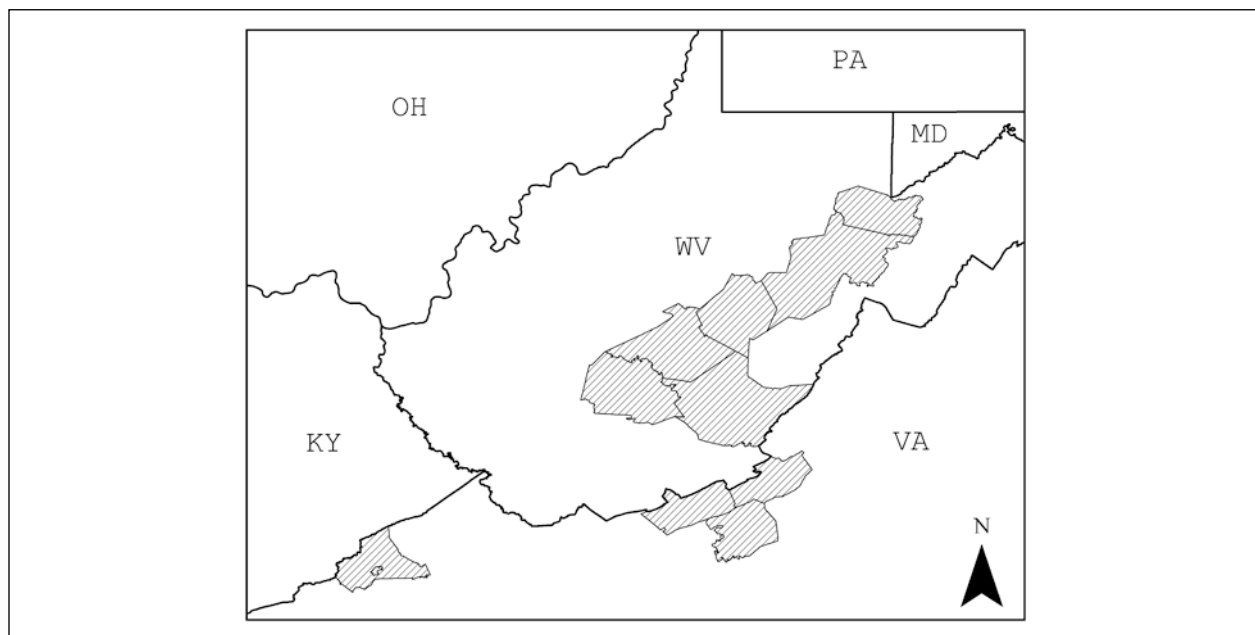


Figure 1.—Location of sample sites in the Central Appalachians. Shaded areas represent counties in which paired stands were located.

METHODS

To adapt REGEN to the Appalachians of Virginia and West Virginia, we created four preliminary RKBs. The four RKBs were designed to predict regeneration on xeric, subxeric, submesic, and mesic sites. The initial rankings and parameters for these four RKBs were based on relevant literature, including silvical characteristics (Burns and Honkala 1990), observed species composition following clearcutting (e.g., Beck and Hooper 1986, Ross and others 1986, Loftis 1989), and site productivity interactions (e.g., Doolittle 1958). The information from these sources and others was modified and supplemented as necessary based on the authors' experience to create rankings and parameters for 33 tree species in the four custom RKBs. These preliminary RKBs were revised based on the results of several iterative trials of model comparisons using field data collected from 48 paired stands across Virginia and West Virginia (Vickers 2009).

Following the calibration and assessment of the four knowledge bases, an additional, independent dataset was collected to validate the performance of REGEN. Field data were collected using a paired-stand sampling approach. This approach utilized sample sites that included both a mature, hardwood-dominated stand free from obvious recent disturbance and an adjacent regenerating clearcut stand with similar site characteristics. According to available landowner records, the mature stands were at least 70 years old, and the regenerating clearcut stands ranged from 5 to 18 years old at the time of sampling. This approach assumed that the two separate management units were once one contiguous stand of similar composition and productivity. Therefore, the mature stands were expected to regenerate in a similar fashion to their paired regenerating clearcut stands if harvested similarly. To ensure similar site productivity between paired mature and regenerating clearcut stands, we obtained estimates of site index using the Forest Site Quality Index (Meiners and others 1984). We also classified paired stands into site classes based on indicator species, as proposed by McNab and others (2002). We applied the species moisture weights from McNab and others (2002) to a list of all tree species present that were ≥ 2 inches d.b.h. in each mature stand to classify each paired stand into one of four site classes: xeric, subxeric, submesic, or mesic.

Seventeen paired stands were sampled in September and early October 2009. In the mature stands, one fixed-radius plot per acre was established with a maximum of 20 plots per stand. Plots were established using a systematic grid with the distance from the stand boundary to the first plot determined randomly, but no less than 132 feet. Advance reproduction was measured in a 0.01-acre circular plot by species and pre-defined REGEN height class (Large: > 4 feet, Medium: ≥ 2 feet, Small: < 2 feet, Germinant: newly germinated seedlings), and d.b.h. of all stems ≥ 2 inches d.b.h. within the plot was measured to account for potential stump-sprouts. At each fixed-radius plot center, a basal area sampling point was also established, on which the basal area of all stems ≥ 2 inches d.b.h. was measured using a 10 BAF prism. Regenerating clearcut stands were sampled at a density of one fixed-radius plot per acre, using 0.004-acre plots. In the regenerating clearcut stands, regeneration was tallied by species, stem origin (seed or sprout), and crown class (dominant, codominant, intermediate, suppressed). Given the complex assemblages of species that occur across this area, we created eight tree species groups after the data were collected to simplify data analysis and presentation: 1) black cherry, 2) conifers, 3) maples, 4) midstory, 5) oaks, 6) other overstory, 7) pioneer, and 8) yellow-poplar (Table 1). These species groups contain either a single species that was numerous throughout the study area or a collection of species that are expected, for the most part, to occupy similar stand structural positions.

Initial tests indicated the data were non-normal; therefore, all analyses were conducted using nonparametric tests. These analyses were conducted in SAS® version 9.2 software using the UNIVARIATE procedure (SAS Institute Inc., Cary, NC).

Table 1.—Species groupings used in this study for the Central Appalachians.

Group	Species
Black Cherry	black cherry (<i>Prunus serotina</i> Ehrh.)
Conifers	e. white pine (<i>Pinus strobus</i> L.), hemlock (<i>Tsuga</i> spp.), pitch pine (<i>Pinus rigida</i> Mill.), shortleaf pine (<i>Pinus echinata</i> Mill.), Table Mountain pine (<i>Pinus pungens</i> Lamb.), Virginia pine (<i>Pinus virginiana</i> Mill.)
Maples	red maple (<i>Acer rubrum</i> L.), sugar maple (<i>Acer saccharum</i> Marsh.)
Midstory	Am. beech (<i>Fagus grandifolia</i> Ehrh.), blackgum (<i>Nyssa sylvatica</i> Marsh.), dogwood (<i>Cornus</i> spp.), e. hophornbeam (<i>Ostrya virginiana</i> [Mill.] K. Koch.), sassafras (<i>Sassafras albidum</i> [Nutt.] Ness.), serviceberry (<i>Amelanchier</i> spp.), sourwood (<i>Oxydendrum arboreum</i> [L.] DC.), striped maple (<i>Acer pensylvanicum</i> L.)
Oaks	black oak (<i>Quercus velutina</i> Lam.), chestnut oak (<i>Q. prinus</i> L.), n. red oak (<i>Q. rubra</i> L.), scarlet oak (<i>Q. coccinea</i> Muenchh.), white oak (<i>Q. alba</i> L.)
Other Overstory	ash (<i>Fraxinus</i> spp.), basswood (<i>Tilia</i> spp.), cucumbertree (<i>Magnolia acuminata</i> L.), Fraser magnolia (<i>Magnolia fraseri</i> Walt.), hickory (<i>Carya</i> spp.), yellow buckeye (<i>Aesculus flava</i> Aiton), yellow birch (<i>Betula alleghaniensis</i> Britton)
Pioneer	black locust (<i>Robinia pseudoacacia</i> L.), pin cherry (<i>Prunus pensylvanica</i> L.f.), sweet birch (<i>Betula lenta</i> L.)
Yellow-poplar	yellow-poplar (<i>Liriodendron tulipifera</i> L.)

To assess the overall performance of the model, the predicted future species composition of dominant and codominant stems from REGEN (which was based on advance reproduction from the mature stands) was compared to the measured dominant and codominant species composition (which was taken from regenerating clearcut stands) for each species group at each paired stand as a percentage. For example, if REGEN predicted that black cherry will make up 25 percent of the future dominant and codominant stems following harvest in a given mature stand, and we measured that only 20 percent of dominant and codominant stems were black cherry in the paired regenerating clearcut stand, then the difference (model error) for black cherry at that sample site was 5 percentage points. This type of comparison was completed for each species group at each sample site. The Wilcoxon Signed Rank test, a nonparametric alternative to a paired *t*-test, was used to test for differences between measured and predicted samples for each species group across all paired stands (Ott and Longnecker 2001). Because all paired stands in our study fell on either subxeric or submesic sites, only those two RKBs were tested with field data. Trends observed in the subxeric and submesic site classes were previously extended into the xeric and mesic RKBs according to the authors' expectations, but those RKBs have not been tested.

RESULTS AND DISCUSSION

An example of our final rankings for the most competitive stems of a few common species across all site classes is provided in Table 2. As might be expected, species that are more xerophytic are generally considered more competitive on xeric and subxeric sites, while more mesophytic species are generally ranked higher on mesic and submesic sites. The final competitive rankings did not always progress in rank across the site-quality gradient as we initially assumed, but these final rankings were found to be the best fit to the calibration data that we were able to achieve during the revision process (Vickers 2009). In addition to developing a competitive ranking between species across the four site classes, we also had to develop rankings for the different size classes of advance reproduction within a single species. The rankings within a species are fairly

Table 2.—Example REGEN competitive rankings for the most competitive forms of several common tree species in the Central Appalachians across four site qualities. Species ranked 1 are considered the most competitive on a given site quality, those ranked 8 are considered the least competitive.

REGEN Competitive Rank	Xeric	Subxeric	Submesic	Mesic
1	Virginia pine	red maple sweet birch yellow-poplar	black cherry basswood sweet birch yellow-poplar	black cherry basswood sweet birch yellow-poplar
2	red maple	black cherry	northern red oak	
3	sassafras northern red oak	northern red oak	red maple	red maple
4		Virginia pine		northern red oak
5				
6	black cherry yellow-poplar	sassafras basswood	sassafras	
7	sweet birch			sassafras
8	basswood		Virginia pine	Virginia pine

straightforward; i.e., stump-sprouts are more competitive than large stems, which are more competitive than medium stems, and so forth. However, the distance between size class rankings for a species may not be uniform. For example, in the submesic RKB, the rankings for yellow-poplar range from one to five consecutively from stump-sprouts to germinants. In contrast, for northern red oak, a stump-sprout is given a rank of two, a large stem is ranked five, and a medium stem is ranked eight. Given the limited number of rankings available, smaller stems of northern red oak are not considered to be viable sources of regeneration in the submesic RKB. For a complete set of finalized rankings and parameters for all four RKBs, see Vickers (2009).

The competitive rankings that we developed in the REGEN model for the Central Appalachians performed better than was expected. The predicted and measured values for each species group at each sample site were used to calculate the mean species composition values and mean model error across all 17 sample sites, which are displayed in Table 3 and Figure 2. Across all 17 paired stands, the species composition predicted by REGEN was within about 4 percentage points of the measured species composition, on average, for all eight species groups used in this study (Table 3). Tests for differences in sample distributions indicated that only the yellow-poplar species group's distribution of measured values was statistically different from the distribution of predicted values using $\alpha = 0.1$. The yellow-poplar predictions were likely impacted by the inconsistencies of its presence and importance in the northern portions of this study, which tend to be transitional forests between central, Allegheny, and northern hardwoods (Braun 1950, Stout 1991). Other researchers have also experienced difficulty modeling early successional species, particularly yellow-poplar, on the Cumberland Plateau of Tennessee (Waldrop and others 1986). Attempts to decrease the rank of yellow-poplar in the iterative model revision and calibration process resulted in drastic underprediction of yellow-poplar on many sites.

Table 3.—Comparison of measured and predicted species composition at 17 paired stands in the Central Appalachians. Species composition values expressed as percentages represent mean values. P values < 0.1 indicate significant differences between measured and predicted samples as calculated by the Wilcoxon Signed Rank test.

Species Group	Measured (%)	Predicted (%)	P value
Black cherry	4	5	0.2661
Conifers	2	2	0.6250
Maples	22	22	1.0000
Midstory	11	9	0.6777
Oaks	17	13	0.4037
Other overstory	9	8	0.7119
Pioneer	22	24	0.8209
Yellow-poplar	13	17	0.0413
Total	100	100	

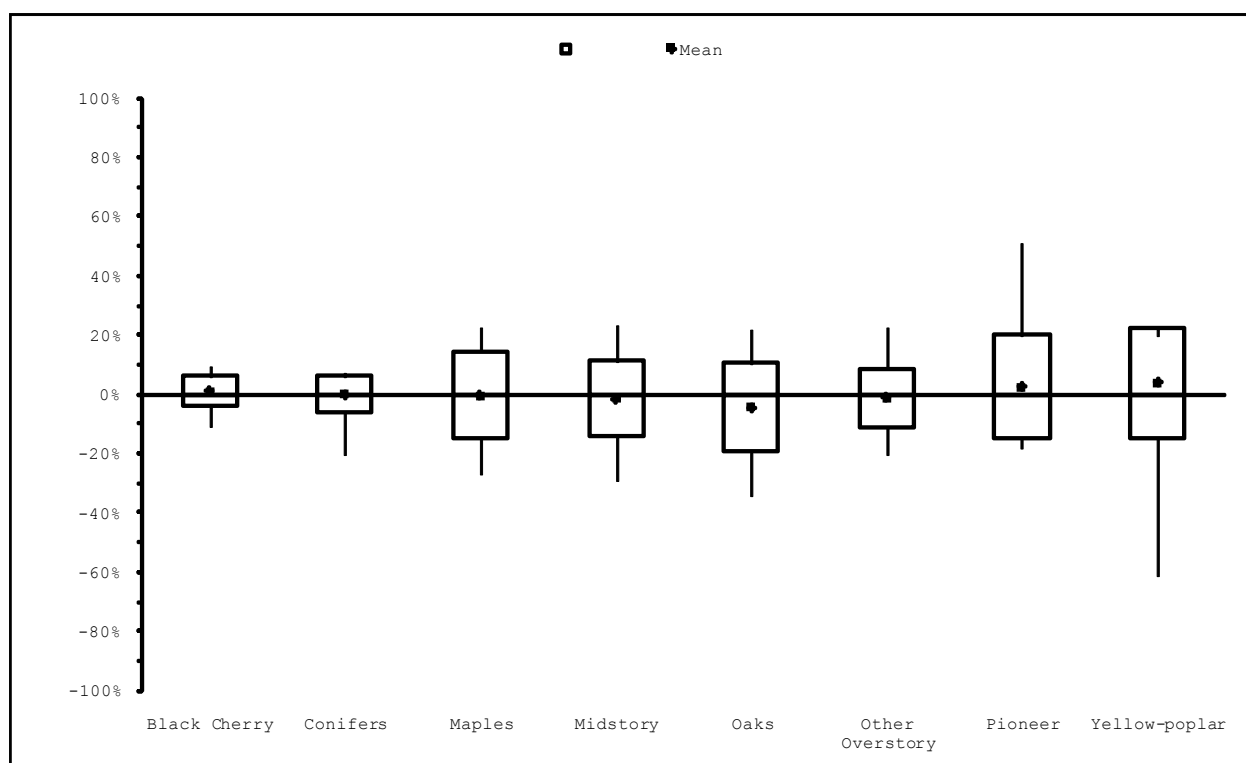


Figure 2.—Mean model error spread for each species group at 17 paired stands in the Central Appalachians. Model error was calculated as predicted – measured. Data points represent mean species group error, open bars represent mean species group error $\pm$ one standard deviation, and vertical lines represent species group error range.

Across all 17 paired stands, 1 standard deviation of the model error was not more than approximately 20 percentage points for any species group (Fig. 2). One standard deviation of model error for the pioneer and yellow-poplar species groups was only slightly more widespread than many of the other species groups, but the overall range in error for those two groups was considerably greater. These early-successional species groups were quite variable in their regeneration patterns, and as such, were difficult to parameterize. The model error for the remaining species groups was typically no more than 5 to 15 percentage points.

Results of this study indicated that the REGEN model can be adapted to predict regeneration in various communities with some success. The REGEN model showed potential to provide sufficient information to assist with a variety of management decisions. Although species groups were used in our analyses, the initial predictions in REGEN were, in fact, for individual species. The comparison of mean values between measured and predicted species composition was highly favorable (Table 3), but there was a considerable amount of variation in the degree of accuracy for a given species group on individual stands (Fig. 2). For this reason, REGEN would likely provide a more accurate portrayal of regeneration tendencies across a large ownership rather than for a single stand at this time. The potential to improve these RKBs without long-term studies conducted across the Central Appalachians, however, is limited. Given the highly stochastic nature of Appalachian hardwood regeneration, REGEN predictions should be subject to the scrutiny of an experienced forester prior to basing management decisions on model output. We nonetheless believe REGEN has the potential to be a useful tool in regeneration silviculture and should help identify stands that may be candidates for preharvest manipulation or future ameliorative treatments necessary to meet management objectives.

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EFFECTS OF PROLONGED STORAGE ON SURVIVAL AND GROWTH OF OUTPLANTED BOTTOMLAND OAKS

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Abstract.—A prominent difficulty during bottomland hardwood afforestation is that sites are often flooded during the preferred months of planting (January–March), which results in delayed planting (April–June) and reduced survival. We monitored growth and survival of oak seedlings planted in 11 different months (February through December) after varying periods of humidified cold storage to investigate the hypothesis that seedlings held over the summer months in cold storage and planted in autumn months would fare better than seedlings planted in late spring and summer. Results for Nuttall oak (*Quercus nuttallii* Palmer) generally agreed with this hypothesis, whereas results for overcup oak (*Quercus lyrata* Walt.) did not. Both species' height growth during the second growing season decreased with increased time in cold storage. These results suggest that although reduced height growth can be expected, long-term storage over the summer months and subsequent planting in autumn need not result in heavy mortality of some bottomland oak species.

INTRODUCTION

Over the course of the past century, considerable acreage of bottomland forest has been deforested and drained for row crop farming throughout the southeastern United States (MacDonald and others 1979, Turner and others 1981). Since the 1980s, natural resource professionals and federal and state agencies have focused on restoring portions of these cleared acres to native hardwood trees through various conservation programs (Stanturf and others 2001). Restoration of bottomland hardwoods has been a recent focus in the management of agricultural wetlands in Tennessee (Johnson 2007).

Professional foresters and contractors often follow conventional tree planting procedures that are well established for upland sites but pose problems in bottomlands. High water tables, overland flooding, poor soil drainage, and other seasonally exacerbated soil conditions make tree planting difficult during the commonly accepted optimum planting period between mid-winter and mid-spring (January through April). These hydrologic obstacles, instead, often cause seedlings to be planted in late spring and summer (from May on). Late planting results in poor survival. In some cases the sites may go unplanted, leading to disposal of seedlings and a follow-up attempt to plant the following year.

A previous investigation involving upland hardwood seedlings suggested that increasing the length of time in cold storage decreases post-planting root growth and percent bud break, and increases stem dieback and mortality (Englert and others 1993). We investigated whether results would be similar with bottomland

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species on a bottomland site. Our hypothesis was that seedlings held over the summer months in cold storage and planted in autumn months would fare better than seedlings planted in late spring and summer.

STUDY AREA

The study was conducted on the University of Tennessee West Tennessee Research and Education Center (WTREC), located in Jackson, TN. The site is located adjacent to the South Fork of the Forked Deer River (35°37'34" N, 88°51'22" W, 120 m mean elevation). It includes a 122-m x 92-m section nested into a larger 49-ha bottomland area that underwent afforestation in winter 2004. The predominant soil type is Waverly silt loam (0 to 2 percent slope), which is deep and poorly drained (Sease and Springer 1957). Flooding of the site occurs five to six times per year (R.M. Hayes, West Tennessee Research and Education Center, pers. comm., 2009) and inundation often lasts several days. The site was used for row crop farming until 2004 when it was enrolled in the Conservation Reserve Program (CRP).

METHODS

Nuttall oak (*Quercus texana* Buckley, NTO) and overcup oak (*Quercus lyrata* Walt., OCO) were planted in this study. NTO and OCO were selected because both species were previously found to tolerate extended inundation on the site available for this study (McCurry and others 2006). All seedlings planted were 1-0 stock and were grown at the Tennessee Department of Agriculture Forestry Division East Tennessee Nursery in Delano, TN. Initial height measurements were taken to the nearest cm using a custom-made PVC pipe calibrated in cm. The average aboveground height at the time of planting, measured from ground to terminal bud, was 51.3 cm for NTO and 104.1 cm for OCO. Initial stem caliper was also measured with a Plasti-cal Digital Caliper (Mitutoyo America Corp., Aurora, IL) to the nearest 1.0 mm at ground level. The average caliper for NTO was 7.9 mm, and for OCO, 10.8 mm. Prior to shipping from the nursery, the roots were dipped in Viterra root dip (potassium propenoate propenamide copolymers, Amereq, Inc., New City, NY) to conserve moisture. The Viterra root dip was mixed at a rate of 14.2 gm per gallon of water. After dipping, they were then packaged (without mulch) into bundles of 25 seedlings. After delivery, the seedlings were stored in a humidified cold room at 2.2 °C, with 94-percent relative humidity. An unforeseen, 30-hour power outage occurred on August 24-25, 2007. The maximum temperature in the cold room during the outage reached 25.2 °C, with an average of 19.6 °C. The relative humidity dropped to a low of 81.9 percent.

The study was established as a randomized complete block design with all treatments appearing once in each of three blocks established in relation to elevation of the site. Twelve treatments, corresponding to plantings in every month of the year, were assigned at random to 12 plots within each block, which resulted in 36 plots for the entire study (Fig. 1). One row containing 20 NTO seedlings on 1-m spacing and a second row containing 20 OCO seedlings on 1-m spacing were established in each plot. With a total of three replications, 60 NTO and 60 OCO seedlings were planted per month. Seedlings for each month were planted successively between 7 and 10 a.m. and between the 10th and 20th day of any given month. No seedlings could be planted during the month of January at the outset of the study in 2007 because the study site was flooded. As a result, the January treatment was dropped from the study.

Site preparation, conducted in August 2006, consisted of a single application of a 2 percent solution of Roundup (glyphosate, Monsanto, St. Louis, MO) in 76-cm bands applied directly over the designated rows. In addition, during the year of planting (2007) and the two following years, seedlings were side-dressed with

the same herbicide at the same rate, once per month (April through September). The band width was 38 cm on both sides of each row. Weeds were controlled carefully throughout the entire study to minimize effects of differences in the abundance of competing vegetation during the study. Mowing between the rows occurred each month during the growing seasons. Survival and seedling heights were recorded in September 2008 and again in 2009.

Data were analyzed through one-way ANOVA with models appropriate for a randomized complete block design. Pairwise comparisons were conducted between months with Tukey's honestly significant difference ($\alpha = 0.05$). All analyses were conducted using SAS, Version 9.9 (SAS Institute, Cary, NC, 2008).

RESULTS

As of September 2009, mean survival calculated across all treatments and sample periods was 78.3 percent for NTO and 32.0 percent for OCO. As expected, survival for both of the species was favorable in the Feb.-Apr. planting treatments; NTO averaged 90.6 percent and OCO, 78.3 percent. Survival for both species was less favorable during the late spring and early summer (May-Aug.); NTO averaged 62.1 percent and OCO, 29.2 percent. There were no differences in mean NTO survival between planting dates in this period, but mean survival for OCO was lower during some late spring and summer planting months. The results for the final planting period (Sept.-Dec.) were promising for NTO. Mean Sept.-Dec. NTO survival averaged 85.4 percent, and was not different in any of these months from survival in February, March, or April. In contrast, OCO survival during all months within this period was zero (Fig. 2). Seedlings were considered dead if there was no indication of living tissue above ground. Scratch testing to reveal green cambium was conducted on questionable seedlings.

Mean seedling height for both NTO and OCO decreased from early to late planting dates (Fig. 3). Mean heights (as of September 2009) for both species were greatest for seedlings planted in the early months of Feb.-Apr.; NTO averaged 173.0 cm and OCO, 164.0 cm. Heights for both species were less for most months within the late spring and summer planting period (May-Aug.) than in the Feb.-Apr. period (Fig. 3); NTO averaged 114.9 cm and OCO, 94.5 cm in the May-Aug planting period. NTO height averaged 76.7 cm in the final period (Sept.-Dec.), and heights were less in this period than in the Feb-Apr period (Fig. 3). No OCO seedlings survived in the final period. When we observed height measurements for NTO during the project, Feb.-Apr. planted seedlings were 131.9 percent taller than Sept.-Dec. planted seedlings in 2008 and 125.6 percent taller than Sept.-Dec. planted seedlings in 2009. Although data on resprouting were not formally collected, resprouting of late planted seedlings was observed to be more prevalent.

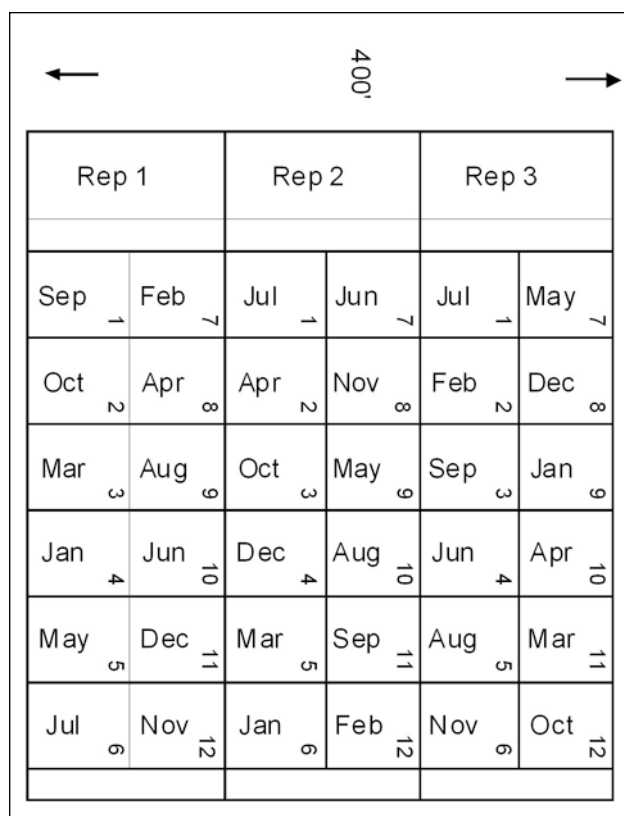


Figure 1.—Study design.

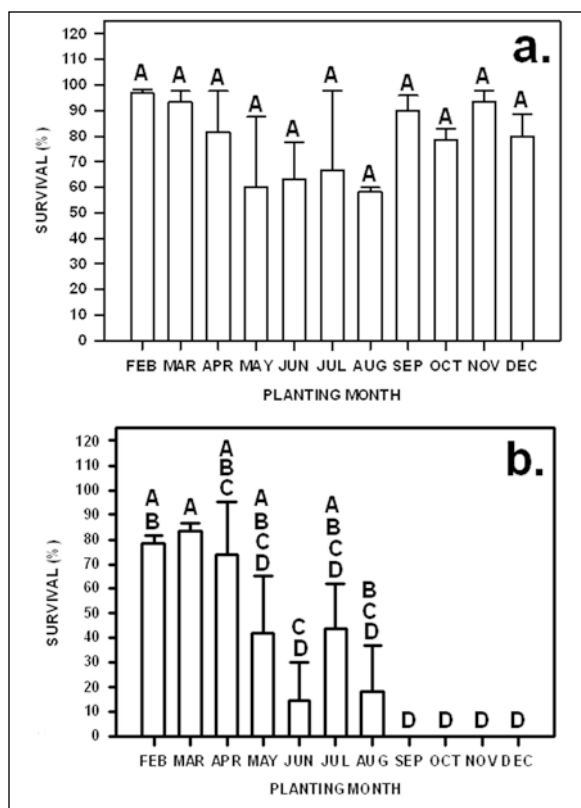


Figure 2.—Mean percent survival of Nuttall oak (a.) and overcup oak (b.) as of September 2009 by planting month treatment. Bars with the same letters are not significantly different at the $\alpha = 0.05$ level. Error bars represent 1 standard error.

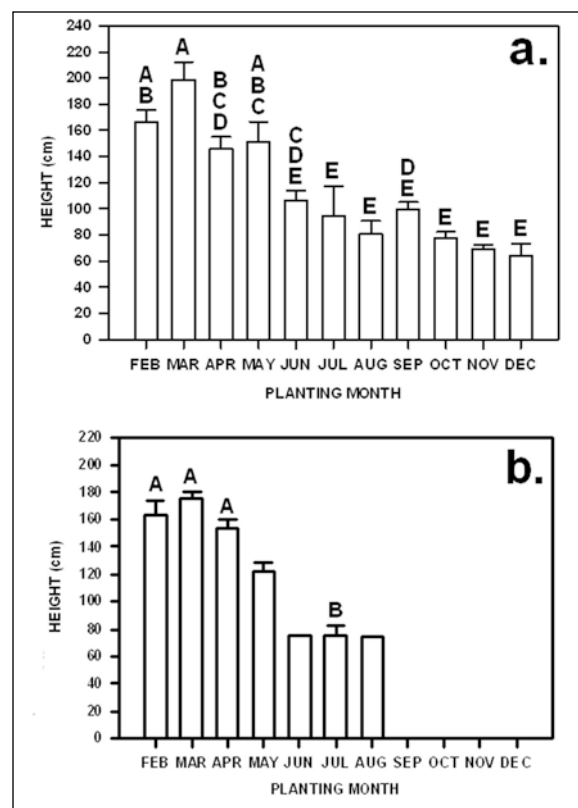


Figure 3.—Mean height of Nuttall oak (a.) and overcup oak (b.) as of September 2009 by planting month treatment. Bars with the same letters are not significantly different at the $\alpha=0.05$ level. Error bars represent 1 standard error.

DISCUSSION

Due to limitations in the number of species studied, the types of stock examined, and other factors, this study is not a definitive test of the hypothesis that holding seedlings in a humidified cold room over the summer months, and then planting them during the autumn months, is a viable solution to the problem of early-season flooding of bottomland restoration sites. The findings do suggest, however, that in the case of NTO, it is at least possible to have acceptable survival rates (80 percent or better) with seedlings planted in September-December. It can be argued that the very different results obtained for OCO may be the result of innate differences in the two oak species. These differences could include differences in respiration rate, desiccation resistance, or other species-specific characteristics. For instance, the two species were shown to differ in carbohydrate changes in response to flooding (McCurry 2006). It is also possible that differences in initial seedling size between the two species influenced the results.

At the time of planting, the initial seedling size and caliper measurements were noticeably larger for the OCO than for NTO. At 104.1 cm (initially), OCO were 103 percent taller than the NTO. Similarly, OCO, with an initial stem caliper of 10.8 mm, were 37 percent larger in caliper than NTO. It is well documented that seedlings with a larger stem caliper experience more favorable survival than those with a smaller caliper (Weigel 1999). Why this was not the case in our project is not clear.

The potential effects of two occurrences during the study should be noted. First, during the year of implementation (2007), the west Tennessee region experienced an extreme drought. Eight months received below normal precipitation in 2007, and year-end total precipitation was 33.4 cm below normal. During the growing season months of May-August, the precipitation deficit was 31.3 cm (NOAA 2009). This drought could have substantially increased mortality overall, particularly in the summer months. Secondly, the power outage that occurred for 30 hours in August that allowed the temperature in the cold room to climb to 25.2 °C may have influenced seedling viability. Since none of the OCO planted in September-December (after this power outage) survived, it is possible to speculate that this occurrence may have had a greater effect on OCO than NTO, although the reasons for this disparity are unclear. If these events had not taken place during the study, survival could have been greater for all planting dates in the study.

Results suggest that, at least in the first two growing seasons, height growth is suppressed by delayed planting. The average height of autumn-planted NTO seedlings, when measured 2 years after establishment (September 2009), was considerably less than the average height of those planted in the spring. The percent difference in height had declined slightly from 2008 to 2009, and if this trend continues, early height differences could become less substantial over the duration of the rotation. Initial height can be important, however, in influencing the competitive status of planted seedlings relative to other vegetation.

The promising results obtained for NTO in this study suggest that additional research involving the performance of delayed plantings of other species used in bottomland hardwood restoration is warranted. We are planning to examine the viability of a range of seedling size classes for an expanded set of species stored for various periods of time in a cold room.

ACKNOWLEDGMENTS

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REDUCED RATES OF CONTROLLED-RELEASE FERTILIZER LOWER POTENTIAL NITROGEN LEACHING FROM A WISCONSIN BARE-ROOT TREE NURSERY

Ryosuke Fujinuma, Nick J. Balster, and Hyung-Kyung Lee¹

Abstract.—Controlled-release fertilizer (CRF) typically increases nitrogen (N) fertilizer uptake and lowers N lost from the rooting zone via leaching. However, questions remain as to whether lower rates of CRF could further increase this efficiency, especially in sandy bare-root nurseries in Wisconsin. We hypothesized that: 1) a reduced CRF application at 60 percent of the conventional rate (CRF60) would yield similar seedling morphology and biomass compared to a conventional application (CONV) and CRF at 100 percent of the conventional rate (CRF100); and 2) the CRF60 would result in significantly reduced potential N leaching (total N inputs minus plant N uptake) relative to the conventional rate. To test these hypotheses, we conducted a 2-year experiment in northern red oak (*Quercus rubra* L.) (NRO) at the largest bare-root nursery in Wisconsin. We applied a total of 259 kg N ha⁻¹ to the CONV (ammonium sulfate) and CRF100 (coated urea) and 157 kg N ha⁻¹ to CRF60 (coated urea) using standard application schedules. After two growing seasons, there was no significant difference in morphology (height, diameter, first-order lateral roots), biomass (stem, roots, total), and N content (stem, root) between treatments. However, the potential N leaching decreased significantly in the CRF treatments (30 percent under CRF100, 88 percent under CRF60) compared to CONV. Net N mineralization also decreased significantly under CRF (59 percent under CRF100, 32 percent under CRF60) compared to CONV. Overall, we conclude the CRF60 treatment would maintain seedling production goals for NRO relative to conventional levels but would significantly reduce potential N leaching at a bare-root nursery in Wisconsin.

INTRODUCTION

Producing large numbers of high-quality northern red oak (*Quercus rubra* L.) (NRO) seedlings in nursery soils characterized by rapid drainage and low nutrient retention requires large annual inputs of nitrogen (N) fertilizer. This condition is exemplified in Wisconsin state nurseries, which produce more than 475,000 NRO seedlings (bare-root) annually on coarse sandy soils (Wisconsin State Nursery 2008). Thus, nursery managers work to maximize N fertilizer uptake and minimize potential N leaching to the environment when N supply exceeds seedling demand (Paramasivam and Alva 1997, Landis and van Steenis 2004, U.S. Environmental Protection Agency 2006). Therefore, the need for efficient N management within these systems is critical to meet production goals while reducing the economic and environmental costs of excess N (Vande Hey 2007).

Controlled-release fertilizer (CRF), a type of enhanced-efficiency fertilizer, typically increases plant N uptake by matching N-release patterns from CRF prills (plastic-coated synthetic granules) with plant N demand

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(Shoji and others 1991). This synchronized N-release from CRF has been shown to be effective for seedling production in sandy nursery soils, as CRF increases the residence time of fertilizer-N in the soil relative to conventional fertilization (Dobrahner and others 2007, Zotarelli and others 2008). For example, in sandy arable fields, potatoes and citrus trees grown under CRF management showed larger N uptake compared with the plants grown with conventional fertilization (Dou and Alva 1998, Alva and others 2003, Zvomuya and others 2003). Consequently, increasing plant N uptake with CRF could reduce potential N leaching and thus provide an advantageous management option for nursery managers seeking to increase N fertilizer use efficiency.

Despite the benefits of CRF to nursery management, finding the most efficient CRF fertilization regime continues to challenge nursery managers in Wisconsin. Although improved modeling has been shown to provide more accurate estimates of N release from CRF (Fujinuma and others 2009), questions remain as to whether reduced rates of CRF will satisfy morphological grading criteria such as stem height, root collar diameter, and number of first-order lateral roots (FOLR) (Stroempl 1985, Clark and others 2000). In a study with white pine (*Pinus strobus*) seedlings, reduced CRF applications resulted in N uptake and morphology similar to those of a conventional N fertilization regime (Dobrahner and others 2007). However, few studies have examined similar reductions of CRF in deciduous tree seedlings such as NRO.

The objectives of this study were to determine whether a reduced rate of CRF application would maintain NRO production criteria and minimize potential N leaching within a bare-root tree nursery in southwest Wisconsin. We hypothesized that: 1) reducing the CRF application to 60 percent of the current conventional rate (CRF60) would yield seedling morphology and biomass similar to those with a conventional (CONV) application and CRF at 100 percent of conventional rates (CRF100); and 2) the CRF60 application would result in less potential N leaching than with the CONV and CRF100 applications. To test these hypotheses, we quantified changes in seedling morphology, biomass allocation, N uptake, and potential N leaching in 2-year-old (2-0) NRO seedlings. All hypotheses were tested relative to the end of a typical 2-year production cycle for NRO in Wisconsin.

MATERIALS AND METHODS

This study was conducted at H.G. Wilson State Nursery in Boscobel, WI (43°13' N, 90°70' W). A typical growing season for Wilson Nursery extends from May to October with the 30-year mean air temperature of 18 °C and cumulative precipitation of 487 mm (National Oceanic and Atmospheric Administration [NOAA] 2009). The soil was classified as a loamy sand Entic Hapludoll (U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NCRS] 2008) (90 percent sand) that was slightly acidic (pH 5.9-6.1). NRO seedlings are grown over a 2-year production cycle to maximize root biomass and caliper for increased survivability when transplanted into field conditions (Vande Hey 2007). Prior to this study, the nursery received an organic matter amendment in fall 2004 to increase soil organic matter content to approximately 2 percent; no additional organic matter was applied over the course of this study. The nursery used irrigation water from the nearby Wisconsin River as needed throughout the 2005 and 2006 growing seasons observed in this study.

This study used a complete randomized design of three fertilizer treatments. Each treatment was replicated three times, where each replicate was established as a randomly selected row (1.2 m by 167 m) within the nursery; each row consisted of 46 contiguous beds (1.2 m by 3.6 m). In an effort to capture the spatial

variability within each row, we first excluded three beds from each end (to avoid edge effects) and then randomly selected three beds (subplot) from the remaining 40 beds to represent the average of each row (replicated). This design yielded a sample size of nine subplots (beds) per treatment. To avoid cross contamination from adjacent rows, seedlings within a 15-cm side buffer were excluded from sampling. Seedlings were planted in fall 2004 from acorns collected from Hayward, WI, yielding a seedling density of approximately 86 seedlings per square meter.

The three fertilizer treatments were as follows: a conventional fertilization (CONV) of 259 kg N ha⁻¹ (approximately 300 mg N seedling⁻¹), CRF100 (N application rate equivalent to that of the CONV), and CRF60 (157 kg N ha⁻¹, 60 percent of the CONV N-application rate). Each treatment was applied to the entire row. Ammonium sulfate (21-0-0, 24 percent sulfur) was used in the CONV treatment. The CRF consisted of coated urea (Multicote[®], 42-0-0, Haifa Chemical Ltd., Haifa Bay, Israel), which was rated to release 80 percent of fertilizer N within 2 months under constant temperature conditions of 27 °C. Applications of ammonium sulfate were evenly divided 11 times over the course of the 2-year production cycle; each individual fertilizer application amounted to 23.5 kg N ha⁻¹. Eight fertilizer applications were applied at 10-day intervals during the first growing season (2005; June to August) and another three applications at 30-day intervals during the second growing season (2006; May to July). The CRF treatments were divided into two applications; 73 percent of the total application was applied in May 2005 and 27 percent of the total application was applied in May 2006. Thus the seedlings under the CRF100 received 188 kg N ha⁻¹ in 2005 and 71 kg N ha⁻¹ in 2006; the seedlings under the CRF60 received 113 kg N ha⁻¹ in 2005 and 43 kg N ha⁻¹ in 2006.

Seedling morphology and biomass were sampled at 2-week intervals from June through October in both the 2005 and 2006 growing seasons. Three seedlings were randomly collected from each subplot using a destructive method. Stem height and root-collar diameter of each seedling were measured on site. In the laboratory, the number of FOLR were counted and then the seedlings were partitioned into leaves, stems, and roots, dried at a constant temperature of 65 °C for 10 days, and weighed (Wilde and others 1979). Each component (leaves, stems, roots) was ground to pass through a 250-µm mesh sieve and homogenized for further chemical analysis. Total N concentration of each component was analyzed using a combustion method with auto-analyzer (Flash EA 1112 CN analyzer, CE Elantech, NJ).

Precipitation plus irrigation water (ml) was summed monthly during the 2005 and 2006 growing seasons (June-September); precipitation and irrigation waters were not measured separately in this study. This volume of water was recorded with a tipping bucket connected to a data logger (HOBO Data Logging Rain Gauge, Onset Computer Corporation, Bourne, MA), which was positioned 90 cm from the ground surface. Three additional funnels (20-cm diameter) were randomly placed at 90 cm from the ground surface within the study area to collect liquid samples for chemical analysis. Each funnel was connected to a 3.6-L reservoir, which contained a drop of 40 g L⁻¹ sodium bromate to suppress microbial activity in the sample. Solution volumes in the reservoir were recorded at 2-week intervals during the 2005 and 2006 growing seasons, and then subsamples were taken for further chemical analyses. Water loss due to evaporation was minimized by using containers painted with a reflective coating and looping the input tube from the funnel. The concentration of inorganic-N (ammonium-N and nitrate-N) in each sub-sample was analyzed using colorimetric methods (4D10a1a for ammonium-N and 4D9a1a for nitrate-N) (USDA-NRCS 2004) with a Lachat QC-8100 auto-analyzer (Lachat Instruments Inc., Loveland, CO).

Net N mineralization rate was estimated using an open-top *in-situ* soil core method described in Brye and others (2002). Two PVC tubes (20 cm long by 5 cm in diameter) were placed in randomly selected beds within each replicate (row). Each PVC tube was first inserted into the soil and then removed along with the soil contained within each core. Bags of cation and anion exchange resins (10 g of Dowex 1X8-100 and 10 g of Dowex 50W8-100) were then stuffed into the bottom of each PVC tube followed by a bag of silica sand (20 g) to deter the adsorption of N from the soil below the tube. The tubes were then reinserted into their respective holes. The resin bags within each PVC tube were replaced with new bags monthly to avoid saturation. To maintain the integrity of the hole, the PVC tubes were carefully removed and reinserted when the soil was moist. The inorganic N stored in the resin bags was extracted using 40 ml of 2-M potassium chloride. The concentration of inorganic N in the extracts was also analyzed using the aforementioned colorimetric methods.

The N retained by woody biomass (U; mg N seedling⁻¹) was estimated as N in the stem and roots at the end of the second growing season (after leaf drop) plus the N retained in leaf litterfall in 2005 and 2006:

$$U = [SN + RN] + [(LN1 + LN2) \times 0.4] \quad [1]$$

where SN is stem N content (mg N seedling⁻¹), RN is root N content (mg N seedling⁻¹), LN1 is leaf N content (g seedling⁻¹) at the end of August 2005, and LN2 is leaf N content (g seedling⁻¹) at the end of August 2006, with 40 percent retained in the litterfall based on published literature for oak species (Aerts 1996, Kolb and Evans 2002).

The potential N leaching (PNL) was estimated by a N mass-balance equation;

$$PNL = WN + F + M - U \times 0.86 \quad [2]$$

where PNL is an estimate of the potential N leaching (kg N ha⁻¹) summed over two growing seasons, WN is the N content of water input (precipitation and irrigation, kg N ha⁻¹), F is the fertilizer-N (kg N ha⁻¹), M is net N mineralization (kg N ha⁻¹), U is the N uptake from Eq.[1], and 0.86 is a parameter to convert units from mg N per seedling to kg N ha⁻¹ given the 86 seedlings m⁻² of seedling density.

Differences in seedling morphology (stem height, root-collar diameter, FOLR) and biomass (stem biomass, root biomass) were evaluated by treatment means per sampling date (every 2 weeks) and at the end of the 2-year production cycle. The N balance in NRO seedlings was also calculated at the end of the 2-year production cycle. The data were analyzed by one-way analysis of variance (ANOVA) using a completely randomized design to evaluate the treatment effects on seedling morphology, biomass and N allocation, net N mineralization, and potential N leaching. Significant differences in means comparisons were identified by using Fisher's least significant difference (LSD) test at $p < 0.05$. All statistical analyses were conducted with JMP 7.0 (SAS Institute Inc., Cary, NC).

RESULTS AND DISCUSSION

In support of our first hypothesis, seedling morphology and biomass of NRO seedlings did not differ by fertilization treatment (Table 1). All fertilization treatments yielded seedlings within acceptable morphological grading criteria (35-50 cm height; >6 mm root-collar diameter; >8 FOLR) for 2-0 NRO in the central United States (Stroempl 1985, Clark and others 2000). In addition, all fertilization treatments yielded a similar range in total seedling biomass, between 41-46 g at the end of the 2-year production cycle ($p = 0.470$) (Table 1). Many studies also report similar stem height, root-collar diameter, and total biomass between

Table 1.—Morphology and biomass of northern red oak seedlings at the end of the 2-year production cycle in Wilson State Nursery, Boscobel, WI. Values are given as means plus one standard error. There was no significant difference between treatments in Fisher's LSD test at $p < 0.05$.

Fertilizer treatment	Stem height (cm)	Root collar diameter (mm)	Number of FOLR [†]	Stem biomass (g)	Root biomass (g)	Leaf biomass (g) [‡]	Total biomass (g)
CONV	57.5 (3.2)	7.4 (0.3)	16 (1)	8.57 (0.68)	13.60 (2.15)	11.48 (1.36)	43.69 (2.45)
CRF100	50.2 (3.6)	6.8 (0.3)	14 (1)	7.11 (0.66)	12.67 (1.32)	8.73 (0.51)	41.05 (1.20)
CRF60	46.3 (5.1)	6.4 (0.7)	16 (2)	6.11 (1.24)	12.68 (2.81)	10.22 (1.00)	45.98 (3.73)

[†]FOLR = first-order lateral root with diameter greater than 1 mm.

[‡]Leaf biomass = sum of leaf biomass at summer 2005 and 2006.

seedlings under CRF applications compared to conventional fertilization (Guertal 2000, Boman and Battikhi 2007, Dobrahner and others 2007). By showing a 60-percent reduction in the rate of CRF does not change the morphology or biomass of the NRO seedlings compared to a CRF100 or conventional treatment, our results suggest that the conventional fertilizer rates over the 2-year management cycle likely exceeded production requirements.

The temporal change in height, diameter, and biomass followed similar trends regardless of the fertilization treatments over the 2-year production cycle (Fig. 1). During the first growing season, stem heights increased from June through August, in contrast to root-collar diameters, which increased threefold from mid-August until the end of the 2005 growing season. This pattern was consistent with general growth curves reported for seedlings in bare-root nurseries (Landis and van Steenis 2004). By the end of July 2006 (year 2), NRO seedlings met the morphological grading criteria previously described (Stroempl 1985), which have been used as a reliable indicator of relative growth potential and outplanting success (Von Althen 1980, Grossnickle 2005). Despite the minimal change in height and diameter growth during the second growing season, the belowground biomass continued to increase regardless of treatment during both growing seasons. Thus, a CRF treatment at 60 percent of the conventional rate also did not alter the temporal dynamics of NRO morphology or biomass allocation.

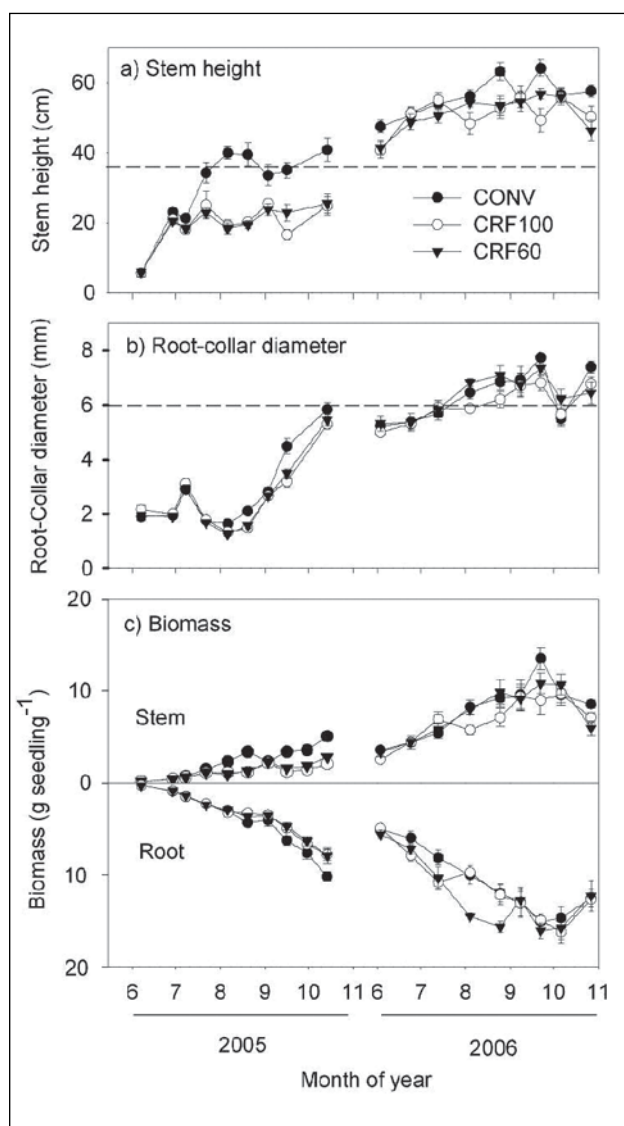


Figure 1.—Stem height, root-collar diameter, and biomass (stem, root) of northern red oak seedlings in Wilson State Nursery, Boscobel, WI, during a 2-year production cycle. The dashed horizontal lines on the graphs of stem height (a) and root-collar diameter (b) refer to morphological grading requirements (Stroempl 1985). Each point indicates a mean plus one standard error.

Similar to the lack of treatment effect on temporal growth dynamics, there was no significant differences in stem and root biomass between treatments, $p = 0.232$ and $p = 0.942$, respectively (Table 1). However, biomass was preferentially allocated to roots relative to stems throughout this study (Table 1; Fig. 1), which concurred with findings from other studies of NRO seedlings grown under a range of N fertilization rates (Salifu and Jacobs 2006, Salifu and others 2009). This pattern of allocation was also observed in N content, where approximately twice the N was allocated to root tissue relative to stems regardless of fertilization treatment (Table 2); similar results have been found in recent studies of NRO seedlings in bare-root nurseries (Salifu and others 2009, Fujinuma and Balster *in review*). Therefore, the reduced amount of N fertilization does not appear to change the preferential allocation of biomass in the roots of NRO seedlings relative to stems within this nursery. Because field trials have shown a competitive advantage of larger seedlings (Grossnickle 2005), we speculate that these larger root systems with proportionally high N content may increase the survival and subsequent growth after transplant into competitive growth environments.

Both CRF treatments significantly decreased potential N leaching compared to the CONV treatment, supporting the second hypothesis in this study ($p < 0.05$). Under the same rate of fertilizer-N, CRF100 decreased potential N leaching by 30 percent compared to CONV. However, CRF60 further decreased potential N leaching by 88 percent relative to CONV (Fig. 2). The contribution of a significantly lower net mineralization and lower fertilizer N inputs from CRF likely explains this additional reduction in potential

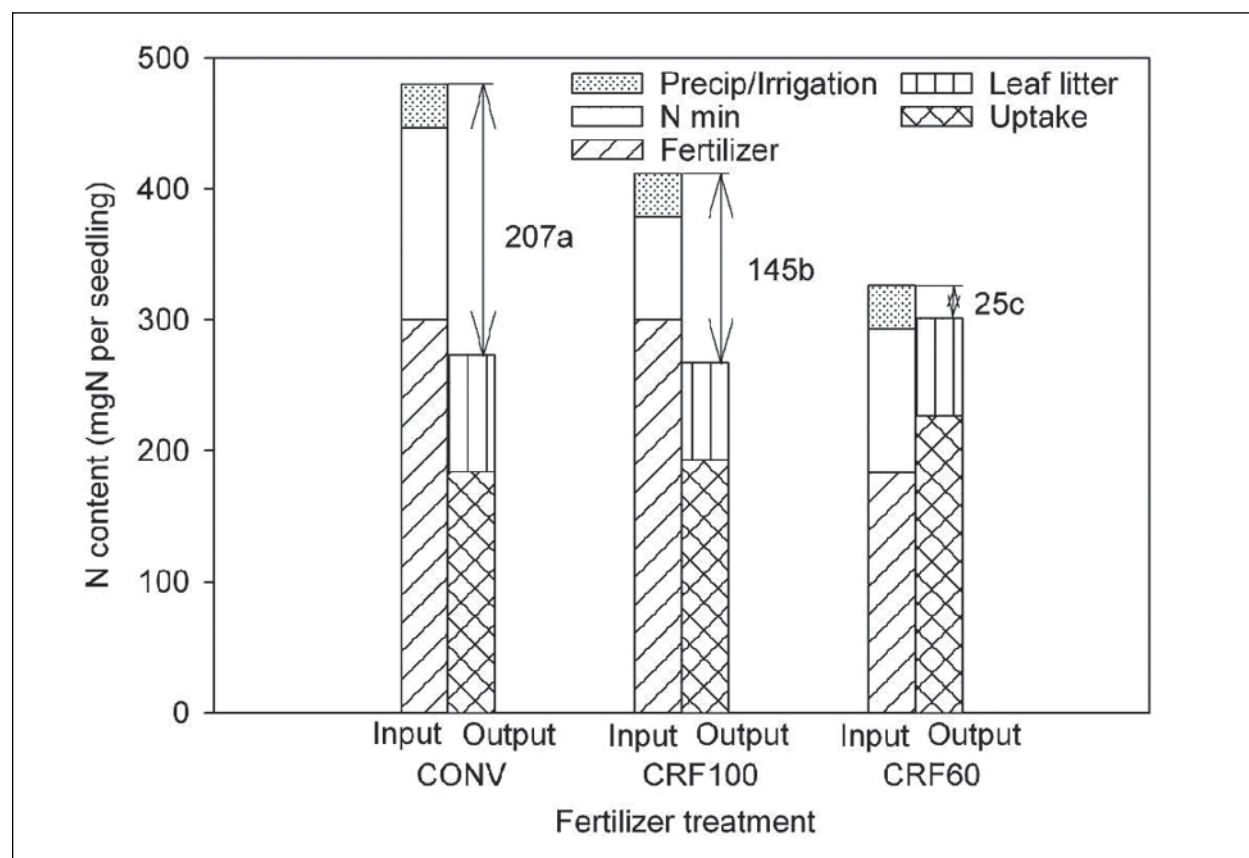


Figure 2.—Potential nitrogen leaching, total inputs, and outputs of N in Wilson State Nursery in Boscobel, WI, over a 2-year production cycle of northern red oak seedlings. Inputs represent total N contributions from fertilization, mineralization, precipitation, and irrigation water. Nitrogen outputs represent seedling N (sum of root N and stem N), as well as leaf litter (maximum leaf N content multiplied by mean retranslocation, 40 percent). Significant differences ($p < 0.05$) are represented by different letters.

N leaching under CRF60 (Köhler and others 2006, Li and others 2007). Although a reduction in N leaching from CRF applications is not a new discovery (Shoji and Kanno 1994, Guillard and Kopp 2004), our results suggest that reductions in the rate of CRF can effect even greater reductions in potential N leaching from the rooting zone without sacrificing production goals in this Wisconsin bareroot nursery.

Differences in fertilization regime appear to change the rate of net N mineralization. In this study, net N mineralization in the CONV treatment at the end of the 2-year production cycle was significantly greater relative to both CRF treatments ($p < 0.05$) (Fig. 2). Results vary by year, however. Net N mineralization under the CRF treatments averaged 34-54 percent lower relative to the CONV treatment in the 2005 growing season, but was not significantly different between treatments in 2006 ($p = 0.39$) (Table 2). Although this lack of significance during the second growing season was not large enough to nullify a difference over the 2-year production cycle, it could be explained by the fact that plots received no organic matter amendments during the study period. Alternatively, the reduced input of N from the CRF treatments may have increased the soil carbon:nitrogen ratio, effectively lowering the contribution of N from the turnover of soil microorganisms (Li and others 2002, McAndrew and Malhi 2002, Chu and others 2005). It should be noted that the N collected in the resin beads may have included the input of soluble N from the fertilizer prills, as N mineralization was measured *in situ* over the course of this study. Based on literature reports of typical N mineralization rates (Köhler and others 2006), however, this artifact of the experiment was likely negligible.

Finally, given the sandy texture of the nursery soil examined in this study, the timing of N leaching loss (and thus fertilization) may be of greatest importance when considering the type of fertilizer used. In other words, despite the higher potential N leaching we measured from the conventional treatments, this leachate may not have reached the water table during the 2005 and 2006 growing seasons within this nursery. By first approximation, the field capacity of a loamy sand soil is approximately $0.13 \text{ m}^3 \text{ m}^{-3}$ (Campbell and Norman 1998). Therefore, given a dry soil, in order for fertilizer N to reach soil water pools at a 100-cm depth (Dobrahner and others 2007), input from precipitation and irrigation must exceed 130 mm. During this study, monthly cumulative precipitation and irrigation ranged from 100 to 250 mm during the 2005 growing season and 100 to 150 mm during the 2006 growing season (Fig. 3), and there were no heavy storm events ($>100 \text{ mm hr}^{-1}$). If we subtract the average in cumulative monthly potential evapotranspiration (148 mm) over the two growing seasons examined in this study (Bland 2009), N leached from the rooting zone (15-20 cm) likely did not reach a meter into this soil profile from June through September in 2005 and 2006. Thus, any excess N stored in the upper soil profile may be susceptible to greater seasonal loss during early spring with the relatively large input of melt water from winter snowmelt in Wisconsin; the 30-year

Table 2.—Nitrogen content in northern red oak seedlings, nitrogen input from fertilizer, net N mineralization, and irrigation/precipitation in Wilson State Nursery, Boscobel, WI. Values are given as means plus one standard error. Different letters within a column indicate significant differences in Fisher's LSD test at $p < 0.05$.

Fertilizer treatment	Stem N (mg N)	Root N (mg N)	Maximum Leaf N (mg N)		Fert. N (kg ha ⁻¹) 2005-2006	Net N mineralization (kg ha ⁻¹)		N in (P+I) (kg N ha ⁻¹)	
			2005	2006		2005	2006	2005	2006
CONV	63a (7)	121a (12)	106a (10)	116a (5)	259	68a (19)	30a (8)	13	15
CRF100	63a (5)	130a (11)	55b (4)	131a (5)	259	23b (8)	17a (3)	13	15
CRF60	72a (8)	155a (17)	56b (7)	129a (23)	157	37ab (8)	30a (14)	13	15

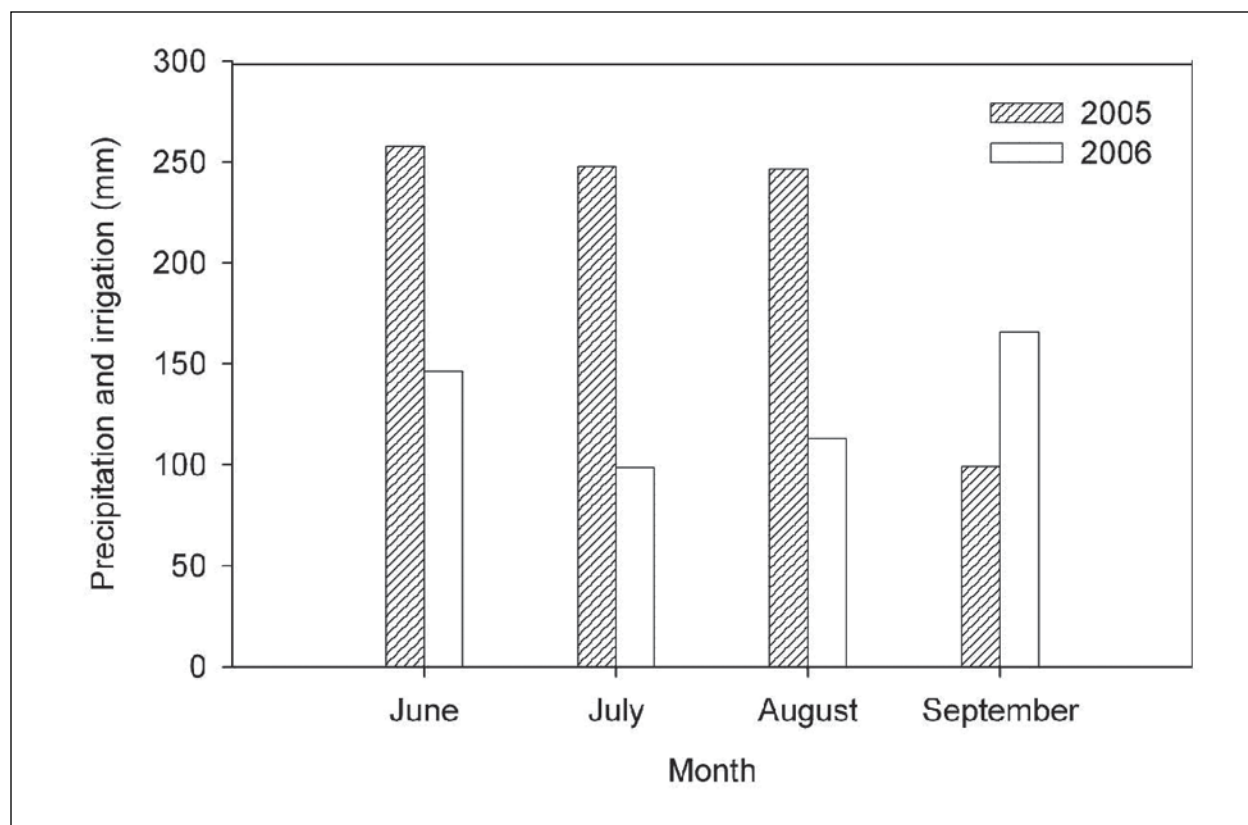


Figure 3.—Sum of monthly precipitation and irrigation at Wilson State Nursery, Boscobel, WI, during 2005 and 2006 growing seasons.

average cumulative precipitation from November to April is 254 mm at this site (NOAA 2009). Given these estimations, we hypothesize that the impact of excess N fertilization on deep groundwater pools within this nursery may be underestimated by measurements that do not quantify soil water dynamics over the entire growing season. Direct measurements of these fluxes, including the capture of soil leachate, would test this hypothesis.

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WHITE OAK SEED SOURCE PERFORMANCE ACROSS MULTIPLE SITES IN INDIANA THROUGH AGE 16

Philip A. O'Connor and Mark V. Coggeshall¹

Abstract.—In 1984, a series of combined provenance/progeny tests was established by the Indiana Department of Natural Resources, Division of Forestry. Three plantings were established using a maximum of 70 open-pollinated families representing 17 natural stands (15 Indiana, 1 Illinois, and 1 Missouri). Height data were collected periodically through age 16. Early analyses (ages 3 and 5) showed demonstrable but statistically insignificant variation based on stand level means, while 10- to 25-percent increases in height could be realized by selecting at the family level and a 30-percent increase could be achieved by selecting the best individuals within families. Similar increases in height growth were noted for both the 10th- and 16th-year measurements. Potentially, a 42-percent increase in height growth could be obtained by selecting the best performing individuals from the best performing families at age 16.

INTRODUCTION

White oak (*Quercus alba* L.) is one of the most commercially important hardwoods in the North Central United States (Rink 1980). As of 2005, the most recent year for which such data have been tabulated, annual harvest of white oak in Indiana approached 45.6 million board feet for use as sawlogs, veneer, and staves (Piva and Gallion 2007). According to the *Hardwood Leader* (December 2009), white oak export volume has continued to increase at a rate of 8 to 12 percent during the preceding quarter. Natural regeneration of all oaks is insufficient to replace stocking of these species, so the majority of reforestation relies on planted nursery seedlings. To address this need, nursery sales of white oak have averaged 292,000 seedlings annually over the past 5 years in Indiana, enough to plant almost 500 acres each year. With this increased demand for regeneration seedlings comes an awareness of the use of well adapted seed sources when new plantings are being established across the state. Growers should consider the significance of provenance since the supply of acorns from any one stand can be irregular (Hubert 2005).

Within their native ranges, oaks vary considerably in growth rate, leaf color, habit, and stature, as well as in ease of transplanting and ability to tolerate variable soil moisture and pH (Bassuk 2001). Trees within a species can exhibit tremendous variation in seed size and appearance. Large acorns will normally produce larger and more vigorous seedlings in the nursery (Coggeshall 1995, Purohit and others 2003). Such maternal effect does not last for the life of the tree, but rather is lost several years after field establishment (Coggeshall 1995). Seed sources, however, can have a great impact on the long-term productivity of white oak plantings (O'Connor and Beineke 2004).

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Several provenance/progeny tests already exist for other oak species (Rink 1980). However, very limited information has been made available for use in assessing the long-term growth rates and other phenotypic traits in white oak in the midwestern United States. To address this need, a series of provenance/progeny test plantings was established in 1984 to quantify variations within these traits for white oak planted in Indiana.

METHODS

Acorns were collected in fall 1982 from one to five parent trees in each of 15 white oak stands throughout Indiana, as well as one stand each from Illinois and Missouri (Table 1). All parent trees exhibited at least average vigor and form when compared with adjacent trees within the same stand. There was no specified minimum distance between parent trees within a stand. All acorns were sown at the Indiana Department of Natural Resources, Vallonia State Nursery, in late October 1982. In fall 1983 1-0 seedlings were lifted and held over winter in refrigerated storage in accordance with standard hardwood nursery procedures. All study seedlings were out-planted in spring 1984 at three locations: Jasper-Pulaski State Nursery, Starve Hollow State Recreation Area, and Harrison-Crawford State Forest (Table 2).

Table 1.—Collection sites for white oak acorns used to produce provenance/progeny test seedlings.

State	Location	N Latitude	W Longitude	Provenance ID	Families per Stand
Illinois	Boskeydell	37° 44'	89° 13'	IL-01	5
Indiana	Vallonia	38° 49'	86° 05'	IN-01	5
	Lake Monroe	39° 08'	86° 27'	IN-02	5
	Shakamak	39° 10'	87° 14'	IN-03	1
	North Vernon	39° 00'	85° 38'	IN-04	1
	Martin SF	38° 42'	86° 45'	IN-05	5
	JP F&W Area	41° 05'	86° 55'	IN-06	5
	Laporte	41° 37'	85° 45'	IN-07	5
	South Bend	41° 40'	86° 12'	IN-08	5
	Rochester	41° 04'	86° 14'	IN-09	5
	Lafayette	40° 25'	86° 53'	IN-10	5
	Marion	40° 33'	85° 40'	IN-11	5
	Connersville	39° 39'	85° 08'	IN-12	1
	Wyandotte Cave	38° 15'	86° 18'	IN-13	5
	Evansville	37° 58'	87° 35'	IN-14	5
	Huffman	38° 07'	86° 45'	IN-15	3
Missouri	Rolla	37° 58'	89° 30'	MO-03	4

Table 2.—Outplanting locations and number of included families for 1983 white oak provenance/progeny test plantings in Indiana.

Provenance Test Site	Number of Families	Location	Latitude, Longitude	Soil Type
Harrison-Crawford State Forest	57	Corydon, IN	38° 15' N, 86° 15' W	Haymond silt loam
Starve Hollow State Recreation Area	63	Vallonia, IN	38° 49' N, 86° 05' W	Bloomfield sandy loam
Jasper-Pulaski Nursery	53	Medaryville, IN	41° 09' N, 86° 54' W	Maumee sandy loam

Site preparation at each location employed the broadcast of an herbicide mixture (glyphosate at 0.6 gallons/acre and simazine at 4.5 pounds/acre) applied prior to planting. Weed control was then strip-applied annually through crown closure utilizing the same mixture.

The outplantings consisted of four-tree open-pollinated family plots arranged in a randomized complete block design. For each family, six blocks were established at Starve Hollow as four-tree squares, with 4 feet between trees in the family plot, and 8 feet between plots. The plantings at Jasper-Pulaski and Harrison-Crawford were installed in four-tree row plots with 8-foot spacing within and between rows. Six blocks were established at Jasper-Pulaski and eight blocks were established at Harrison-Crawford. Due to seed availability and seedling survival, not all sources are represented at all three test locations. The Starve Hollow planting was subsequently thinned from four tree family plots to the single best tree in each family plot at age 7. The other planting sites remain unthinned.

Total height data were collected on all live trees in each planting during the 1999/2000 dormant season. To minimize the confounding effects of various conditions at each planting site (i.e., thinning at Starve Hollow, depredation by white-tail deer [*Odocoileus virginiana* Zimmerman] at Jasper-Pulaski), the plot means were presented as percentages of each overall plantation mean.

Data were analyzed using basic statistical functions available with Microsoft Excel Analysis ToolPak™ (Microsoft Corporation, Redmond, WA). Separate analyses of variance were conducted at the family and stand (provenance) level. The software utilized did not allow for including interaction between family and stand in the analysis. When the null hypothesis was rejected, the mean separations were determined using least significant difference (LSD) analysis.

RESULTS

Overall plantation survival at Harrison-Crawford, Starve Hollow, and Jasper-Pulaski was 84.9, 81.6, and 48.0 percent respectively. Most of the mortality was evident prior to the fifth-year measurements.

Growth of the trees at Starve Hollow generally exceeded twice that of either of the other planting locations. Stand-level (provenance) variation was not significant for measured heights at any of the planting locations. In contrast, family-level (progeny) variance was significant for 16-year heights across all three plantings ($p < 0.05$). Figures 1, 2, and 3 show the families whose heights exceeded the plantation mean by at least one standard deviation at Harrison-Crawford, Starve Hollow, and Jasper-Pulaski respectively. Family mean separations were determined by LSD.

Comparisons with 10-year height data (data not shown) suggest 82-percent parity at the level of mean plus one standard deviation for families in the Harrison-Crawford planting. By this we mean that 82 percent of the families exceeding the plantation mean by at least one standard deviation at 16 years had also done so at 10 years of age. Only 50- and 46-percent parity was demonstrated for those in the Jasper-Pulaski and Starve Hollow plantings, respectively.

Family IN 10-04 from Lafayette performed well across all three planting sites. In contrast, IN 07-04 was a high performer at both the northern site (Jasper-Pulaski) and the southern site (Harrison-Crawford) but was only average at the Starve Hollow planting.

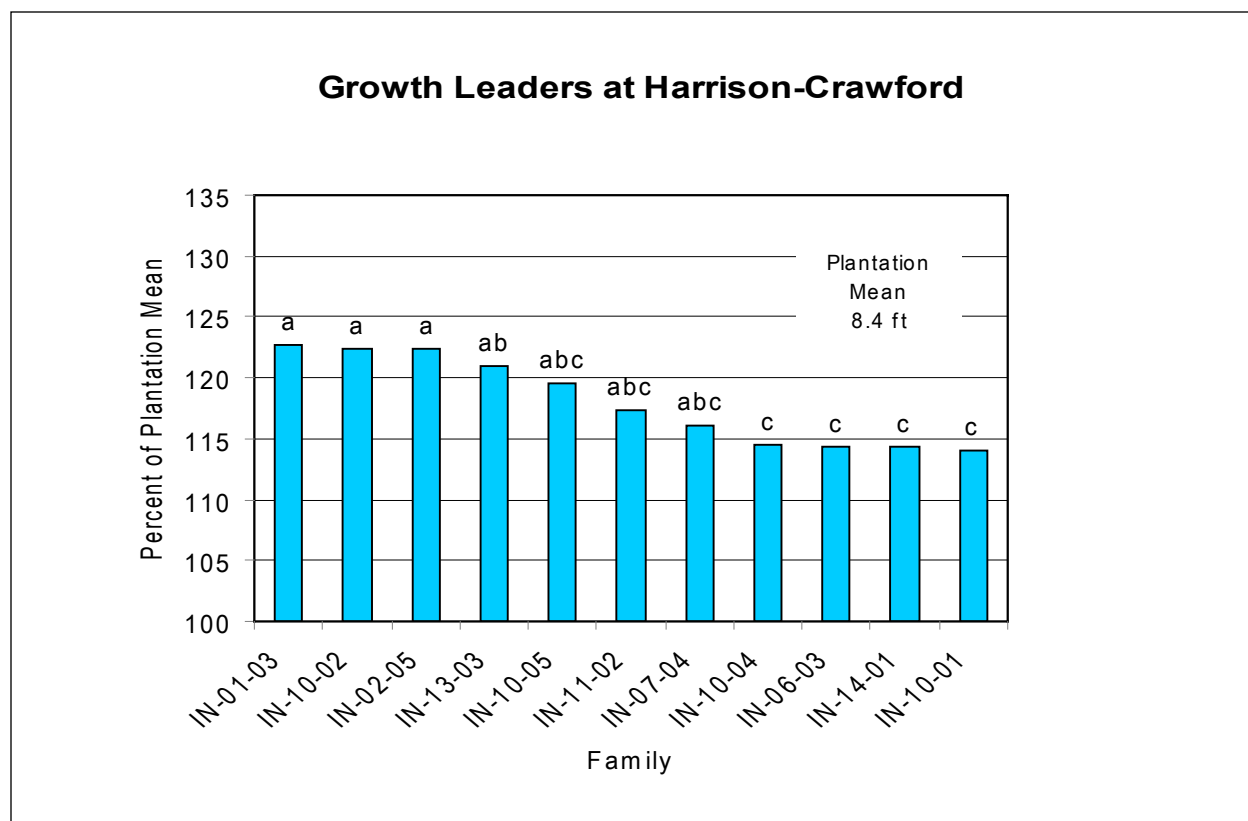


Figure 1.—Families whose 16-year mean heights exceeded the plantation mean by at least 1 standard deviation at Harrison-Crawford.

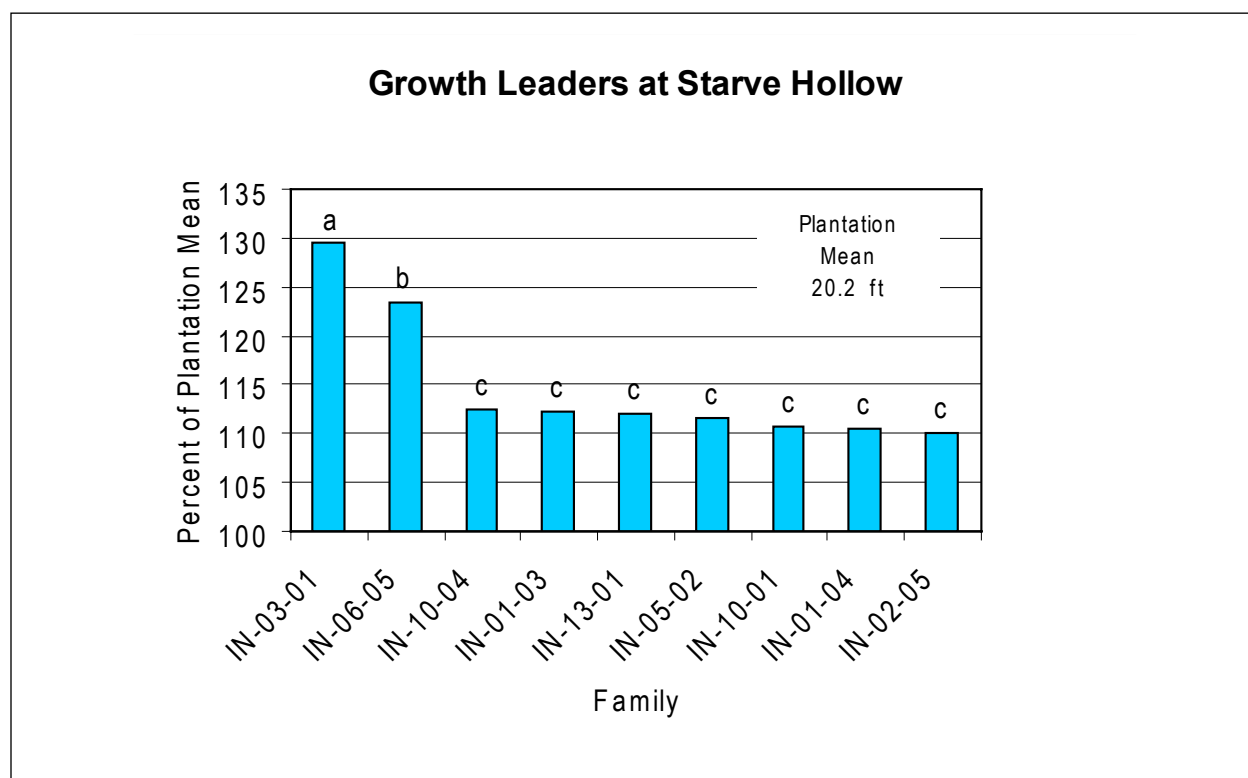


Figure 2.—Families whose mean 16-year heights exceeded the plantation mean by at least 1 standard deviation at Starve Hollow.

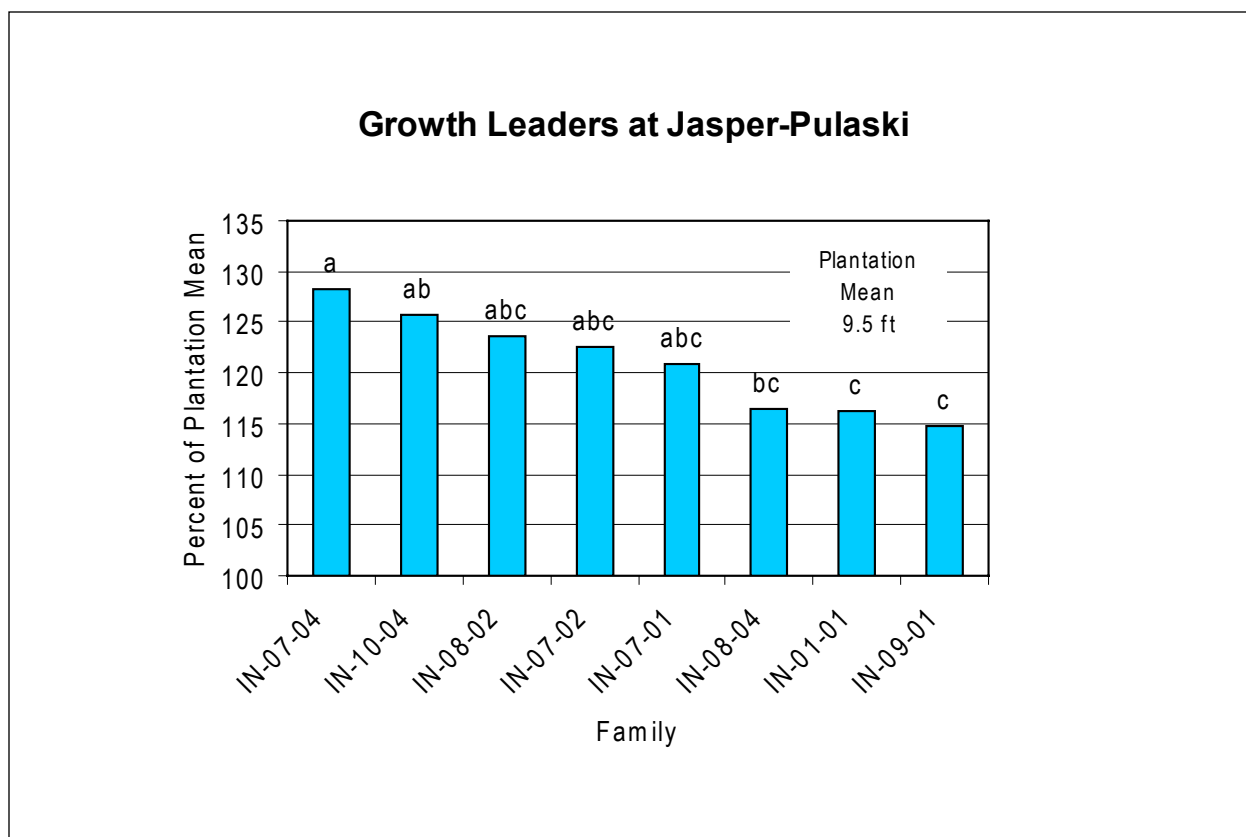


Figure 3.—Families whose 16-year mean heights exceeded the plantation mean by at least 1 standard deviation at Jasper-Pulaski.

The best-performing families were: IN 01-03 at Harrison-Crawford with an average height of 10.3 feet, which represents the potential for a 22.8-percent increase in height growth; IN 03-01 at Starve Hollow with an average height of 26.2 feet, representing a potential 29.6-percent increase, and IN 07-04 at Jasper-Pulaski with an average height of 12.3 feet, representing a potential height increase of 28.3 percent after 15 growing seasons.

Within the fastest-growing families, individual trees exhibited even greater improvement in height growth. For example, the best-performing tree within IN 14-04 at Harrison-Crawford grew at 209.7 percent of the plantation mean. At Starve Hollow, the best tree within the best family (IN 03-01) grew 142.2 percent of the plantation mean. Similarly, for IN 10-04 at Jasper-Pulaski the height gain was 182.7 percent of the plantation mean.

DISCUSSION

The higher growth rate at Starve Hollow was the result of several factors. The 1990 thinning, which eliminated all but the best trees in each family plot, increased the plantation mean by 22.5 percent and extended the free-to-grow period prior to crown closure in contrast to the other two study sites. In addition, the close proximity of this planting site to the Vallonia Nursery facilitated timely and regular weed control during the study period.

Ten-year height growth appears to be a good indicator of sixteenth-year height growth at Harrison-Crawford, but not at the other two planting sites. The Harrison-Crawford planting has not been thinned and is assumed to be fully stocked with an average tree height of 8.4 feet. At Starve Hollow, the lower parity may be due in great part to the effect of the 1990 thinning, which retained only the best trees for each family plot. Similarly, the Jasper-Pulaski site suffered high mortality associated with predation by white-tailed deer, leaving a number of large, irregular canopy gaps that likely favored the growth of some trees in some family plots at random.

While the influence of stand origin was not significant in this study, it is suggested that increases in 15-year height growth of planted white oak seedlings will be obtained by utilizing seeds from specific family origins. For example, family IN 10-04 and IN 07-04 were among the best performers in both northern and southern Indiana (Figs. 1, 3) and should be considered as superior seed sources for future regeneration plantings throughout the state. It is recommended that a combination of those high-performing seed sources from the families shown in Figures 1, 2, and 3 will give a higher likelihood of successful plantings with potential for improved growth rates over the long run.

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HYDROLOGY AND HARVESTING

EVALUATING BEST MANAGEMENT PRACTICES FOR EPHEMERAL CHANNEL PROTECTION FOLLOWING FOREST HARVEST IN THE CUMBERLAND PLATEAU – PRELIMINARY FINDINGS

Emma L. Witt, Christopher D. Barton, Jeffrey W. Stringer, Daniel W. Bowker, and Randall K. Kolka¹

Abstract.—Most states in the United States have established forestry best management practices to protect water quality and maintain aquatic habitat in streams. However, guidelines are generally focused on minimizing impacts to perennial streams. Ephemeral channels (or streams), which function as important delivery systems for carbon, nutrients, and sediment to perennial streams, are comparatively unprotected. An examination of the effectiveness of three types of streamside management zones around ephemeral channels is under way at the University of Kentucky's Robinson Forest, located in southeastern Kentucky. Treatments include: 1) no equipment limitation with complete overstory removal and unimproved crossings; 2) no equipment limitation with retention of channel bank trees and improved crossings; and 3) equipment restrictions within 7.6 m of the channel with retention of channel bank trees and improved crossings. The following improved crossing types were studied: wooden portable skidder bridges, steel pipe/culverts, and PVC pipe bundles. Water samples were taken during storm flows using automated water samplers and were analyzed for total suspended solids and turbidity. Initial results indicated that harvest operations resulted in increased sediment movement in ephemeral channels over unharvested controls. All improved channel crossings reduced the amount of sediment input over that of an unimproved ford. However, the 7.6-m equipment restriction zone did not provide additional sediment reductions.

INTRODUCTION

Forestry best management practices (BMPs) have been established in most states to protect water quality and aquatic habitat during forest harvests. Frequently, these guidelines vary based on the stream type impacted by the harvesting operation; ephemeral channels receive the least protection.

Ephemeral channels are distinguished from intermittent and perennial streams based on hydrology. Kentucky's Forestry Best Management Practices describe ephemeral channels as those that flow during or directly after precipitation or in response to snow melt and conduct surface water directly or indirectly to perennial streams (Stringer and Perkins 2006). Perennial streams flow continuously except in extreme drought, and intermittent streams flow during the wet season (Fritz and others 2008). Ephemeral channels commonly lack defined banks and generally have large amounts of organic matter in the channel bed (Hansen 2001). Svec and others

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(2005) used channel geometry and watershed characteristics to distinguish among the three stream types in eastern Kentucky.

Ephemeral channels are an important component of the headwater stream system, which encompasses first- to third-order streams draining areas of less than 2 km², including perennial, intermittent, and ephemeral flow regimes (Horton 1945, Adams and Spotilia 2005). A majority of a watershed's stream length is located in the headwater system. Studies have found that the headwaters can encompass from 60 percent to 80 percent or more of the entire watershed network (Gomi and others 2002, Benda 2005, Wipfli and others 2007). Headwater streams are able to effectively deliver water, fine sediment, and fine particulate organic matter downstream, as well as store coarse sediment and large woody debris (MacDonald and Coe 2007, Wipfli and others 2007). Additionally, ephemeral channels, as a component of the overall headwater system, provide important habitat to a variety of biota (Meyer and others 2007).

During harvesting operations, ephemeral channels are often crossed by driving directly through the channel rather than crossing with an improved or elevated crossing. This procedure is common because many ephemeral channels are relatively small and running water is frequently absent at the time of harvest. However, these channel crossings can lead to increased sediment movement from the ephemeral channels to downstream reaches (Davies and Nelson 1993). Improved crossings potentially could decrease the amount of sediment delivered downstream.

The objective of this study was to determine how alternative forestry BMPs influence ephemeral channel suspended solids and turbidity. We assessed the following three BMPs: 1) no equipment limitation with complete overstory removal and unimproved crossings, 2) no equipment limitation with retention of channel bank trees and improved crossings, and 3) equipment restrictions within 7.6 m of the channel with retention of channel bank trees and improved crossings. In addition, we compared several improved stream crossing techniques to unimproved crossings and unharvested conditions. These treatments were developed in part to help provide information on the effectiveness of Kentucky's Best Management Practices for Water Quality Management (Stringer and Perkins 2001).

STUDY AREA

Robinson Forest is located in the Cumberland Plateau region of southeastern Kentucky (Fig. 1). Six harvested watersheds (two per BMP treatment) and two unharvested control watersheds were used in the study. Topographically, Robinson Forest is characterized by steep slopes with well drained residuum or colluvial soils formed from sandstone, shale, and siltstone. The forest is classified as mixed-mesophytic forest with oak (*Quercus* spp.), hickory (*Carya* spp.), and yellow-poplar (*Liriodendron tulipifera*) as dominant overstory species. Mean annual precipitation at Robinson Forest is 117.5 cm (Cherry 2006).

Channel slope for the monitored ephemeral channels ranged from 7 to 26 percent with a mean slope of 19 percent. The slope of the skid trails on either side of the crossings ranged from 1 to 23 percent with a mean of 9 percent. The height of the trail relative to the channel ranged from 0 to 2.7 meters, with a mean of 0.97 meters.

Skid trails were constructed along the contour at various intervals from the top to the bottom of the slopes. The majority of channel crossings occurred with either cable or grapple skidders (John Deere models 540

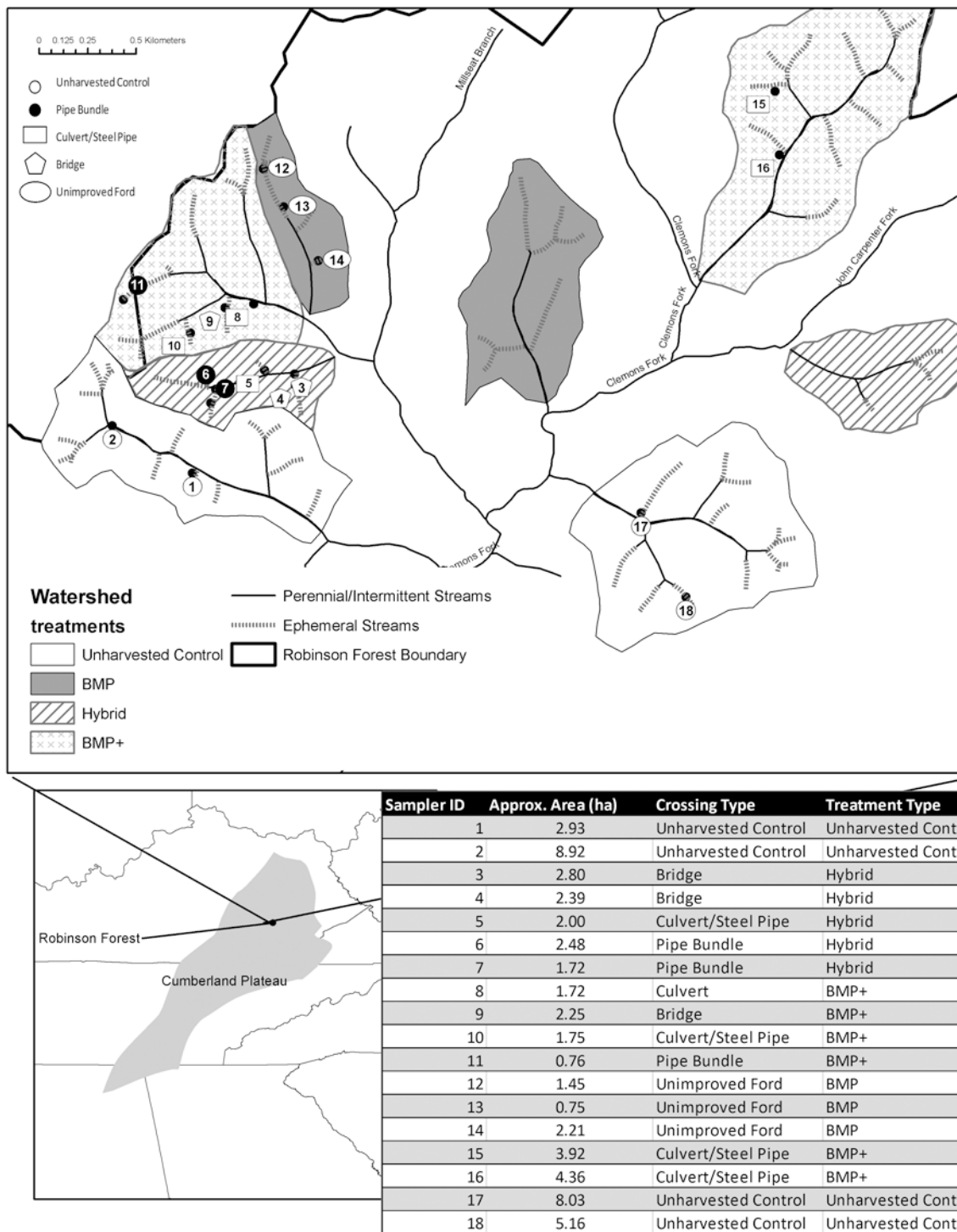


Figure 1.—Location of Robinson Forest and treatment watersheds.

and 648 and Caterpillar models 525 and 545). Additional equipment used during the harvest that crossed ephemeral channels included: Timbco swing-armed feller bunchers (model 445 or 445EXL) and John Deere bulldozers (model 650, 700, or 800). The feller bunchers and bulldozers ran on tracks; the skidders used air-filled rubber tires.

METHODS

TREATMENTS

Three watershed treatments were applied to two watersheds each (Fig. 1). The current regulations in Kentucky for ephemeral channels require that channels be crossed with improved crossings where feasible. Where improved crossings are not feasible, channels should be crossed at right angles with fords. Retention of overstory trees and equipment limitations in or directly adjacent to the channel are not required. Blockage of ephemeral channels with logging debris (soil, root wads, tree tops, or tree sections) is not permitted by Kentucky's mandatory BMP law. This treatment is referred to as the "BMP treatment." The second treatment, referred to as a "Hybrid treatment," requires the use of improved channel crossings, as well as retention of the nearest overstory tree along the channel on both banks (tree stringer). The final treatment is the "BMP-plus treatment," which also requires improved crossings and retention of a tree stringer along both banks, as well as a 7.6-m zone from the channel in which equipment is not permitted (equipment limitation) (Table 1).

The improved crossings tested were portable wooden skidder bridges, steel pipes/culverts, and PVC pipe bundles. Portable wooden skidder bridges consisting of three panels were installed and removed over channels using wheeled grapple skidders. The steel pipe/culvert treatment used either corrugated steel or solid steel pipe placed into channels and backfilled with at least 10 cm of soil. PVC pipe bundles (Mason and Moll 1995) were constructed according to Blinn and others (1998). The PVC pipe bundles were constructed of at least twenty 9-cm diameter PVC pipes threaded together with steel cable (Reeves and others 2008). Pipe bundles were laid in the channel and allowed to conform to the channel bottom, covered with a layer of geo-textile fabric, and overlaid by at least 20 cm of soil. The purpose of the overlain soil was to minimize damage to the pipe bundle during crossing by wheeled skidders, both loaded and unloaded, as part of normal harvesting operations. On average, skid trail crossings were in use for a limited time period, generally 2 to 6 weeks. Crossings were removed after the skid trail was retired. During retirement, structures were removed and in the case of the culvert/pipe and PVC pipe bundle, the majority of the overlain sediment was removed from within the channel using a bulldozer.

Table 1.—Ephemeral channel best management practices applied during treatment. The current Kentucky regulations are implemented in the "BMP" treatment. A "stringer" refers to retention of the nearest overstory tree to the channel along both banks. In addition to the three treatments, ephemeral channels in unharvested areas are monitored as a control treatment.

Treatment	Sreamside management zone width (m)	Overstory retention	Crossing type
BMP	0	0%	Unimproved
Hybrid	0	Stringer	Improved
BMP+	7.6	Stringer	Improved

SAMPLE COLLECTION AND ANALYSIS

Sixteen ephemeral channels in six watersheds were monitored from July 2008 to November 2009. In two of the 16 channels, multiple channel crossings were monitored using two samplers, for a total of 18 sampling points. Estimated catchment areas of the channels measured from the sampling point ranged from 0.75 ha to 8.9 ha, with a mean area of 3.1 ha (Fig. 1, estimated from topographic maps and geographic information systems data).

Measureable precipitation was recorded on approximately 40 percent of the days in the period sampled. Of the 84 total events during the sampling period, 35 (42 percent) had total precipitation of greater than 12.7 mm and 21 (25 percent) had total precipitation of greater than 25.4 mm. The total number of sampled events was 25, with a mean precipitation of 44 mm (range: 9 mm to 240 mm). Partial events (events that did not activate all samplers) were included in the analyses due to differences in sampler deployment dates, precipitation gradients, channel flow, and equipment functioning that made obtaining a complete sample set from a specific event difficult.

ISCO automated pump samplers (Teledyne ISCO, Lincoln, NE) equipped with liquid level actuators were used to collect samples following storm events that resulted in ephemeral channel flow. Actuators were placed directly in the dry channel bed to activate the sampler when flow began in response to a precipitation event. Events that were not of sufficient duration or intensity to result in channel flow were not sampled. One 200-ml sample was collected every 30 minutes beginning at the start of a flow event and ending 24 hours later, resulting in a 9.4-L composite sample. A composite sampling method was chosen due to the time associated with analyzing multiple samples for each event and sampling location. Of the composite sample, a 1.5-L sub-sample was used for analysis.

Samples were analyzed for two parameters: total suspended solids and turbidity. Total suspended solids was determined gravimetrically using a 0.45- μ m filter according to American Public Health Association guidelines (Greenberg and others 1992). Turbidity (measured in formazin turbidity units [FTU]) was analyzed using a Hanna portable turbidity meter (model HI 93703, Hanna Instruments, Woonsocket, RI).

Harvesting began in June 2008 and was completed in October 2009. Because monitoring of each channel began as soon as possible following the start of its use as a crossing site, the number of samples for each crossing type and treatment varied.

Data were grouped according to treatment and crossing type. Of the 18 crossings, three were crossed with pipe bundles, five with culverts, and three with bridges; three were forded without improved crossings, and four remained unharvested. Three channels were harvested using the BMP treatment, three used the Hybrid treatment, and six used the BMP-plus treatment. Statistical differences were determined using a protected least significant differences test (one-way ANOVA followed by nonparametric Mann-Whitney tests) (SAS Institute Inc., Cary, NC). Percentage comparisons (increase or decrease) were calculated from mean total suspended solids and turbidity measurements from the ephemeral sampling points.

RESULTS

Preliminary data indicate a pronounced effect on total suspended solids and turbidity when improved crossings are used (Fig. 2). When bridges were used, total suspended solids was reduced by 88 percent

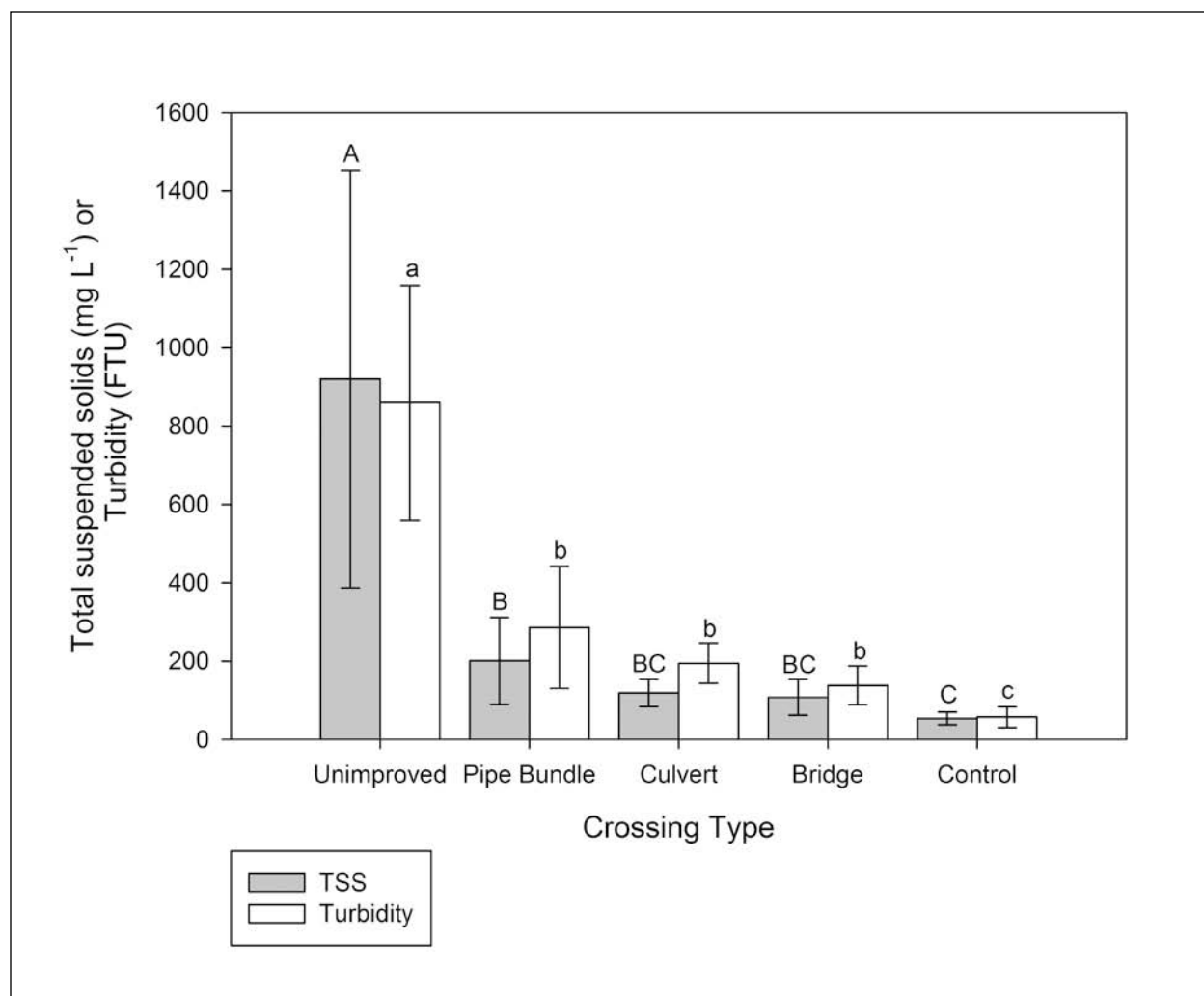


Figure 2.—Effect of crossing type on total suspended sediment and turbidity. Letters indicate significant differences at the $\alpha = 0.05$ level; capital letters indicate differences in total suspended sediment and lowercase letters indicate differences in turbidity. Formazin turbidity units (FTU) were used to measure turbidity. “Control” refers to data from unharvested ephemeral channels.

compared to unimproved crossings ($p = 0.003$, $n = 26$). Culverts resulted in an 87-percent reduction in total suspended solids ($p = 0.002$, $n = 39$), and pipe bundles reduced total suspended solids concentrations by 78 percent ($p = 0.01$, $n = 24$) compared to unimproved crossings. Similar reductions in turbidity were measured, with bridges resulting in decreases of 84 percent compared to unimproved crossings ($p = 0.004$, $n = 25$), culverts reducing turbidity by 77 percent ($p = 0.008$, $n = 37$), and pipe bundles reducing turbidity by 67 percent ($p = 0.02$, $n = 22$). While all improved crossings showed a significant reduction in suspended solids compared to fords, there was no statistical difference among improved crossings.

Among the improved crossing types, bridges resulted in the smallest increase in suspended solids (100 percent) compared to unharvested channels ($p = 0.11$, $n = 53$). Suspended solids in channels using culvert/pipes was 120 percent higher than in unharvested channels ($p = 0.08$, $n = 66$) and in those using PVC pipe bundles was significantly higher (275 percent) than in unharvested channels ($p = 0.004$, $n = 51$). Significant increases in turbidity were measured between the unharvested control and bridge (140-percent increase; $p = 0.0001$, $n = 46$), unharvested control and culvert (240-percent increase; $p < 0.0001$, $n = 58$), and unharvested control and pipe bundle (400-percent increase; $p = 0.0001$, $n = 43$).

Differences were also measured in total suspended solids ($p < 0.0001$, $n = 107$) and turbidity ($p < 0.0001$, $n = 97$) when the cumulative treatment impacts were considered (Fig. 3). In the BMP treatment, total suspended solids was 500 percent higher than in the Hybrid treatment ($p = 0.004$, $n = 34$) and was 655 percent higher than in the BMP-plus treatment ($p = 0.001$, $n = 47$). In addition, significant increases in turbidity were measured between the BMP and Hybrid treatments (260-percent increase; $p = 0.007$, $n = 31$), as well as between the BMP and BMP-plus treatments (390-percent increase, $p = 0.004$, $n = 43$).

Compared to the unharvested control, total suspended solids concentrations in the Hybrid treatment were 185 percent higher ($p = 0.006$, $n = 61$) and turbidity was 320 percent higher ($p < 0.0001$, $n = 52$). Total suspended solids concentrations in the BMP-plus treatment were nearly 130 percent higher than in the unharvested control ($p = 0.047$, $n = 74$), with a 210-percent increase in turbidity ($p < 0.0001$, $n = 56$).

Total suspended solids concentrations at the ephemeral channel crossings were 17 times higher in the BMP treatment compared to the unharvested control ($p = 0.0001$, $n = 45$). Turbidity was 15 times higher in the BMP treatment compared to the unharvested control ($p = 0.0001$, $n = 37$).

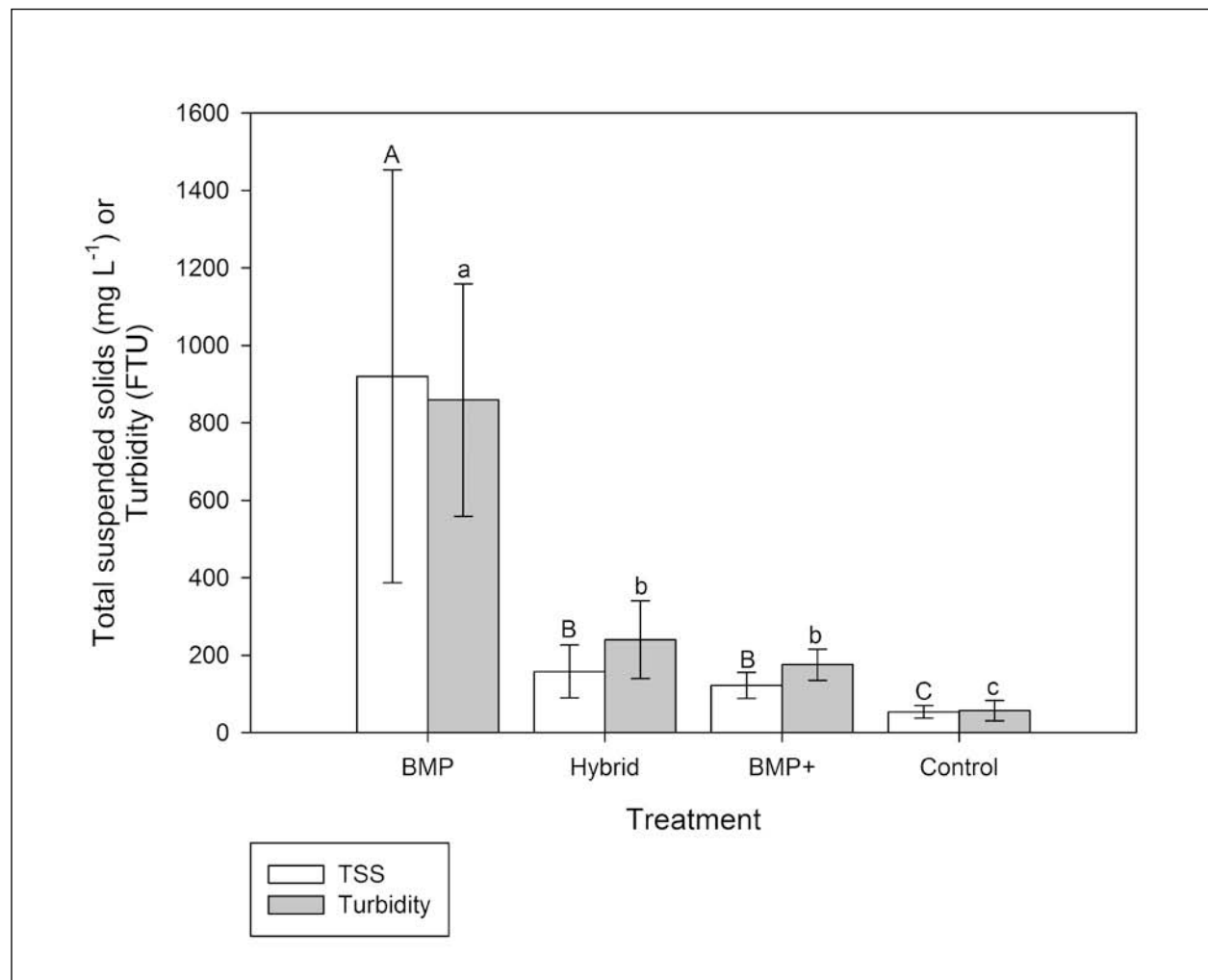


Figure 3.—Effect of treatment on total suspended sediment and turbidity. Letters indicate significant differences at the $\alpha = 0.05$ level; capital letters indicate differences in total suspended sediment and lowercase letters indicate differences in turbidity. “Control” refers to data from unharvested ephemeral channels.

DISCUSSION

A prolonged drought in summer and fall 2008 during harvest operations may have influenced differences among the crossing types. Due to the lack of precipitation and subsequent flow in ephemeral channels, all of the samples were taken after crossings were retired. Therefore, data from this study measured the impacts of harvesting and improved crossing use on ephemeral channel sediment dynamics following retirement of crossings, rather than determining the ability of improved crossings to protect channels during harvesting activities.

Comparisons among the crossing types showed that any improved crossing type decreased suspended solids and turbidity when compared to unimproved crossings. Additionally, the use of bridges and culverts in ephemeral channel crossings resulted in total suspended solids concentrations that were not statistically different from concentrations measured in unharvested ephemeral channels. These results clearly indicate that either a bridge or a culvert used in ephemeral channels minimizes sediment introduction to these channels following forest harvest.

Pipe bundles were able to significantly decrease total suspended solids concentrations and turbidity compared to the unimproved crossings. However, suspended solids concentrations from channels crossed using pipe bundles were nearly 90 percent higher than from channels crossed with bridges and 70 percent higher than from channels crossed with culverts. While not statistically different, the higher suspended solids concentrations measured in channels crossed using pipe bundles may result from the difficulties associated with controlling sediment introduction to the channel during installation and removal of pipe bundles on such steep slopes. The complication encountered while using pipe bundles in this study resulted from the soil fill depth required on the downstream side of the crossing to make the skid trail level. Removal of this fill using a bulldozer was difficult, and given that much less fill would be required when using a pipe bundle on a gentler slope, these ephemeral channels may simply be too steep for pipe bundles to be as effective an option as bridges or culverts as an improved crossing. Reeves and others (2008) found that suspended solids production from PVC pipe bundles used for similar crossings on a relatively flat stream gradient did not differ significantly from values from portable skidder bridges or culverts. The difference in findings indicates that the topographic condition and stream or channel gradient must be taken into account when crossing recommendations are developed.

No significant differences in total suspended solids were detected between the Hybrid and BMP-plus treatments. This result could indicate that sediment originating from areas adjacent to the channel is negligible compared to the sediment introduced from the channel crossings.

CONCLUSION

Preliminary data suggest the use of improved crossings significantly decreases sediment production and transport in ephemeral channels impacted by forest harvesting operations. Although the 7.6-m equipment limitation zone did not provide additional sediment reductions over that of the improved crossing and tree stringer alone, its influence on thermal protection, as well as on carbon and nitrogen dynamics, warrants further study and will be evaluated. Ephemeral channel monitoring is ongoing, including measurements of discharge. The final dataset will include sediment load data and particle size analysis.

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

EVALUATION OF EROSION AND SEDIMENTATION ASSOCIATED WITH TRACKED VEHICLE TRAINING

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Abstract.—A project was designed to assess erosion and sedimentation associated with tracked vehicle training in the Ft. Knox Military Reservation in Kentucky. The project provided an extensive physical and biotic characterization of the training area, including hydrology, water quality, soils, and vegetation. To determine any changes in channel morphology, cross sections were established and surveyed in June 2008 and 2009 in the four stream channels. All four sampled streams experienced net erosion, losing between 0.31 and 44.45 m³ of sediment a year. Otter Creek, the final receiving stream for the training areas, was sampled weekly from March 2008 through December 2008 just before it entered the training area and upon exiting the training area. Stream TSS concentrations were greater exiting compared to entering the training area. Results will provide baseline data to assist in the development of a plan for sustainable management of the training areas.

INTRODUCTION

The Department of Defense owns more than 10.1 million ha of land dedicated to military training. Military land managers face the difficult task of balancing necessary and realistic military training with protecting and maintaining environmental resources. Natural or anthropogenic changes (such as military training) to the landscape may alter the hydrologic response across a range of scales from a point to an entire drainage basin (Wigmosta and others 2009). Streams are connected to their surrounding watersheds; therefore, disturbances to the linked terrestrial ecosystems are important determinants of stream ecosystem functions and properties (Vannote and others 1980). Disturbances (e.g., soil compaction, and vegetation removal) occurring in military training areas can have significant impacts on the streams draining these areas; increased surface runoff can increase sediment inputs to streams. Managers of military land in the past have overlooked the interrelationships between soil and vegetation variables and the hydrologic impacts of tracked vehicle training (Fuchs and others 2003).

Military training activities involving large-vehicle maneuvers are a land use that has consistently shown negative effects across a variety of terrestrial ecosystems (Quist and others 2003). In terms of increased scour and sediment deposition in streams, disturbances on military lands are similar to forest land clearing for urban and agricultural development (Howarth and others 1991, Quist and others 2003). Most sediment transport from upland erosion in logged areas is facilitated by compacted, disturbed surfaces such as skidder tracks or roads (Croke and others 1999). However, because disturbance from military training is continuous and often lasts for decades (compared to short-term urban or agricultural development, where denuded soils are stabilized by vegetation), catchments within military installations may be subjected to prolonged, repeated

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surficial soil disturbance that can lead to sustained impacts on stream ecosystems (Maloney and others 2005). Large tracked and wheeled vehicles can traverse thousands of hectares in a single training exercise. Soil compaction, loss of vegetative cover, soil erosion, and increased abundance of introduced species are common responses to military training (Goran and others 1983, Milchunas and others 1999, Quist and others 2003).

Soil compaction and the resulting erosion are major impacts of tracked vehicles on military lands. The degree of soil compaction typically varies in proportion to the surface pressure applied (Goran and others 1983). Compaction can be slight with a single vehicle pass, while repeated passes may cause permanent damage to the landscape. An M-1 Abrams tank weighs 58 metric tons (unarmed) and applies approximately 9,200 kg m⁻² of standing ground pressure on the soil surface (Fuchs and others 2003). These tracked vehicles are especially damaging to soil surfaces, and the skidding effect associated with sharp turns crushes and uproots the vegetation and compacts the soil (Diersing and others 1990, Fuchs and others 2003). The collapsed pore structure of the soil reduces infiltration, increases runoff, negatively impacts soil fertility, and inhibits vegetation growth.

It is necessary for land managers to properly manage these areas to keep the land usable for training while minimizing the environmental impacts of the training activities. The Army Sustainable Range Program (SRP) (Army Regulation 350-19) defines responsibilities and policies for maintaining Army-controlled lands. Specifically, the Integrated Training Area Management Program (ITAM) of the SRP is designed to integrate mission requirements with environmental requirements and conservation management practices to ensure that Army lands remain viable for future military training operations. To develop a management plan, a full assessment of the land conditions is needed. The objective of this study was to characterize two extensively used training areas on the Ft. Knox Military Reservation in Kentucky and investigate the impacts training had on soil erosion and stream water quality. This characterization will provide insight into baseline conditions following 40 years of training on this site and a basis for ITAM coordinators to develop a sustainable management plan for these training areas.

STUDY AREA

This study focused on training areas (TAs) 9 and 10 within the Ft. Knox Military Reservation, approximately 50 km southwest of Louisville, KY (Fig. 1). Watersheds in TAs 9 and 10 were delineated and surface flow paths modeled using Light Detection and Ranging (LIDAR) data. Watersheds ranged in size from approximately 12 to 39 ha. Two watersheds within each TA were chosen to examine site and stream characteristics and ultimately to investigate sediment rates exiting the TAs. This area of Kentucky features karst uplands with more than 20 sinkholes in each TA and steep hills along Otter Creek. The upland area, where training occurs, is underlain by the Mississippian St. Louis Limestone. This formation is particularly susceptible to karstification, resulting in typical surficial karst features and limited surface stream development (Connair and Murray 2002).

Soils are fine-grained silts and clays and have significant erosion potential. Within the studied watersheds, soils were fairly homogenous throughout, with Baxter soils and Hammack-Baxter complex soils on the uplands (where vehicle training occurs) and Westmoreland-Caneyville complex soils on the steep slopes along Otter Creek. Baxter soils are severely to very severely eroded and are characterized by very gravelly silty clay loam on the top 15 cm, followed by gravelly clay to a depth of 230 cm. Hammack soils are eroded soils similar to Baxter, except they are overlain by thin fine-silty loess. They are characterized by silty loam on the top 23 cm,

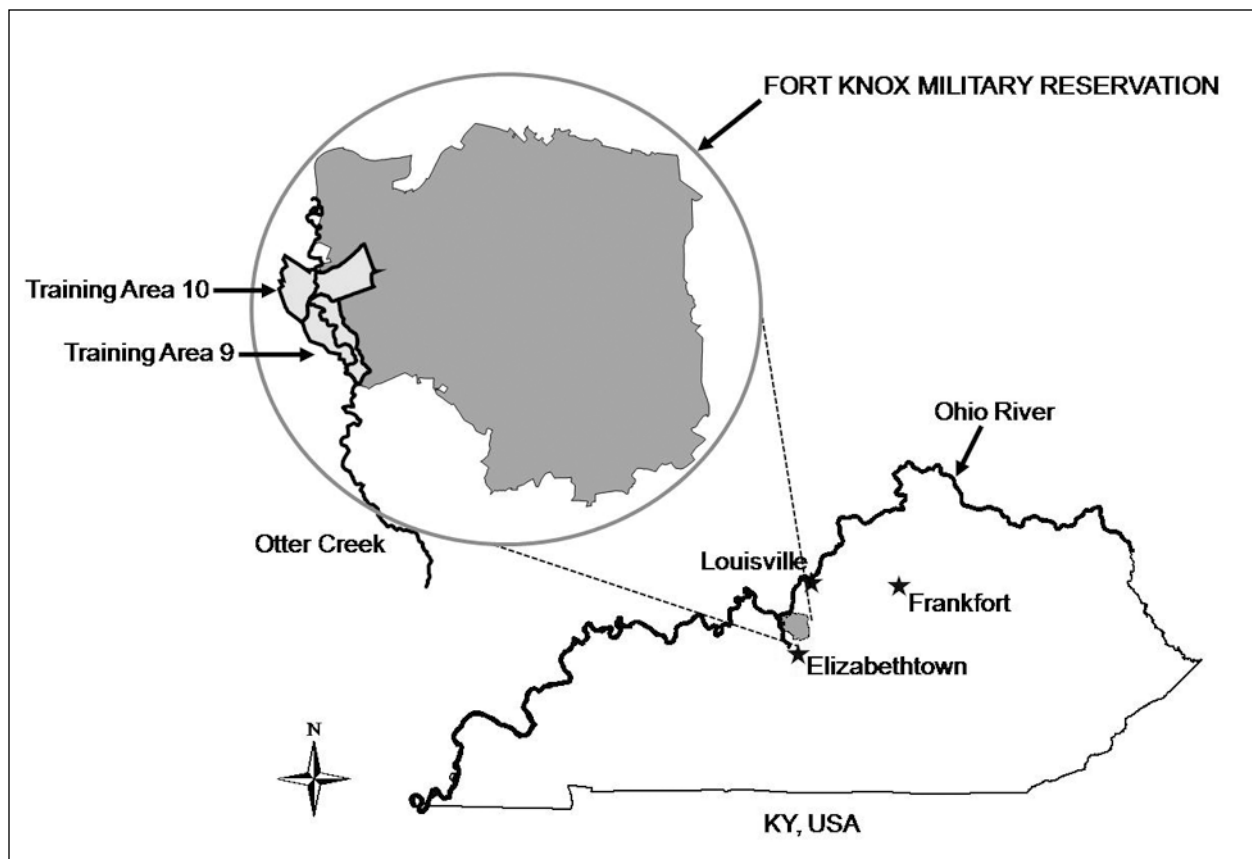


Figure 1.—Study area map of Ft. Knox Military Reservation, KY.

followed by silty clay loam (23-68 cm), extremely gravelly silty clay loam (68-102 cm), and clay (102-218 inches). Plot locations and areas of vehicle training all resided within Baxter or Hammack-Baxter complex soils. The heavily used portions of the TAs are characterized by expanses of exposed mineral soil.

The TAs are hydrologically connected, both through surface and subsurface drainage, to nearby Otter Creek. Otter Creek originates northwest of Elizabethtown, meanders north, and flows approximately 11 km through the reservation, where it then enters Otter Creek Park before draining into the Ohio River. Rills and large gullies exist throughout the TAs and are unstable and ever-changing as a result of continual training. Sediment resulting from severe erosion in the TAs is being transported off-site through surface runoff into the ephemeral streams and through the well developed underground karst drainage network via sinkholes. This area has sustained extensive tracked and wheeled vehicle training operations for more than 40 years. More than 20,000 soldiers utilize the area every year. Tracked vehicles cross the available land area at Ft. Knox at a rate of $20.78 \text{ km yr}^{-1} \text{ ha}^{-1}$ (Goran and others 1983). With the amount of bare area available for tracked vehicle usage in these watersheds and the assumption that vehicles are still using the area at this rate (tracked vehicle training has increased since this estimate was made, but no new numbers have been reported), tracked vehicles are crossing these watersheds approximately 91 to 357 km yr^{-1} (Table 1).

Table 1.—Watershed area, sinkhole drainage area, bare area, and tracked vehicle use separated by watershed in Training Areas 9 and 10 at Ft. Knox Military Reservation, KY.

WS ^a	WS area (ha)	Sinkhole drainage area (ha)	Surface drainage area (ha)	Bare area (ha)	Tracked vehicle usage (km yr ⁻¹)
9-1	11.62	1.54	10.08	4.40	91.43
9-2	15.93	11.49	4.44	5.97	124.09
10-1	14.24	11.42	2.82	8.06	167.49
10-2	38.54	29.00	9.54	17.19	357.17

^aWS refers to watershed. Tracked vehicle usage is based on an estimated usage at Ft. Knox of 20.78 km yr⁻¹ ha⁻¹ (Goran and others 1983). Bare area is based on on-screen digitized aerial photos (2002) and refers to any area devoid of vegetation.

METHODS

SITE CHARACTERIZATION

To gain a better understanding of the landscape, plots were chosen in each watershed for soil and vegetation assessments based on land cover, i.e., road, grass, early succession, and forest. Roads were defined as primary trails within the training area and were completely devoid of vegetation. Grass plots were not currently being used as part of the trail network and consisted of native and introduced species, primarily sericea lespedeza (*Lespedeza cuneata* [Dum. Cours.] G. Don.). Plots considered early succession were patches of open canopy stands and shrubs, while forested plots were closed canopy mature stands. Watersheds contained 37 to 57 percent bare area (no vegetation was present and the land could potentially be used as roads) while the remainder of the land was vegetated. Forested areas were common throughout the training areas and primarily surrounded sinkholes or resided on the uplands along Otter Creek. Early succession and grass areas were less common. Some grass areas changed seasonally as training was often transferred to grass areas that were not staked off during wet portions of the year.

Three fixed 1/10 hectare plots per land cover were established in each watershed for surface soil characterization in August 2007. These plots were randomly selected among all the potential sites within the watersheds. All plots were located on uplands and had similar soil types (Baxter and Hammack-Baxter soils). Within the plots, five subplots were randomly selected to characterize current surface soil conditions based on bulk density, infiltration rate, particle-size class, and penetration resistance. Using the bulk density core method (Culley 1993), we measured soil bulk density by taking a core from each subplot within the watersheds. Cores were oven-dried and weighed to determine weight per given volume and the average bulk density of each land cover was calculated for each watershed. Infiltration rates were determined for each subplot by pouring a known volume of water into a single ring infiltrometer and recording the time it takes the water to infiltrate the soil. The average water infiltration rate of each land cover was calculated for each watershed.

Using the hydrometer method (Sheldrick and Wang 1993), we determined the percent of sand, silt, and clay from a soil sample of each subplot. The average soil particle-size distribution of each land cover was calculated for each watershed, from which soil texture was determined. An Eijkelkamp penetrometer was used to record soil resistance to pressure for each subplot and the average penetration resistance of each land cover was calculated for each watershed (Eijkelkamp Agrisearch Equipment, the Netherlands). Significant differences

in bulk density, infiltration rate, and penetration resistance between land covers were examined using PROC GLM and Tukey's multiple comparison test in SAS (SAS Institute, Cary, NC). Before testing for significant differences, we transformed infiltration data using the natural log to satisfy normality and equal variance assumptions of parametric statistics. All other soil data did not violate these assumptions based on normal probability plots and Levene's test in SAS. For the vegetation assessment, five 1-m² subplots were randomly chosen in each land cover plot and ground cover species were recorded. In addition to the five subplots, all trees greater than 5 cm diameter breast height (d.b.h.) were identified in the forest and early succession plots.

STREAM MORPHOLOGY

To determine any changes in channel morphology, cross-sections were established and surveyed in the four stream channels draining the training areas. Each cross-section was 10 to 20 m apart, depending on stream length. Rebar permanently marked the cross-sections for subsequent surveying. Using a Topcon total station (Topcon Positioning Systems, Livermore, CA), the cross-sections were surveyed in June 2008 and June 2009. Changes in channel morphology were determined by calculating the cross-sectional areas derived from elevation measurements and comparing the 2 years. To calculate the change in volume at each cross-section, the change in area was assumed to be uniform across a 1-m section of stream. An estimated change in total channel volume was calculated by multiplying the average cross-sectional area of the stream by total stream length. This calculation allowed for an overall estimation of deposition or scour within the stream channel.

STORM SAMPLING

In February 2008, stream gauging stations were established in the four watersheds to measure discharge and collect storm samples for total suspended solids determination. To continuously monitor stream stage levels, an OTT Thalimedes (OTT Hydrometry, Kempton, Germany) (a float and pulley system with integrated datalogger) was installed in a PVC pipe and set to monitor stage levels at 5-minute intervals. Rating curves were developed using Manning's Equation to estimate discharge from the continuous stage data (Gore 1996). Manning's Equation uses measurements of cross-sectional area, hydraulic radius, and slope, and an estimate of channel roughness, to calculate discharge at various stage levels. These rating curves allowed for estimation of sediment yields leaving the TAs through surface area drainage.

At each gauging station, an Isco (Teledyne Isco, Inc., Lincoln, NE) automated sampler (Model 6712) was installed and connected to the Thalimedes. When the stream level exceeded a predetermined water level, a signal was sent from the Thalimedes to the Isco to initiate sampling. The Isco was programmed to take a water sample at 30-minute intervals until the stage fell back below the initiation level. Storm samples were collected and returned to the lab for TSS analyses using membrane vacuum filtration methods outlined by the Environmental Protection Agency (US EPA 1999). In addition to instantaneous samples, which provide the TSS concentration at a given time, the event mean concentration (EMC) of each storm event was calculated. The EMC represents the flow-weighted average TSS concentration during a storm event. The flow-weighted average is the sum of the TSS loads calculated for a series of storm samples divided by the sum of the discharges calculated for each of the storm samples.

Additionally, between March 2008 and December 2008, Otter Creek was sampled weekly where it enters and exits the TAs. Weekly sampling captured both baseflow and stormflow events. Samples were returned to the lab for TSS determination via vacuum filtration (US EPA 1999). Significant differences between TSS concentrations entering and exiting the TAs were analyzed using the Kruskal-Wallis test for nonparametric data in SAS.

RESULTS

SITE CHARACTERIZATION

Surface soil conditions were characterized based on bulk density, infiltration rate, particle-size class, and penetration resistance (Figs. 2-5). Road surface soils had the highest bulk density values in all watersheds, with a mean of 1.41 g cm^{-3} and a maximum of 1.81 g cm^{-3} . Grass, early succession, and forest land covers averaged 1.25 , 1.14 , and 1.06 g cm^{-3} , respectively. Surface soils on roads have been heavily compacted by military vehicles, decreasing pore space and thereby increasing bulk density. Roads had the lowest infiltration rates because of the high bulk densities, with a mean infiltration rate of 0.10 cm min^{-1} . Forest and early succession plots had the highest infiltration rates within the watersheds. Grass plots were similar to roads, with very low infiltration rates. This finding was expected, as the grass plots represented areas that received vehicle traffic for at least a portion of the year. Typically, this traffic occurred during the wet seasons of the year (winter, spring), when the vehicles spread out from the primary trail network. Additionally, roads in the TAs generally consisted of a higher clay percentage than did other land covers, likely from erosion of the surface horizon, leaving subsoils with higher clay content exposed. As expected, roads had the highest overall resistance to penetration, followed by grass, early succession, and forest land covers. Common ground cover species found within each plot and major overstory species identified within the forested plots are listed in Tables 2 and 3. *Sericea lespedeza*, introduced into the TAs to seed barren areas, was the primary ground cover species in disturbed areas. Seedling regeneration is limited in forested plots where sediment deposition is significant; Japanese honeysuckle (*Lonicera japonica* Thunb.), poison-ivy (*Toxicodendron radicans* (L.) Kuntze), and Allegheny blackberry (*Rubus allegheniensis* Porter) were the predominant ground cover species in forested areas.

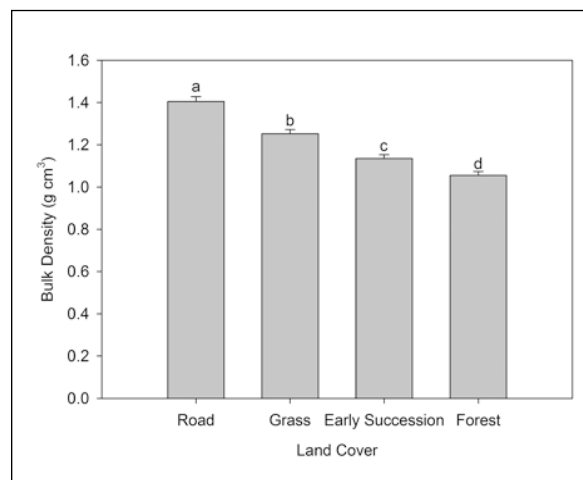


Figure 2.—Soil bulk density with standard errors by watershed and land cover in training areas 9 and 10 at Ft. Knox Military Reservation, KY. Different letters indicate a significant difference ($\alpha=0.05$) in bulk density between land cover. Sample size equals 60 for each land cover.

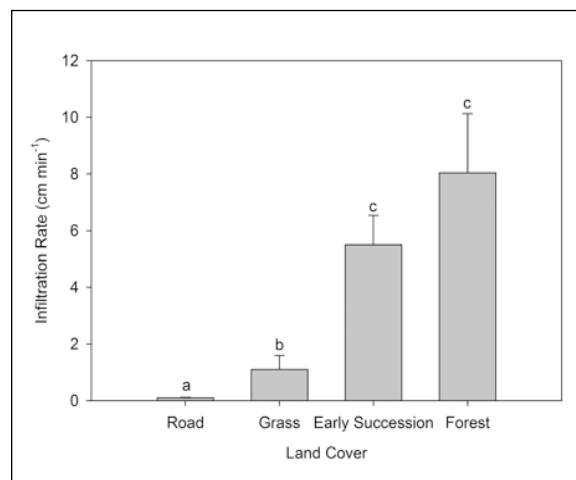


Figure 3.—Soil infiltration rates with standard errors by watershed and land cover in training areas 9 and 10 at Ft. Knox Military Reservation, KY. Different letters indicate a significant difference ($\alpha=0.05$) in infiltration rate between land cover. Sample size equals 60 for each land cover.

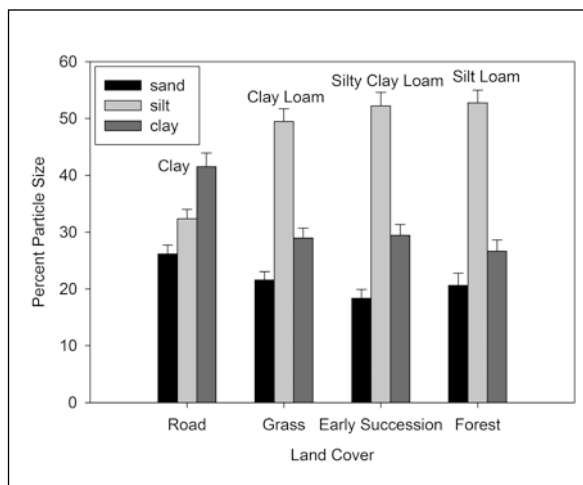


Figure 4.—Soil texture with standard errors by watershed and land cover in training areas 9 and 10 at Ft. Knox Military Reservation, KY.

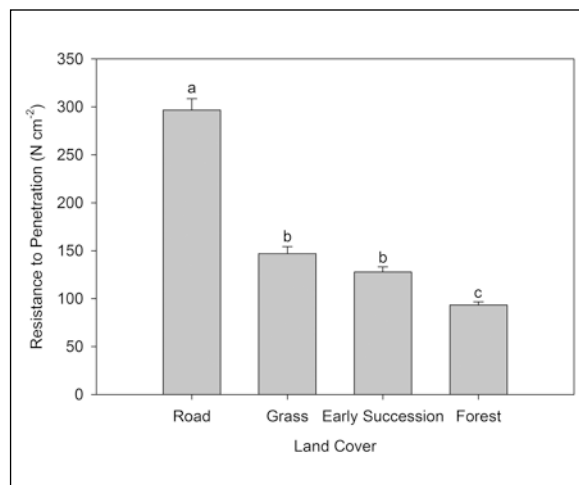


Figure 5.—Mean soil resistance to penetration with standard errors by land cover in training areas 9 and 10 at Ft. Knox Military Reservation, KY. Different letters indicate a significant difference ($\alpha=0.05$) in penetration resistance between land cover. Sample size equals 60 for each land cover.

Table 2.—Common ground cover species in land cover plots (five 1-m² subplots within each land cover) in training areas 9 and 10 at Ft. Knox Military Reservation, KY.

Common name	Scientific name	Native or introduced	Main plot type(s) ^a
Annual marsh elder	<i>Iva annua</i> L.	Native	G
Black medick	<i>Medicago lupulina</i> L.	Introduced	E
Blackberry	<i>Rubus allegheniensis</i> Porter	Native	E, F
Coralberry	<i>Symphoricarpos orbiculatus</i> Moench.	Native	F
Crabgrass	<i>Digitaria</i> spp.	Introduced	F
Dallisgrass	<i>Paspalum dilatatum</i> Poir.	Introduced	E
Eastern poison-ivy	<i>Toxicodendron radicans</i> (L.) Kuntze	Native	E, F
Fall panicgrass	<i>Panicum dichotomiflorum</i> Michx.	Native	G
Giant foxtail	<i>Setaria faberi</i> Herrm.	Introduced	E
Goldenrod	<i>Solidago missouriensis</i> Nutt.	Native	G, E
Japanese honeysuckle	<i>Lonicera japonica</i> Thunb.	Introduced	E, F
Periwinkle	<i>Vinca minor</i> L.	Introduced	F
Sericea lespedeza	<i>Lespedeza cuneata</i> (Dum. Cours.) G. Don.	Introduced	G, E
White heath aster	<i>Symphyotrichum ericoides</i> (L.) G.L. Nesom.	Native	G, E, F
White snake root	<i>Ageratina altissima</i> (L.) King & H. Rob.	Native	F
Woodland sunflower	<i>Helianthus divaricatus</i> L.	Native	E
Yellow foxtail	<i>Setaria pumila</i> (Poir.) Roem. & Schult.	Introduced	G, E

^aG refers to grass; E refers to early succession; F refers to forest.

Table 3.—Major overstory species (> 5 cm diameter breast height) in forested plots (1/10 ha) in training areas 9 and 10 at Ft. Knox Military Reservation, KY. All species are native to the area.

Common name	Scientific name
American elm	<i>Ulmus Americana</i> L.
^a American sycamore	<i>Platanus occidentalis</i> L.
Black cherry	<i>Prunus serotina</i> Ehrh.
Black locust	<i>Robinia pseudoacacia</i> L.
Black willow	<i>Salix nigra</i> Marsh.
Blackjack oak	<i>Quercus marilandica</i> Munchh.
Boxelder	<i>Acer negundo</i> L.
Common persimmon	<i>Diospyros virginiana</i> L.
^a Eastern cottonwood	<i>Populus deltoids</i> Bartram ex Marsh.
Eastern redcedar	<i>Juniperus virginiana</i> L.
Flowering dogwood	<i>Cornus florida</i> L.
Kentucky coffeetree	<i>Gymnocladus dioicus</i> (L.) K. Koch
Pin oak	<i>Quercus palustris</i> Munchh.
Red maple	<i>Acer rubrum</i> L.
^a Sassafras	<i>Sassafras albidum</i> (Nutt.) Nees
Shingle oak	<i>Quercus imbricaria</i> Michx.
Swamp white oak	<i>Quercus bicolor</i> Willd.

^aThree most common species.

STREAM MORPHOLOGY

Forty-two cross-sections were surveyed. A cross-section example from watershed 9-1 is provided (Fig. 6). Each cross-section underwent scour or deposition, or was unchanged. The average cross-sectional volume change and the estimated change in total stream channel volume are provided for each watershed (Table 4). While all streams experienced a net change of scour, losing between 0.31 and 44.45 m³ of sediment over the year, each stream had cross-sections that underwent sediment deposition. The largest changes in cross-sectional volumes resided in watershed 9-1, ranging from a loss of 1.77 m³ to a gain of 0.72 m³ over a 1-m section of stream. This stream had areas of deep incision and widening, compared to the other watersheds. Watershed 10-2 experienced the smallest range in volume change (-0.16 m³ to 0.16 m³). Much of the stream channel in this watershed is bedrock, with some cross-sections remaining unchanged between the 2 years.

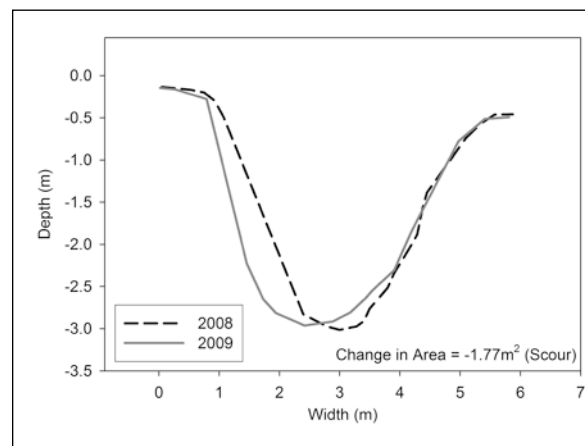


Figure 6.—Example of a stream cross-section in watershed 9-1 comparing the 2008 and 2009 morphology survey in training areas 9 and 10 at Ft. Knox Military Reservation, KY.

Table 4.—Changes in stream channel morphology in training areas 9 and 10 at Ft. Knox Military Reservation, KY.

Stream	Mean change in cross-section volume (m ³ yr ⁻¹)	Total change in channel volume (m ³ yr ⁻¹)	Net stream channel change
9-1	-0.17 ± 0.26	-27.80	Scour
9-2	-0.37 ± 0.26	-14.81	Scour
10-1	0.01 ± 0.03	-0.31	Scour
10-2	-0.11 ± 0.07	-44.45	Scour

STORM SAMPLING

Watersheds

Twenty-six storm events were captured in the four watersheds combined. An example hydrograph of a storm event in watershed 10-2 and the associated TSS samples are provided (Fig. 7). Mean TSS concentrations of each watershed (based on instantaneous samples) ranged from 314 to 6,413 mg L⁻¹ (Table 5). TSS EMCs of each watershed (based on flow-weighted samples) ranged from 640 to 10,521 mg L⁻¹ (Table 5). Using the rating curves developed from continuous stage data, sediment yields were calculated on a per-area basis. Mean TSS yields ranged from 6 to 127 metric ton yr⁻¹ ha⁻¹ (Table 5). It is estimated that TAs 9 and 10 are exporting 27 and 40 metric ton yr⁻¹ ha⁻¹, respectively, through surface water drainage.

Otter Creek

Forty-three samples (31 baseflow and 12 stormflow) were taken. Mean TSS concentrations were higher exiting than entering the TAs during baseflow, stormflow, and combined flows (Fig. 8). Additionally, 74 percent of combined flow samples, 63 percent of baseflow samples, and 100 percent of stormflow samples had higher values exiting versus entering the TAs.

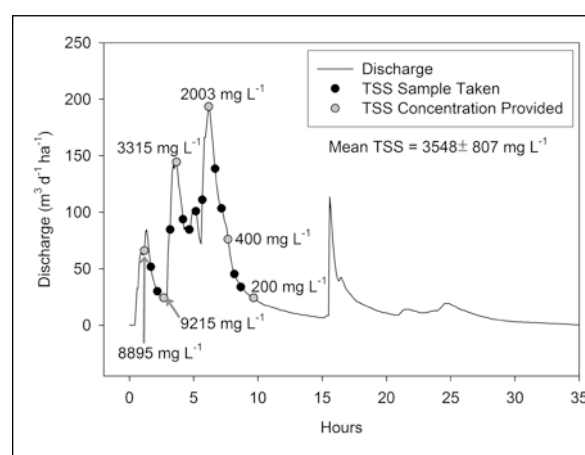


Figure 7.—Example of a sampled storm event in training areas 9 and 10 at Ft. Knox Military Reservation, KY: Watershed 10-2, April 19, 2009, 3.12 cm rainfall. Total suspended solids (TSS) concentrations measured at 30 minute intervals along the hydrograph.

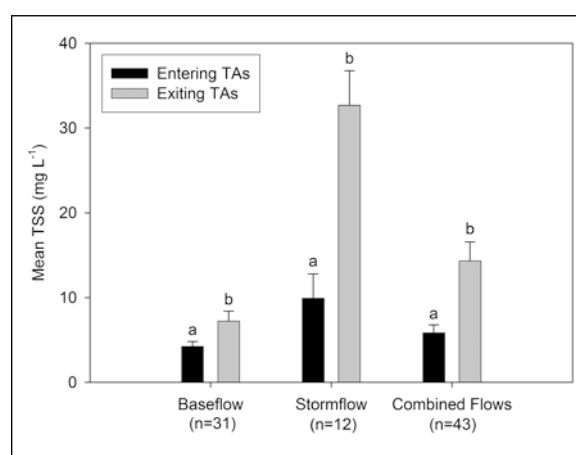


Figure 8.—Total suspended solids (TSS) concentrations with standard errors during baseflow and stormflow sampling of Otter Creek, the receiving stream of training areas 9 and 10 at Ft. Knox Military Reservation, KY. Different letters indicate a significant difference ($\alpha=0.05$) in TSS within flow categories. 'TA' refers to training area.

Table 5.—Mean watershed discharge and total suspended solids concentrations and loads during storm events in training areas 9 and 10 at Ft. Knox Military Reservation, KY.

WS ^a	TSS EMC (mg L ⁻¹)	TSS (mg L ⁻¹)	TSS (metric ton yr ⁻¹ ha ⁻¹)	Discharge during sampling (m ³ d ⁻¹ ha ⁻¹)
9-1	741 ± 56	591 ± 62	33 ± 6	127 ± 15
9-2	10,521 ± 6,040	6,412 ± 1,288	23 ± 19	6 ± 2
10-1	640 ± 294	314 ± 80	26 ± 6	114 ± 17
10-2	3,666 ± 1,761	2,443 ± 420	43 ± 10	40 ± 3
TA 9	7,727 ± 4,541	4,084 ± 817	27 ± 12	49 ± 8
TA 10	3,018 ± 1,413	1,817 ± 307	39 ± 8	55 ± 5
Overall	4,587 ± 1,781	2,765 ± 391	33 ± 7	52 ± 5

^aWS refers to watershed; TSS refers to total suspended solids; EMC refers to event mean concentration; TA refers to training area.

DISCUSSION

The soil and vegetation disturbance from training activities has resulted in bare ground, significant surface compaction, and erosion down to the subsoils in many areas. These effects are damaging the water quality of the subsurface drainage network since much of the eroded soil is transported into sinkholes through flowpaths and gullies in the TAs. Groundwater tracer tests were performed on nearby sinkholes and sinking streams (injection sites were not within the boundaries of TAs 9 and 10) to determine groundwater flowpaths and identify the potential for contamination of McCracken Springs, a source of public water supply located on TA 9 (Taylor and McCombs 1998). Contaminants, such as increased sediment, could be expected to resurface at McCracken Springs within 1 to 5 days, depending on distance of the entry point and hydrologic flow conditions (Taylor and McCombs 1998). Additionally, soil from the TAs is deposited directly into the ephemeral stream networks via overland flow. The subsurface and surface drainage network combine to create an extensive network of flowpaths that are collecting and transporting sediment into Otter Creek, the final receiving stream of these TAs.

The effects of both catchment and localized disturbances such as military training, forest clearing, agricultural practices, and urban development can be particularly strong during storm events. Houser and others (2006) found mean storm event TSS concentrations ranging from 847 to 1,881 mg L⁻¹ in watersheds with high disturbance at Ft. Benning Military Installation in Georgia. Lenat and Crawford (1994) revealed higher TSS concentrations during storm events in urban watersheds (440 mg L⁻¹) followed by agricultural (267 mg L⁻¹) and forested (198 mg L⁻¹) watersheds in North Carolina; they estimated sediment yields in urban watersheds to be 1.32 metric ton yr⁻¹ ha⁻¹ compared to 0.291 metric ton yr⁻¹ ha⁻¹ in the forested watershed. Comparatively, this study found elevated mean TSS concentrations exiting the training areas during storm events (314 to 6,413 mg L⁻¹) and much greater yields of sediment leaving the TAs per yr (23 to 43 metric ton yr⁻¹ ha⁻¹) than with urban or agricultural disturbance. Additionally, mean watershed TSS EMCs (640 to 10,521 mg L⁻¹) in this study were much greater than that of a 90th-percentile urban site (300 mg L⁻¹) sampled as part of the U.S. Environmental Protection Agency's Nationwide Urban Runoff Program (US EPA 1983).

Soil texture is related to soil bulk density and is an indicator of a soil's susceptibility to compaction. Fine-textured loamy and clayey soils are generally considered more vulnerable to compaction and therefore are more susceptible to impacts of heavy loads (Milchunas and others 1999). Bulk densities are generally higher

deeper in the soil profile, where the organic matter content is low, fewer roots exist, and the overlying weight of the soil profile has increased compaction. Roads in the TAs have been eroded of upper soil horizons as a result of training exercises and consequently have high bulk densities compared to forested areas, where the topsoil is still intact. High bulk densities and decreased vegetative cover have reduced infiltration into the soil. Thurow and others (1993) found an increase in bulk density on wet-tracked sites at Ft. Hood, TX, with a bulk density of 1.13 g cm^{-3} in the control and 1.33 g cm^{-3} after 10 passes of an M2 Bradley Infantry Fighting Vehicle; infiltration rates also declined as the number of passes over wet soils increased. Comparatively, Whitecotton and others (2000) characterized surface soils after 2 years of intensive training via foot traffic at the U.S. Air Force Academy in Colorado. They found a mean bulk density and infiltration rate of 1.37 g cm^{-3} and 0.63 cm min^{-1} in high-use training sites versus 1.04 g cm^{-3} and 3.83 cm min^{-1} in reference sites with little to no training. Both Whitecotton and others (2000) and this study revealed significantly higher soil bulk density and significantly lower infiltration rates in the heavily used areas compared to the reference and forest areas. Reduced infiltration generates more surface runoff into the sinkholes and ephemeral streams of TAs 9 and 10, resulting in continued erosion. Sediment transported off-site is impairing the water quality of Otter Creek, as evidenced by higher TSS concentrations exiting the TAs than entering the TAs.

MANAGEMENT IMPLICATIONS

Without restorative measures, erosion will continue along with the resulting impacts on water quality in the subsurface system and Otter Creek. Erosion causes loss of soil fertility, impacting the success of terrestrial restoration activities even after disturbance has ceased. In much of the TAs, topsoil has been removed, leaving compacted soils deeper in the profile exposed and thereby increasing the difficulty of vegetation regeneration. Though natural weathering processes such as freezing and thawing and renewed root growth can eventually restore the structure of the compacted soil (spanning 1,000 years or more), continued use of the area keeps the surface soils in a compacted state (Goran and others 1983). U.S. Department of Defense leadership and land managers at the Ft. Knox Military Reservation are concerned about the environmental impacts of training activities and are directing the implementation of best management practices to reduce the amount of sediment entering the sinkholes and ultimately Otter Creek.

TAs 9 and 10 at Ft. Knox have undergone continued repeated training for more than 40 years. Areas that receive constant, intense use, such as these TAs, may have degraded beyond the point of restoring the ecological integrity of the landscape, but active management could allow continued use for training and reduce off-site sediment transport.

It is difficult to assess the success of vegetation recovery in this area because training was so widely dispersed throughout the site. However, vegetated buffer zones have been a long-standing means of protecting streams from anthropogenic disturbances. These buffer zones have been incorporated into best management practices in forestry, agriculture, and engineering, with varying degrees of success for intercepting sediment inputs. Grass cover is considered one of the most effective agents in filtering and depositing sediment from surface runoff because of the extensive dense ground cover (Wischmeier and Smith 1978). Grass buffers should be extended and maintained along the primary connected concentrated flow paths to promote sustained herbaceous cover. Additionally, tank trails within sinkhole buffers should be cyber-staked (a marking methodology that informs troops of areas of restricted use) and replanted to reduce sediment transport into the sinkholes.

Gully stabilization would also reduce soil erosion and help maintain the areas for future training. Depending on the severity of gully incision and widening, appropriately sized check dams could be installed to stabilize the channel and limit further incision and widening. Similarly, stabilization of the ephemeral streams draining the watersheds would likely reduce the scouring evident in this study.

CONCLUSIONS

Disturbance over the past decades in the military vehicle training areas has caused significant soil compaction and soil loss by erosion, leaving subsoils exposed. Altering the soil in this way can have detrimental effects on vegetation growth and sustainability. With continued loss of vegetation, realistic training efforts may be difficult as there are fewer places for soldiers and vehicles to take cover in training exercises. In addition to the training challenges, ecosystems linked to the TAs are being degraded, impacting water quality in the underground karst network and Otter Creek. The U.S. Department of Defense is taking a proactive approach in addressing the environmental impacts of training activities by implementing best management practices, such as vegetated buffer zones, to reduce the amount of sediment entering sinkholes and streams.

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

DETECTING THE EFFECTS OF FOREST HARVESTING ON STREAMFLOW USING HYDROLOGIC MODEL CHANGE DETECTION

Nicolas P. Zegre and Nicholas A. Som¹

Abstract.—Knowledge of the effects of forest management on hydrology primarily comes from paired-catchment study experiments. This approach has contributed fundamental knowledge of the effects of forest management on hydrology, but results from these studies lack insight into catchment processes. Outlined in this study is an alternative method of change detection that uses a simple rainfall-runoff model to generate hypotheses about the impact of forest harvesting on streamflow processes. We develop and apply our approach in a managed headwater catchment in the Fernow Experimental Forest in West Virginia. Monte-Carlo simulations are used to generate hydrologic model parameter distributions prior to and following clearcut harvesting. Differences in the distributions of pre- and post-harvest models are proxy for changes in the catchment processes involved in streamflow generation. Changes in model parameters show that harvesting increased the volume of water contributed to catchment storage, thereby increasing baseflow volume, decreasing the time for the catchment to drain, and increasing stormflow response. The change detection approach presented in this study is a useful way to advance our understanding of catchment processes and the effects of disturbance on hydrology.

INTRODUCTION

The effect of forest management on catchment-scale hydrology remains a central research interest in water resources management worldwide (Bosch and Hewlett 1982). There is little doubt of the relationship between forests and hydrology, but the sustainability of water resources depends on both the ability to detect changes that occur following forest harvesting and understanding process-level changes in runoff generation, storage, and movement of water through forested catchments.

Traditional approaches for discerning the effects of forest management on hydrology have focused on the paired-catchment approach, where statistical models relate streamflow in a harvested catchment to streamflow in an undisturbed catchment. Results from a plethora of these studies conducted across the globe generally show streamflow increases following harvesting (e.g., Bosch and Hewlett 1982, Campbell and Doeg 1989, Moore and Wondzell 2005, Eisenbies and others 2007). Changes to process-level hydrology, however, are inferred rather than measured as the paired-catchment method relates streamflow measured at the outlets of two watersheds. Although this approach has greatly increased understanding of the effects of forest management on hydrology, it has not illuminated catchment processes.

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Studies evaluating the effects of forest harvesting on streamflow generation mechanisms are limited due in part to the difficulties associated with observing and measuring catchment processes. Though advances using stable isotope hydrology (e.g., Sklash and Farvolden 1979) have been useful in identifying water sources, partitioning stormflow, and determining water residence time, comprehensive understanding of process-level changes are limited in space and time.

In this study we use a simple rainfall-runoff model to simulate the effects of forest harvesting on daily streamflow to generate hypotheses regarding process-level changes in catchment hydrology resulting from disturbance. Pre- and post-harvest models are compared to shed light onto how forest harvesting affects catchment processes and streamflow generation.

METHODS

STUDY SITE AND HYDROMETRIC DATA

Long-term research conducted at the Fernow Experimental Forest offers a unique opportunity to develop and test the efficacy of detecting change using hydrologic models. The Fernow (Fig. 1) is located in West Virginia and was established in 1934 by the U.S. Department of Agriculture Forest Service, to study the effects of forest and watershed management in the northern and central Appalachian Mountains. Results from forest harvesting studies conducted at the Fernow show that forest harvesting can increase annual streamflow and is related to both the intensity of harvest and basal area (ba) removed (Adams and others 2004).

The Fernow annually receives on average 1,480 mm of precipitation that is evenly distributed throughout the year. Slopes are steep and average elevation for the basin is 762 meters. Vegetation in the Fernow is characterized as mixed mesophytic, currently dominated by oaks (*Quercus* spp.), yellow-poplar (*Liriodendron tulipifera*), and sugar maple (*Acer saccharum*).

This study was conducted using daily streamflow and climate records for Fernow watershed 7 (WS7). WS7 is a small (0.24 km²) east-facing headwater catchment that has continual streamflow monitoring since

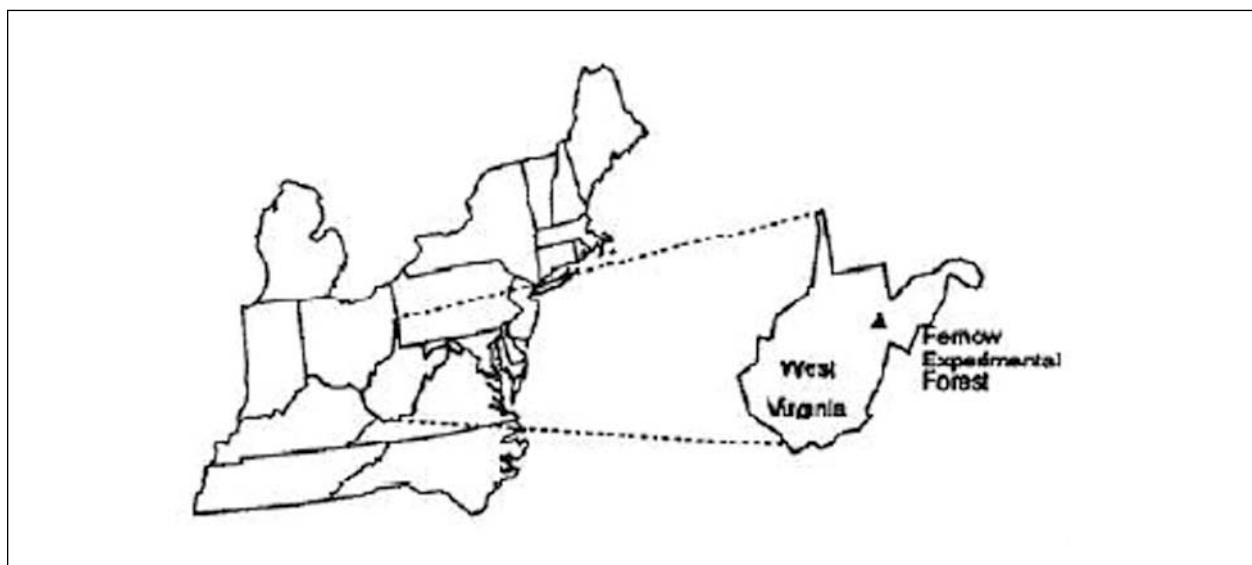


Figure 1.—Location of the Fernow Experimental Forest, West Virginia.

November 1956. The upper half of WS7 was clear-cut harvested from November 1963 to March 1964, removing approximately 49 percent of the watershed basal area. WS7 was maintained barren with herbicides from May 1964 through October 1969. The remaining 51 percent basal area was removed from the lower half of the catchment during a second harvesting treatment applied between October 1966 and March 1967. The entire catchment was maintained barren with herbicides until October 1969, when the catchment was allowed to recover naturally. The extent of the current study, however, is limited to the pre-harvest period (November 1956-October 1963) and the period following the first harvest treatment (November 1963-October 1966).

HYDROLOGIC MODELING AND IDENTIFICATION

The Identification of unit Hydrographs And Component flows from Rainfall, Evaporation, and Streamflow data (IHACRES) (Jakeman and others 1990) model was used to simulate daily streamflow prior to and following forest harvesting in a headwater catchment. IHACRES simulates streamflow using daily inputs of areal precipitation and air temperature. The model consists of two modules, a nonlinear rainfall loss module that converts rainfall to effective rainfall, the proportion of rainfall that eventually becomes streamflow, and a linear unit hydrograph module that represents the transformation of effective rainfall to streamflow (Fig. 2). The model is defined by six parameters; three (t_w , f , and c) are used in the nonlinear module and the remaining three (t_q , t_s , v_s) relate to the linear module (Post and Jakeman 1999). The parameter " t_w " (days) is the time constant governing rate of water loss from the catchment at 20 °C, f varies the rate of catchment water loss due to a unit change in temperature, and c (mm) is a scale parameter that minimizes bias so that the volume of effective rainfall is equal to the total streamflow. The parameter " t_q " (days) is the time constant governing the rate of quickflow recession of streamflow, t_s (days) is the time constant governing the rate of slowflow or baseflow recession of streamflow, and v_s is the ratio of slowflow to total flow. The parameter set, $\tau = \{t_w, f, t_q, t_s, \text{ and } v_s\}$, is determined directly from the observed rainfall, streamflow, and temperature while c is optimized such that total volume of modeled effective precipitation is equal to total volume of observed streamflow. Detailed descriptions of the IHACRES model can be found in Post and Jakeman 1996 and 1999.

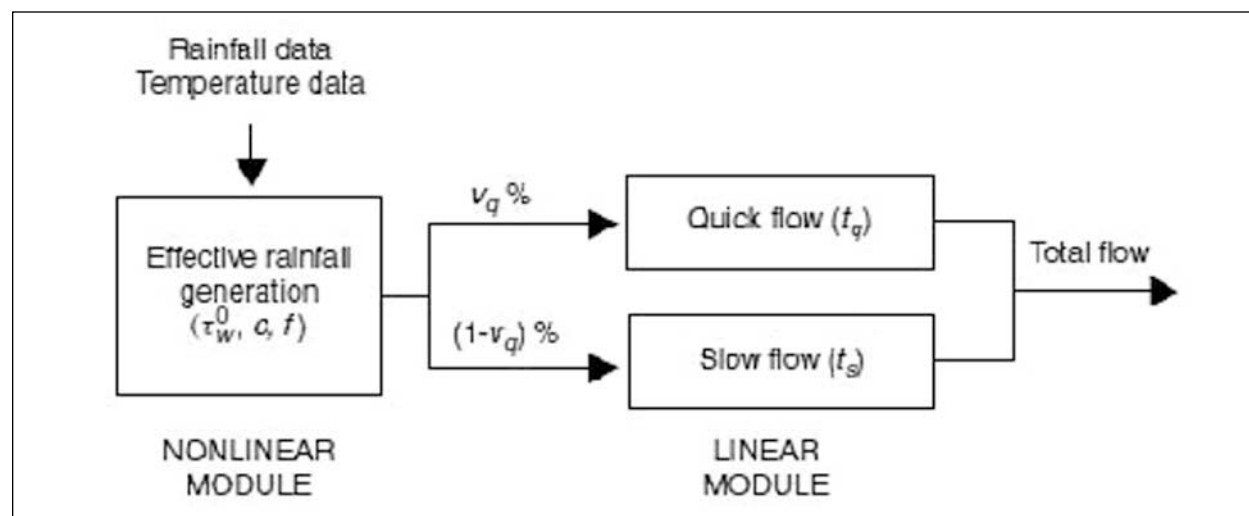


Figure 2.—Schematic of the IHACRES rainfall-runoff model. Parameters are defined as follows: t_w is the time constant (days) governing rate of water loss from the catchment at 20 °C; f is the rate of catchment water loss due to a unit change in temperature (unitless); c (mm) is a scale parameter used to minimize bias so that the volume of effective rainfall is equal to the total streamflow; t_q is the time constant (days) governing the rate of quickflow recession of streamflow; t_s is the time constant (days) governing the rate of slowflow or baseflow recession of streamflow; and v_s is ratio of slowflow to total flow (from Kokkonen and others 2003).

IHACRES was used to simulate daily streamflow in WS7 using long-term precipitation and air temperature records (3,283 days). The IHACRES model was calibrated using observed daily streamflow measured at the outlet of WS7 for the pre-harvest (2,188 days) and post-harvest (1,095 days) periods.

Many studies have demonstrated the difficulty of identifying, calibrating, and validating hydrologic models. Uncertainties in model structure, parameterization, and observed data make the search for an optimal parameter set identified from manual calibration not practical and model validation impossible (Oreskes and others 1994). Therefore, we used the generalized likelihood uncertainty estimation (GLUE) (Beven and Binley 1992) method to identify populations of appropriate models used for this analysis. GLUE is a Bayesian-like parameter estimation method that allows the conditional assessment of model uncertainty using prior knowledge of model performance to construct probability distributions for each set of model parameters. GLUE requires Monte-Carlo simulations using a large number of parameter sets to assess the performance of each model for simulating streamflow relative to observed streamflow. We used Monte-Carlo simulations to randomly sample values for each parameter from non-informative uniform distributions to generate 5,000 candidate models. The Nash-Sutcliffe measure of efficiency (NSE) (Nash and Sutcliffe 1970) was used to evaluate the goodness of fit between observed and simulated streamflow and identify populations of the best performing models. We used a $NSE > 0.4$ to define behavioral models, those models that acceptably reflect the behavior of the modeled system (Beven 2001).

PARAMETER CHANGE DETECTION

The populations of model parameters identified from the best fitting models (Monte-Carlo) were used to develop 'behavioral' distributions that show probable and acceptable ranges of the likely model parameters that have good agreement between simulated and observed streamflow for the pre- and post-harvest periods. Box and whisker plots and the nonparametric two-sided Wilcoxon rank sum test were used to test the null hypothesis that there are no differences between pre- and post-harvest parameter distributions. As model parameters are proxy for the state variable responsible for streamflow generation, rejection of the null hypothesis suggests changes in the underlying processes that control the timing and distribution of runoff during respective periods.

RESULTS

ANNUAL PRECIPITATION AND STREAMFLOW CHANGES

Precipitation and streamflow varied by year and season (Fig. 3). Water year 1958 (1 October 1957 - 30 September 1958) (WY 58) and WY 57 were the wettest and driest years during this study with approximately 1,751 and 1,287 mm, respectively. The largest peakflow event (52 mm) occurred on 5 March 1963, during the preharvest period. Runoff ratios, calculated as the ratio of total streamflow to total precipitation, averaged 56 percent (51-59 percent) and 52 percent (40-61 percent) during the pre- and post-harvest periods, respectively. Streamflow measured during WY 60 exhibited nonstationarity and was excluded from analyses. Nonstationarity implies systematic change in mean and/or variance over time and violates assumptions of time-series analysis.

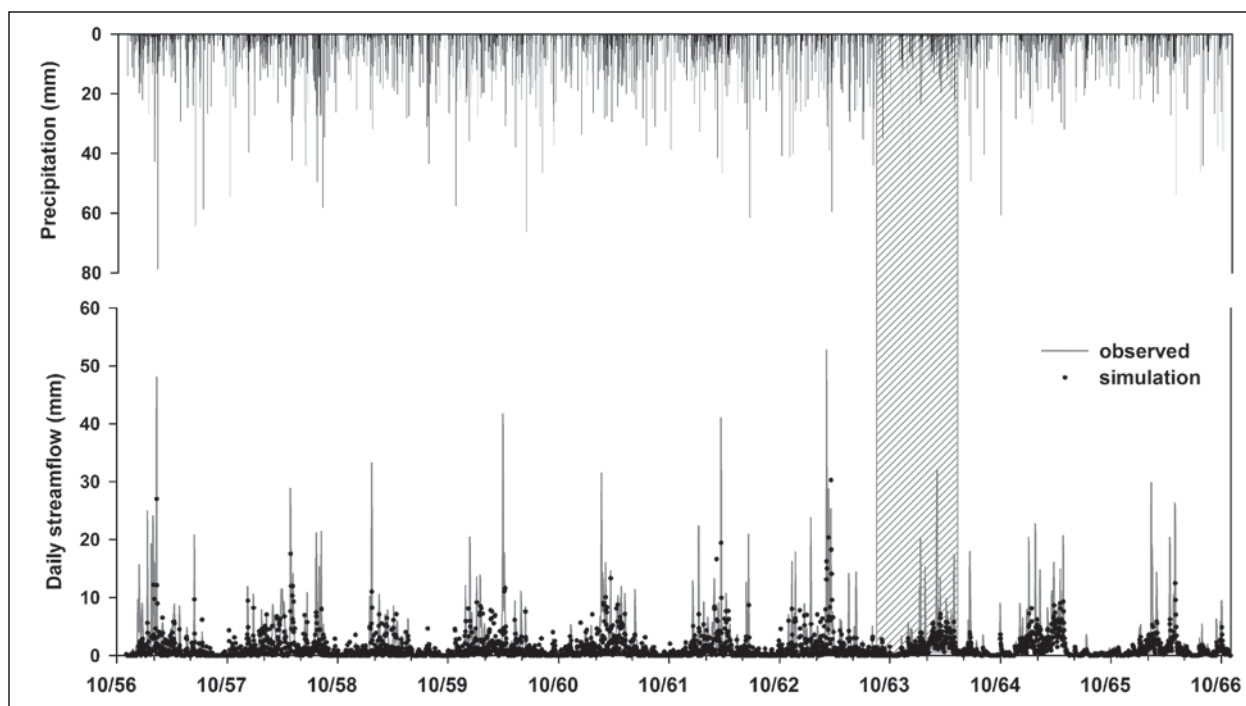


Figure 3.—Daily precipitation and observed and simulated streamflow during study period for Watershed 7 of the Fernow Experimental Forest, WV. The plotted simulated streamflow represents the median streamflow of all behavioral models identified by GLUE. Shaded area notes harvest treatment period.

CHANGES IN PARAMETER DISTRIBUTIONS

The two-sided Wilcoxon rank sum test was used to detect decreasing and increasing shifts between pre- and post-harvest parameter distributions. Significant differences ($\alpha = 0.05$) between pre- and post-harvest distributions were detected for linear module components while parameters in the nonlinear effective rainfall module were robust to change (Table 1). Parameter distributions for v_s and t_q increased while t_s decreased following forest harvesting (Fig. 4). Median values for post-harvest parameter distributions increased by 31 and 141 percent, respectively, for v_s and t_q , and decreased by 33 percent for the parameter t_s .

Table 1.—Pre- and post-harvest model parameter distributions from Monte-Carlo simulations. +++ (---) signify significant increases (decreases) or no change (n/c) between pre- and post-harvest distributions. See methods section for definition of IHACRES model parameters.

Model Parameter	Median value of pre-harvest parameter distributions (1956-1963)	Median value of post-harvest parameter distributions (1963-1966)	Change in post-harvest parameter distributions relative to pre-harvest
t_w	15	15	n/c
f	2	2	n/c
v_s	0.6	0.8	+++
t_s	7	5	---
t_q	0.5	1	+++

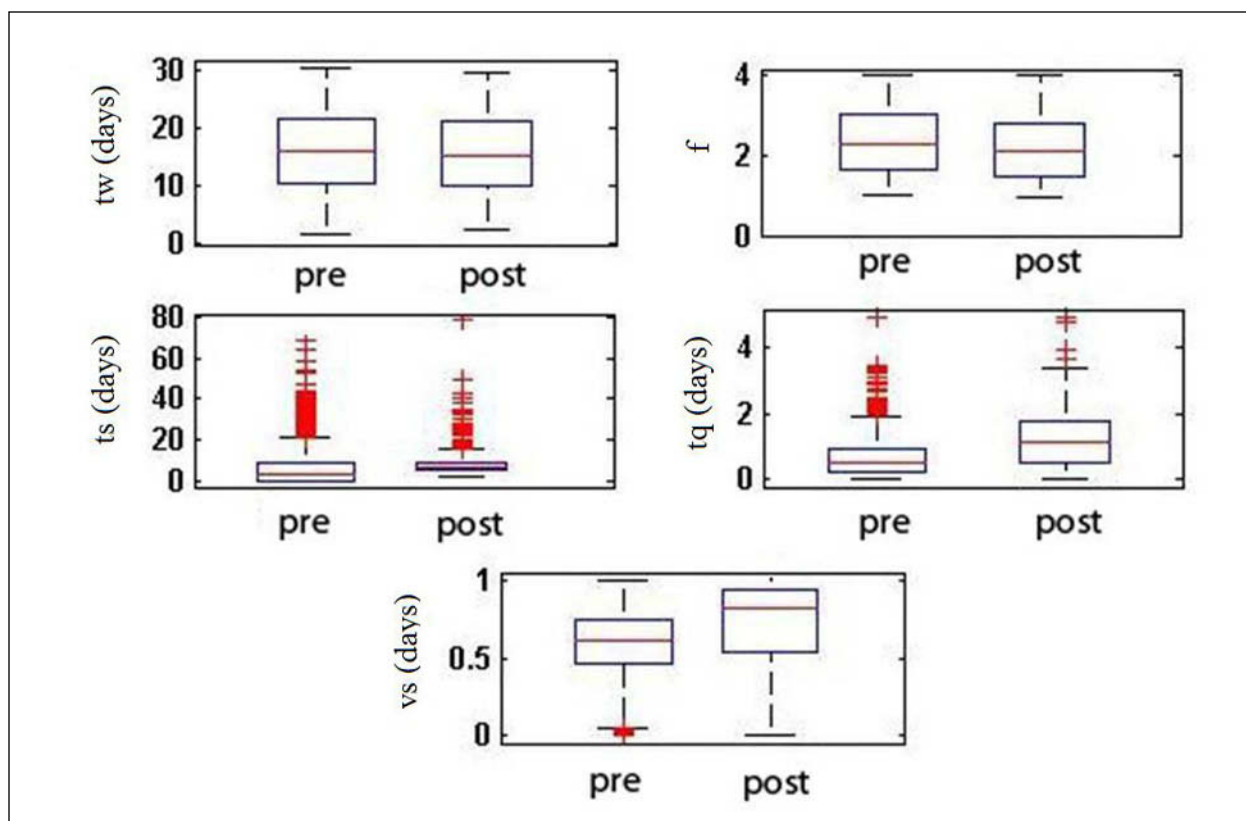


Figure 4.—Box and whisker plots for pre- and post-harvest parameter distributions for model simulations using the IHACRES model.

DISCUSSION

MODELING THE EFFECTS OF FOREST HARVESTING ON STREAMFLOW

Behavioral populations of the IHACRES model successfully captured the shape and timing of the hydrograph during pre- and post-harvest periods but consistently underestimated peak stormflows. Errors in simulated streamflow were attributed to model structure, parameterization, and uncertainty and measurement error in input data.

The parameter set $\tau = \{t_w, f, v_s, t_s, t_q\}$ characterizes the dynamic hydrologic response of a catchment (Fig. 2) (Kokkonen and others 2003). Monte-Carlo simulations were used to construct probable distributions of τ to model the hydrologic responses of WS7 during pre- and post-harvest periods. Different distributions of τ between these periods are proxy for changes in the catchment processes resulting from forest harvesting.

Significant changes in linear module parameters indicate that a larger proportion of rainfall is contributed to catchment storage and streamflow generation. Forest harvesting typically augments streamflow by reducing canopy interception and transpiration, and modifying soil moisture conditions (e.g., Bosch and Hewlett 1982, Eisenbies and others 2007). These effects are reflected by increases in parameters v_s , the proportion of streamflow derived from baseflow, and t_q , the rate of quickflow recession; and by decreases in t_s , the rate of baseflow recession. Changes in model parameters, therefore, show that harvesting increases the volume of

water contributed to catchment storage, thereby increasing baseflow volume (v_s), decreasing the time for the catchment to drain (t_s), and increasing stormflow response (t_q). These responses were shown by Rothacher (1973) and Adams and others (1991). Rothacher (1973) showed increases in soil moisture and streamflow following partial (25-percent area) clear-cut harvesting of a forested catchment in the Pacific Northwest. Soil moisture averaged more than 10 cm greater in harvested areas of the watershed and streamflow increased by as much as 160 mm.

Our results contrast with those of Post and others (1996), who used the IHACRES model to evaluate land-cover changes on the hydrologic responses of Picaninny Creek, a mountain ash (*Eucalyptus regnans*)-dominated catchment in south east Australia. Model parameters that govern the rate of catchment drying (t_w), recession (t_q and t_s), and volumetric throughput of quickflow and slowflow (v_s) did not change between pre- and post-harvest periods. These outcomes suggest that (1) harvesting-induced changes were relatively small compared to natural variability (Post and Jakeman 1996); and/or (2) the presence or absence of mountain ash was not important in determining the hydrologic response of Picaninny Creek (Post and Jakeman 1996).

The changes in parameter distributions in our study clearly show that forest harvesting altered the hydrologic response of WS7 in the Fernow Experimental Forest during the first harvesting period. Differences in hydrologic responses between WS7 and Picaninny Creek are explained mostly by differences in soil depth and harvesting treatments and in part by vegetation and location. For example, soil depth in the Fernow averages 1 meter (Adams and others 2004), in contrast with 10 to 15 meters in Picaninny Creek (Post and Jakeman 1996). Soil depth is a primary factor determining how fast subsurface water moves through a catchment. The deeper soils in Picaninny Creek translate to a baseflow recession of ~45 days (Post and Jakeman 1996), compared to 5 days during the post-harvest period in WS7 (Table 1). In addition, Picaninny Creek was burned and reseeded immediately following harvesting, whereas WS7 was treated with herbicides and left barren for 5 years following harvesting.

WHERE DO WE GO FROM HERE?

This study serves as a useful proof-of-concept to show the utility of hydrologic modeling for detecting and discerning the effects of forest management on catchment hydrology. The next step is to extend our approach to model the effects of the subsequent management activities, such as harvesting, on the lower half of the catchment (1966) and on the natural recovery period commencing in 1969. In addition, we are conducting catchment- and hillslope-scale experiments using stable isotopes to characterize residence time, partition hydrographs, and identify the sources of streamflow generation.

CONCLUSION

The change detection approach presented in this study is a useful way to advance our understanding of catchment processes and the effects of disturbance on hydrology. Results from this study corroborate the hypothesis that forest harvesting increases streamflow by decreasing the volume of precipitation lost to interception and transpiration, thereby increasing antecedent soil moisture and streamflow. Though our approach was developed and tested to evaluate the effects of disturbance in a managed forest ecosystem, we contend that our method is applicable for evaluating and forecasting the effects of other disturbances, such as insect denudation and directional climate change, on streamflow and catchment processes.

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

HARVESTING PRODUCTIVITY AND DISTURBANCE ESTIMATES OF THREE SILVICULTURAL PRESCRIPTIONS ON THE DANIEL BOONE NATIONAL FOREST, KENTUCKY

Jason D. Thompson, Bob Rummer, and Callie Schweitzer¹

Abstract.—A large-scale silvicultural assessment designed to examine the effectiveness of five treatments in reducing the potential impacts of gypsy moth infestation and oak decline was implemented on upland hardwood forests in the Daniel Boone National Forest in southeastern Kentucky. The study was authorized via the Healthy Forests Restoration Act of 2003. The goal of the treatments was to improve the health and vigor of the stands through five different prescriptions. Three of the treatments required mechanized harvesting to achieve the desired stand condition, one treatment utilized herbicides to treat the midstory vegetation, and the final treatment was an unaltered control. This paper discusses the harvesting productivity and soil disturbance associated with the three mechanized harvest treatments.

INTRODUCTION

A large-scale study was implemented to assess the effectiveness of five different silvicultural treatments in limiting the impacts of gypsy moth and oak decline on the Daniel Boone National Forest in southeastern Kentucky. Oak decline is a naturally occurring condition attributed to advanced tree age and adverse climate and site conditions. The combination of these stresses with the effects of forest pests can lead to the premature death of oak-dominated forest stands. Occurrences of oak decline have been recorded since the early 1900s throughout New England, the Middle Atlantic, and the Southeastern United States. Symptoms of oak decline include a progressive dying back from the tips of the branches and dwarfed or sparse foliage and premature autumn leaf color and leaf drop (Wargo and others 1983).

The gypsy moth is a nonnative pest first introduced to the United States in 1869. Over the past 140 years the moth has slowly spread from its introduction point in Boston, MA, to include the entire Northeast and now the Southern Appalachians. The gypsy moth feeds on tree foliage. Cyclic outbreaks of the insect can result in the defoliation of entire stands. The stresses of gypsy moth and other pests, such as shoestring root rot and the two-lined chestnut borer, in combination with oak decline, can lead to mass mortality of oak-dominant forests (Hoyle 2007).

Previous research suggests that the impact of oak decline and gypsy moth infestations can be limited by preparing the forest before stress introduction. Silvicultural treatments aimed at decreasing competition and increasing regeneration and tree vigor have been shown to lessen the impacts of oak decline and gypsy moth infestations.

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This study implements four silvicultural treatments and a control to test their effectiveness against oak decline and the gypsy moth. The study is being conducted by a multidisciplinary research team composed of U.S. Department of Agriculture Forest Service and university researchers. Research will include studies of not only the silvicultural effects, but also the effects on wildlife, soils, and selection of harvesting methods and equipment. This paper focuses on the production, efficiency, and impacts of the harvesting systems used to implement the silvicultural prescriptions.

STUDY AREAS

The study was implemented on the London Ranger District of the Daniel Boone National Forest in southeastern Kentucky. The harvesting units are located west of London in the Cold Hill area with oak- and hickory-dominated stands ranging in age from 70 to 150 years. The units are generally located on broad ridges with some moderate side slopes (up to 30 percent). Soils are highly weathered, low-fertility Ultisols. Stand density before harvest ranged from 100 to 120 ft²/ac and 140 to 160 stems/ac.

The study was designed as a randomized complete block on two site types (dry-mesic and dry-xeric) with five treatments (Table 1): shelterwood with reserves, oak shelterwood, B-line thinning, oak woodland, and a control (Schweitzer and others 2008). There are three replicates for a total of 30 units equaling almost 600 acres. Eighteen of the 30 units were mechanically harvested, six of the units were treated chemically, and the remaining six were retained undisturbed as controls. Harvesting began in May 2007 and was completed in August 2009.

METHODS

A mechanical tree-length harvesting system was used to harvest all units. The system consisted of a feller buncher, grapple skidder, and a knuckleboom loader. Trees larger than 23 inches diameter at breast height (d.b.h.) were felled with a chainsaw. All limbing and topping was performed with a chainsaw in the stand. Products removed from the units included hardwood sawtimber and biomass logs. A biomass log was any material greater than 3 inches in diameter, reasonably straight, at least 10 feet long, and not meeting merchantability specifications of a saw log.

The Forest Operations Research Unit of the U.S. Forest Service in Auburn, AL, measured the productivity and efficiency of the harvesting system and its impact on the stand. Due to the number of acres to be treated and the extended time frame taken to complete the harvesting, data recorders were used to capture gross

Table 1.—Treatments on the research study examining sustaining oak systems amid threat of gypsy moth infestation and oak decline on the Daniel Boone National Forest.

Treatment	Prescription
1. Control	No burn, no disturbance
2. Shelterwood w/ Reserves	10-15 ft ² /ac residual BA ^a , mechanical harvest
3. Oak Shelterwood	60-75 ft ² /ac residual BA, herbicide treatment
4. Thinning to B-line	Gingrich's Stocking Chart, mechanical harvest
5. Woodland Thinning	30-50 ft ² /ac residual BA, mechanical harvest

^aBA = Basal area

harvesting data. Detailed observation of the chainsaw limbing and topping component of the harvesting system was not performed. A utilization rate of 50 percent was used for the chainsaw and laborer. The goal was to measure the machine productive time required for the harvesting system to harvest each unit of each treatment. These data along with the amount of timber removed from each unit allowed for the calculation of unit productivity and an estimate of cost and efficiency.

A Yellow Activity Monitoring System (YAMS) activity recorder was fitted to each machine in the harvesting system to capture productive machine hours (PMH). The YAMS recorders are capable of recording 114 hours of productive machine time before requiring a download. Assuming 8 productive hours a day, the recorders have to be downloaded only approximately every 14 working days or every 3 weeks (Thompson 2002). The feller buncher was also equipped with a MultiDAT Jr. data recorder/global positioning system unit (Castonguay Électronique Inc., Pointe-Claire, Quebec) to provide spatial data regarding the cutting pattern and acreage cut per day. Timber volume removed from each unit was calculated from the consumer-scale load tickets which were supplied by the timber purchaser to the Forest Service as part of the timber sale agreement.

Each unit was assessed using the point transect method (McMahon 1995) to measure the site disturbance caused by the harvesting operations. Nine mutually exclusive disturbance classes were used. Each sample point was further classified by a location type (Table 2).

RESULTS AND DISCUSSION

The research study required that the harvesting be completed with a mechanical harvesting system. The harvesting equipment initially consisted of a rubber-tired feller buncher, a tracked swing-to-tree feller buncher, two grapple skidders, and a knuckleboom loader. The rubber-tired feller buncher was equipped with a shear felling head with the assumption that smaller biomass material could be felled more effectively with such a machine. After a few weeks, however, the contractor decided that it was better to do all felling with the tracked feller buncher. To balance system productivity, the rubber-tired feller buncher and one of the grapple skidders were removed from the crew. Later in 2007, the tracked feller buncher was replaced with a similar machine with a disc saw. This machine remained with the crew for the rest of the study. Adverse (wet) weather conditions resulted in the extension of the harvesting well beyond the desired completion date of the end of the second growing season (winter 2008). The wet weather and the resulting slower harvesting productivity led to the decision to add a second harvesting crew (contractor 2) in June 2008. The second harvesting

Table 2.—Disturbance classes and location types used to describe harvesting impacts.

Disturbance Class	Location
Undisturbed	Landing
Disturbed w/ litter in place	Primary skid trail
Litter removed & topsoil exposed	Secondary skid trail
Litter removed & mineral soil exposed	Stand area
Litter & soil mixed	Road
Soil exposed > 4 inches	Other (stream bed, streamside management zone)
Non-soil (e.g., stumps, rocks, logs)	
Logging slash	
Soil deposited on top of ground	

contractor was equipped similarly to the first. In early 2009, the first harvesting crew (contractor 1) stopped participating in the timber harvesting. The equipment models and operating cost for both crews are listed in Table 3. System costs were calculated using the machine rate method (Miyata 1980, Brinker and others 2002) and include ownership, operating costs, and labor costs, but do not include profit and overhead.

In an effort to complete the harvesting by the end of the second growing season, the harvesting crews were allowed to work through the normal winter shutdown period (December to April). The crews were closely monitored and not allowed to exceed site disturbance limits set by the Forest Service. Working through the winter months did speed up harvesting but included extended periods of idle time when conditions were too wet.

In 2007, when harvesting began, productivity measurement was complicated by constantly changing personnel and equipment in the harvesting system. The addition of the second harvesting contractor also complicated the process of measuring productivity by doubling the number of machines to monitor. Of the 18 units harvested, nine were completed by contractor 1, eight were completed by contractor 2, and one unit was started by contractor 1 and finished by contractor 2. Table 4 summarizes harvesting results by unit.

The shelterwood with reserves units averaged 27 acres and took an average of 10 weeks to harvest. The woodland thinning and thinning to B-line units averaged 28 and 26 acres in size and took an average of 7 and 6 weeks respectively to harvest. Figure 1 illustrates time to harvest by treatment. Woodland thinning Unit 20 took 4 months to harvest. This unit was started by contractor 1 and then later finished by contractor 2. The unit was the largest unit in the study (48 ac) and consisted of several long, thin ridge tops which contributed to the extended time to harvest.

Tons per acre removed varied, as expected, by treatment with an average of 77 tons/ac on the shelterwood units, 25 tons/ac on the B-line thinning units, and 42 tons/ac on the woodland thinning units. Table 5 shows the removal percentage for each treatment. The thinning treatment, which had the lowest tons per acre removed, also had a much lower percentage of basal area (BA) removed compared to the percentage of

Table 3.—Harvesting system machine types and estimated costs.

Machines	SMH ^a	PMH ^b
Contractor 1		
1999 John Deere 648G Grapple Skidder	\$34.65	\$53.31
1999 Timberjack 2618 Tracked Feller-Buncher	\$43.12	\$66.34
2007 John Deere 335 Knuckleboom Loader	\$48.76	\$75.01
Chainsaw	\$0.80	\$1.60
Manual Laborer (Wage + fringe benefits)	\$12.36	\$24.72
System	\$139.69	\$220.98
Contractor 2		
1998 John Deere 648G Grapple Skidder	\$34.65	\$53.31
1998 Timberjack 608 Tracked Feller-Buncher	\$43.33	\$66.67
1985 Hawk Knuckleboom Loader	\$23.56	\$36.24
Chainsaw	\$0.80	\$1.60
Manual Laborer (Wage + fringe benefits)	\$12.36	\$24.72
System	\$114.70	\$182.54

^aSMH = Scheduled Machine Hours, ^bPMH = Productive Machine Hours

Table 4.—Summary of harvesting data.

UNIT	TREATMENT ^a	CONTR. ^b	AREA (AC)	TIME (WKS)	TONS		PMH ^c	TONS/PMH
					REMOVED	TONS/AC		
2	2. Shelter	1	31	9	2,136	68.9	501	4.26
12	2. Shelter	2	16	9	1,523	95.2	458	3.33
29	2. Shelter	1	30	7	1,510	50.3	356	4.24
35	2. Shelter	1	29	12	2,865	98.8	641	4.47
16	2. Shelter	1	27	10	2,207	81.7	510	4.33
32	2. Shelter	2	27	13	1,858	68.8	572	3.25
18	4. Thin	2	21	4	526	25.1	197	2.68
21	4. Thin	1	27	4	409	15.2	193	2.12
25	4. Thin	2	17	4	540	31.8	164	3.30
33	4. Thin	1	26	5	593	22.8	312	1.90
11	4. Thin	2	21	7	532	25.4	237	2.25
17	4. Thin	2	43	10	1,283	29.8	384	3.34
4	5. WThin	1	20	5	1,079	53.9	256	4.22
19	5. WThin	2	28	6	997	35.6	258	3.86
22	5. WThin	1	27	4	907	33.6	239	3.80
23	5. WThin	2	19	5	706	37.2	228	3.10
15	5. WThin	1	28	5	1,025	36.6	222	4.61
20	5. WThin	1/2	48	16	2,653	55.3	716	3.71

^aTreatments, see Table 1. ^bContractor, see Table 3. ^cProductive Machine Hours.

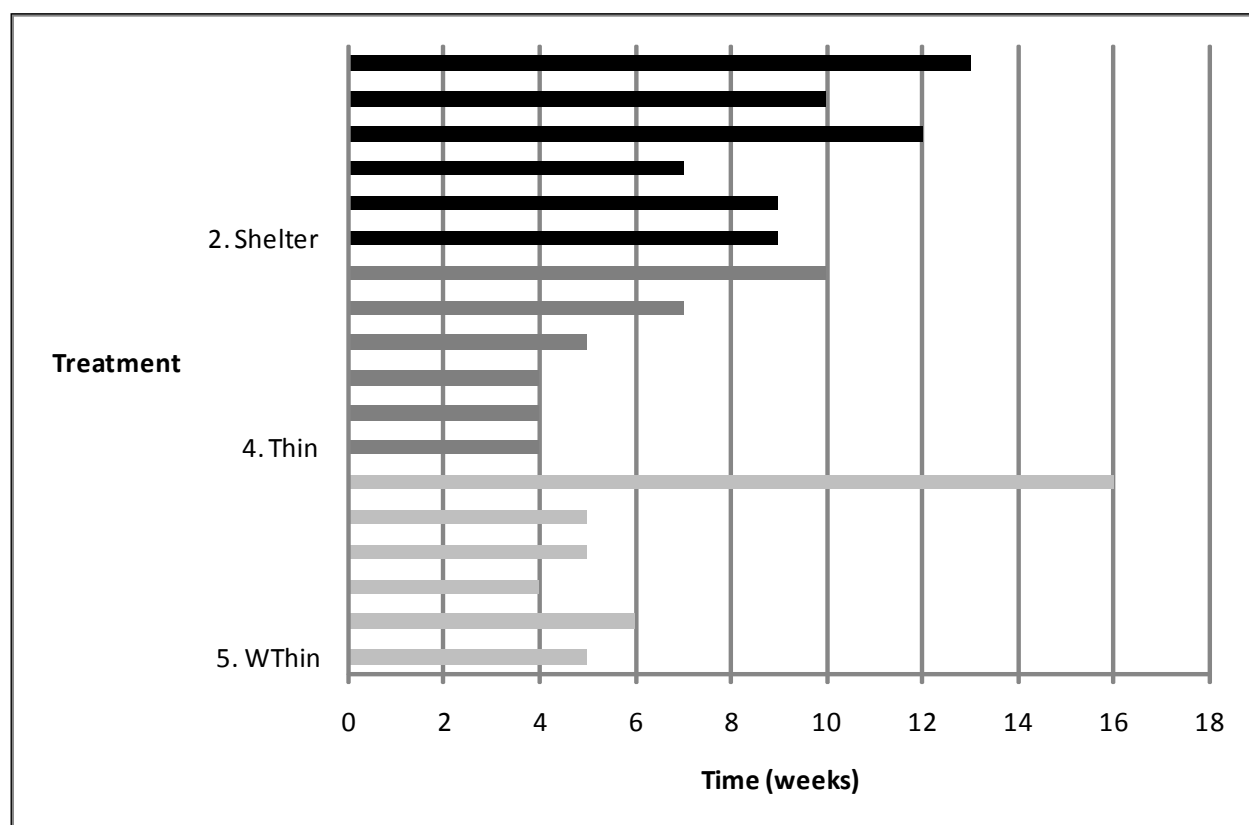
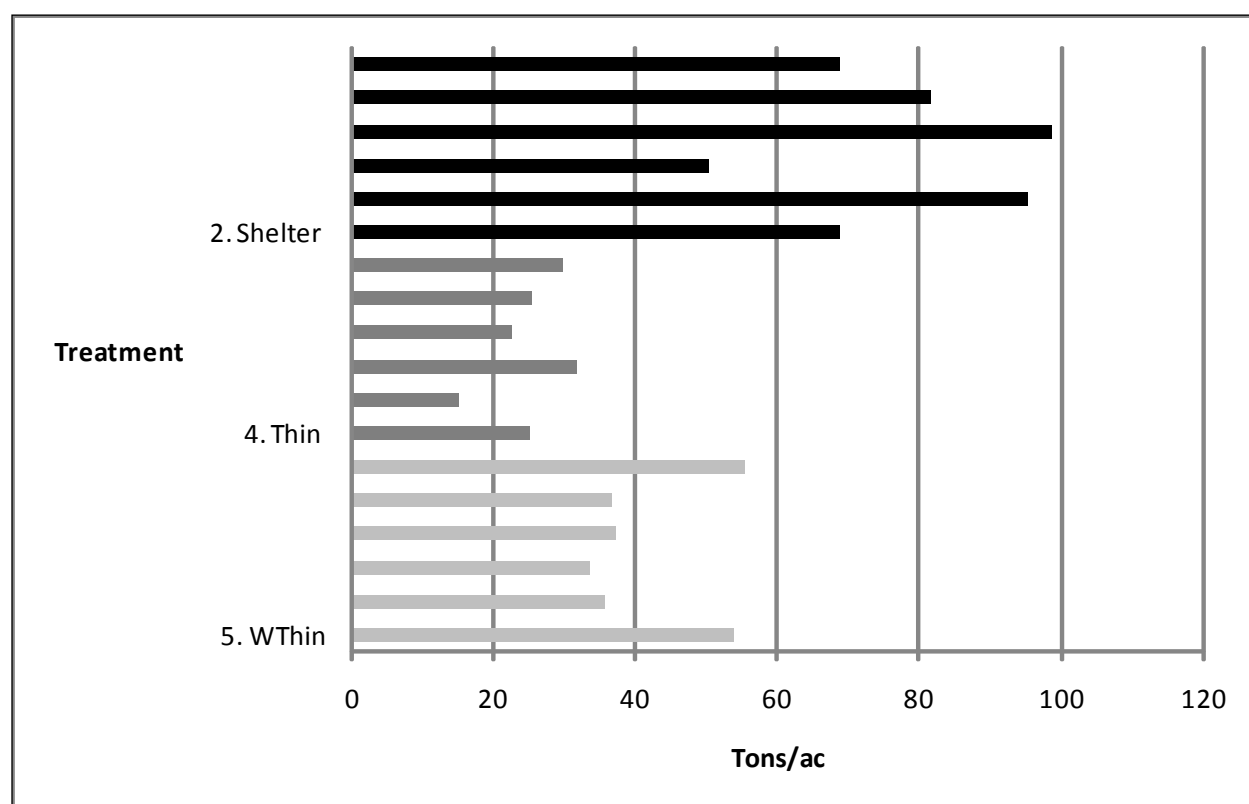


Figure 1.—Gross time to harvest units by treatment.

Table 5.—Percentages of basal area (BA), stems/acre (SPA), and biomass removed.

Treatment	Percent Removed		Percent Removed (tons)	
	BA (ft ²)	SPA	Biomass	Saw logs
2. Shelter	78	87	54	46
4. Thin	32	60	78	22
5. WThin	43	68	65	35

stems per acre (SPA) removed. The percentage of biomass tons removed was also much lower than the percentage of saw log tons removed (Fig. 2). These results indicate that more small stems were removed in the thinning treatment, resulting in lower harvesting productivity. The shelterwood treatment, on the other hand, had high percentages of both BA removed and SPA removed, indicating a heavier removal across all stem diameters. The product removal percentages were close to being equally split between biomass and saw logs. Average productivity by treatment decreased as the percent of biomass removed increased. The shelterwood treatments had the highest productivity (3.98 tons/PMH) and the least percent of biomass removed (54 percent). The thinning treatments had the lowest productivity (2.60 tons/PMH) and the highest percentage of biomass removed (78 percent).

**Figure 2.—Tons/ac removed from harvest units by treatment.**

A gross system utilization rate of 30 percent was calculated for all units. This number was calculated using the number of weeks to harvest each unit, the productive machine hours for each unit, and the assumption that a typical working year is 2000 hours. Therefore, the overall average utilization rate calculated is lower than the actual system utilization because it does not factor in days lost to weather delays and days not scheduled to work, but it still reflects a low utilization rate. An additional factor affecting system utilization rates was availability of labor. Both harvesting contractors, at times, utilized one operator to run both the feller buncher and loader, which drastically reduced the utilization of the two machines and the system. Contractor 2 operated in this manner for every unit harvested, as reflected in a lower average production rate of 3.14 tons/PMH compared to 3.77 tons/PMH for contractor 1.

Sarles and Whitenack (1984) reported system productivity of 11 to 13 tons per productive hour for a three-man chainsaw crew using a truck-mounted crane in a West Virginia hardwood stand. Long and others (2002) reported tracked feller buncher productivity of 20 to 108 tons/PMH also in West Virginia hardwood stands. Average d.b.h. was 16.1 inches and overall system productivity was not reported. Sarles and Whitenack's (1984) study shows that even a manual felling/cable yarding system is capable of producing four times the tonnage of the harvesting productivity of the current study and Long and others' study shows that a feller buncher alone is capable of producing 5 to 30 times that of the current study.

Cost per ton was also estimated for each unit. The shelterwood with reserves and woodland thinning units averaged \$53/ton, while the thinning to B-line units averaged \$80/ton. This additional \$27/ton reflects the low average harvest volume (25 tons/ac) compared to 77 tons/ac and 42 tons/ac for the shelterwood and woodland thinning treatments, respectively.

Site disturbance was sampled on all harvested units. The disturbance categories are combined into five main disturbance types in Table 6 to highlight differences among treatments. The category "Soil exposed" combines the three disturbance classes "Litter removed & topsoil exposed," "Litter removed & mineral soil exposed," and "Litter & soil mixed" from Table 2. The "Slash" disturbance class was included to illustrate the amount of biomass left in the units and combines the disturbance classes "Slash" and "Non-soil." "Non-soil" accounted for a very small percentage of data points, of which the majority was downed logs. Eighteen percent of the shelterwood with reserves units were classified as "Slash," while 9 percent and 11 percent of the thinning to B-line and woodland thinning units, respectively, were classified as "Slash." "Slash" was defined as any piled limbs and tops located at the survey point. Figure 3 averages the classes in Table 6 by treatment. "Soil Exposed" ranged from 24 percent on the thinning to B-line units to 32 percent on the shelterwood with reserves units. "Litter in Place" and "Undisturbed" combined, accounted for 49, 59, and 67 percent in shelterwood with reserves, woodland thinning, and thinning to B-line, respectively. Thinning to B-line units had the highest percent area in "Undisturbed" and "Litter in Place," which reflects the least average volume of timber removed (25 tons/ac).

The disturbance classes for location are summarized as "Stand Area" and "Skid Trails." "Skid Trails" consists of all non-"Stand Area" classes, including primary skid trails, secondary skid trails, landings, and internal roads. Shelterwood unit 35 has one of the highest percentages in skid trails (27 percent), probably a reflection of the contractor's efforts to become familiar with the prescription requirements and a high turnover in personnel. Woodland thinning Unit 4 has the highest percent of area in trails (28 percent). This unit had unusual topography that may have contributed to the disturbance value. It was a long, narrow unit aligned along

Table 6.—Summary of site disturbance sampling.

UNIT	TREATMENT ^a	DISTURBANCE TYPE (PERCENT OF TOTAL AREA)					LOCATION TYPE	
		Undisturbed	Litter in place	Slash	Soil exposed	Soil on top	Stand area	Skid trails
2	2. Shelter	18	29	15	36	2	95	5
12	2. Shelter	12	31	18	36	2	91	9
29	2. Shelter	19	33	12	36	1	91	9
35	2. Shelter	17	24	25	34	0	73	27
16	2. Shelter	12	47	16	24	0	89	11
32	2. Shelter	10	38	23	28	2	87	13
18	4. Thin	35	24	10	30	0	89	11
21	4. Thin	35	28	4	31	1	85	15
25	4. Thin	15	47	10	27	1	89	11
33	4. Thin	32	36	12	20	1	81	19
11	4. Thin	41	34	9	16	0	86	14
17	4. Thin	31	40	10	18	1	86	14
4	5. WThin	30	23	7	36	3	72	28
19	5. WThin	19	39	17	25	0	80	20
22	5. WThin	31	25	7	37	0	81	19
23	5. WThin	24	32	15	28	0	92	8
15	5. WThin	19	48	11	21	1	90	10
20	5. WThin	24	39	9	25	2	86	14

^aTreatments, see Table 1.

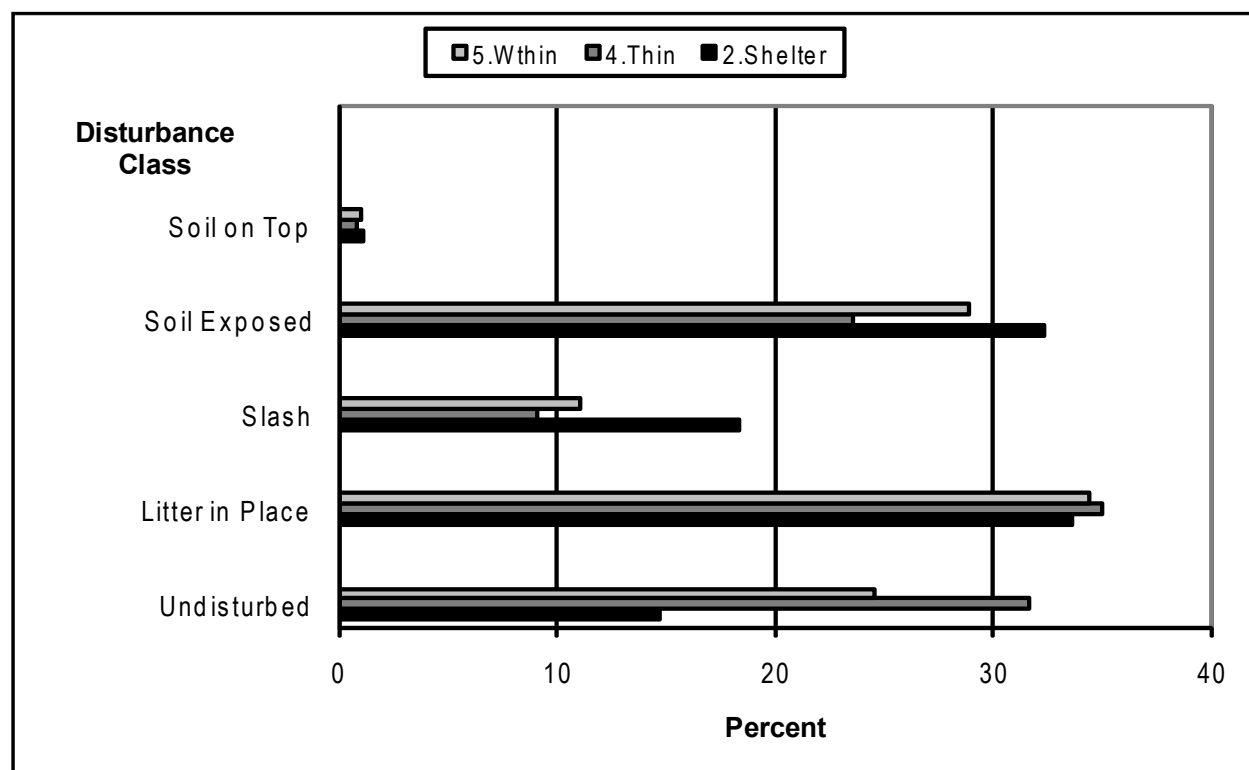


Figure 3.—Percentage of area in various disturbance classes.

a short ridge and consisted of steep slopes on both sides and along one end. The contractor was forced to winch timber from some sections of the unit. Figure 4 illustrates that, on average, all three treatments had average percent in trails between 12 percent and 16 percent.

Mitchell and Trimble (1959) reported on a manual felling and tracked dozer/cable skidding (with a wheeled arch) operation in West Virginia with percent area in skid trails ranging from 2.5 percent to 7 percent. Mitchell and Trimble (1959) also noted that a horse logging operation in West Virginia averaged 12 percent area in skid trails. Hatchell and others (1970) reported an average area of 33.8 percent occupied by primary and secondary skid trails and landings on pine harvesting operations in the Atlantic coastal plain using wheeled and tracked skidders and manual felling. Kochenderfer (1977) measured 10.3 percent of harvested area in skid trails from a manual felling wheeled cable skidding operation. These studies do not match the type of machines used in the current study on the Daniel Boone National Forest but show that a grapple skidder, tracked feller buncher system is within normal ranges for site disturbance.

CONCLUSIONS

The longevity of this study (May 2007-August 2009), the addition of a second harvesting contractor (and subsequent retraction of the first contractor), changing personnel and machines within crews, and adverse weather conditions all complicated the analysis of this study. The resulting productivity of the harvesting systems was very low. The data nevertheless indicate one potential treatment effect on harvesting productivity. The volume of small-diameter (biomass) stems removed from the thinning to B-line treatment resulted in lower productivity than measured on the other two treatments. The larger range of stem diameters and

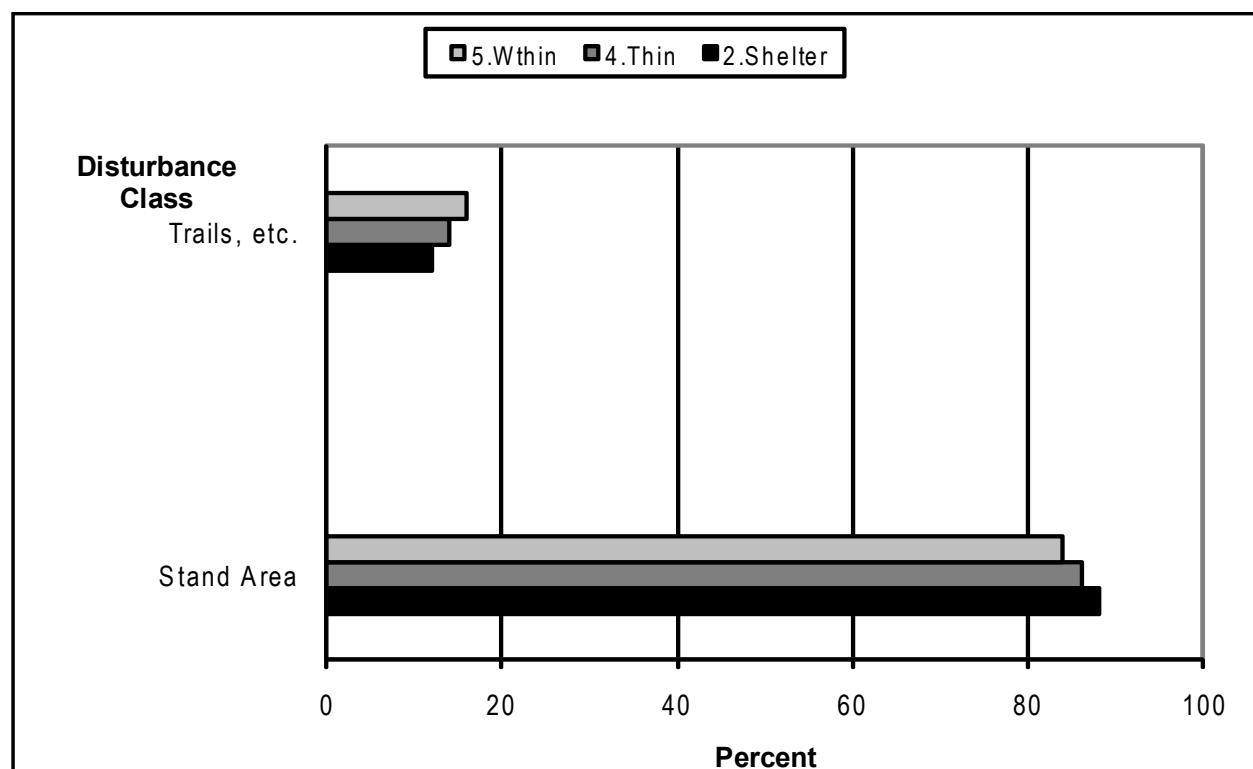


Figure 4.—Percentage of harvested area in general stand area and skid trails, landings, and roads.

number of stems removed from the shelterwood with reserves treatment led to the highest productivity of all three treatments. It is reasonable to assume that a more fully utilized and stable feller buncher, grapple skidder harvesting system could greatly out-produce the systems monitored in this study.

Soil disturbance resulting from mechanized ground-based harvesting appeared to be related to the volume removed. More removal was associated with higher levels of soil disturbance and more coverage of logging residues. The amount of stand area impacted by skid trails was relatively low, averaging between 12 percent and 16 percent of total area. This range is in line with those measured by other studies.

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

INVASIVE SPECIES

CAN OUR CHESTNUT SURVIVE ANOTHER INVASION?

Lynne K. Rieske and W. Rodney Cooper¹

Abstract.—Plant breeders and land managers have been actively pursuing development of an American chestnut with desirable silvicultural characteristics that demonstrates resistance to the chestnut blight fungus. As progress towards development of a blight-resistant chestnut continues, questions arise as to how these plants will interact with pre-existing stresses. The Asian chestnut gall wasp is an introduced invader exploiting chestnut in eastern North America and contributing to the complex of stressors facing chestnut restoration efforts. The gall wasp is a potentially devastating pest that causes globular galls on actively growing shoots of all *Castanea* species. Gallling reduces tree vigor, prevents normal shoot development, reduces or eliminates nut production, and can cause tree mortality. The persistent spread of this exotic, invasive insect threatens chestnut production and restoration efforts throughout the eastern United States. We have been characterizing associates of the Asian chestnut gall wasp in eastern North America to more fully understand gall development and what factors regulate gall wasp populations. The natural enemy complex has been characterized, and interactions between a native parasitoid and an exotic parasitoid that was introduced for Asian chestnut gall wasp control are being evaluated. We are also evaluating the extent to which surrounding vegetation influences natural enemy occurrence. Our ultimate goal is to gain an understanding of the ecological interactions, dispersal patterns, and mechanisms regulating gall wasp populations in eastern North America.

INTRODUCTION

Historically, American chestnut, *Castanea dentata* (Marsh.), was a dominant component of the northern and central hardwood forests of eastern North America, and at one time was among the most abundant tree in portions of the Appalachians (Braun 1950). However, reports of chestnut mortality attributable to black ink disease, *Phytophthora cinnamomi*, occurred as early as the mid-1800s (Crandall and others 1945), and the accidental introduction of the chestnut blight fungus, *Cryphonectria (Endothia) parasitica*, in 1904 virtually eliminated American chestnut from its former range. Nevertheless, extensive efforts at developing disease-resistant varieties have recently met with some success (Griffin 2000).

Selecting for resistant relic American chestnut, and/or hybridizing susceptible American chestnut with resistant Chinese chestnut, *C. mollissima*, followed by repeated selection of resistant progeny and backcrossing with American chestnut, produces blight-resistant products with many of the desirable growth and morphological characteristics of the American chestnut (Jaynes and Dierauf 1982, Griffin 2000). Blight-resistance is not straightforward, however, and can be adversely affected by altitude (Jaynes and Dierauf 1982), cold temperatures (Griffin and others 1993), xeric conditions (Gao and Shain 1995), and vegetative

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competition (Griffin and others 1991). Additionally, resistance to one stressing agent may affect susceptibility to a second stressor (Chapin 1991), suggesting that blight-resistance could alter chestnut's susceptibility to herbivory.

As progress towards development of a blight-resistant chestnut continues and the prospects for the return of chestnut to the eastern North American landscape become more tangible, questions arise as to how pre-existing stresses will affect these plants. Hybrids developed in response to the chestnut blight fungus have not been assessed for herbivore resistance (Rieske and others 2003, Kellogg and others 2005). Such novel herbivores as the Asian chestnut gall wasp (ACGW), *Dryocosmus kuriphilus* (Hymenoptera: Cynipidae), further threaten efforts to deploy disease-resistant chestnut in eastern North America (Payne and others 1975).

The Asian chestnut gall wasp is a potentially devastating exotic insect that causes globular twig, shoot, and leaf galls on actively growing shoots of all *Castanea* species. Gallling reduces fruiting and nut yield, suppresses shoot elongation and twig growth, reduces tree vigor and wood production, and can kill trees (Yasumatsu 1951, Payne and others 1975, Dixon and others 1986, Anagnostakis and Payne 1993, Kato and Hijii 1997). Gallling also prevents infested shoots from producing new shoot growth and reproductive flowers, thereby reducing or eliminating nut production. Chestnut production and chestnut restoration efforts throughout the eastern United States are threatened by the persistent spread of this exotic, invasive insect.

The gall wasp was initially introduced into North America in Byron, GA, in 1974 (Payne and others 1975); within 7 years of its introduction, it had devastated the fledgling chestnut industry in Georgia (Anagnostakis and Payne 1993). The gall wasp has since expanded its distribution north-northeasterly. Since its accidental introduction into North America on Chinese chestnut in 1974, ACGW has expanded its host range to include five *Castanea* species, and its geographic range to encompass 11 eastern states (Stehli 2003, Cooper and Rieske 2007, Rieske 2007, Rieske unpubl. data). Additional infestations are likely to be reported as the gall wasp continues to expand its range.

The gall wasp has one generation per year and reproduces asexually (Payne and others 1976, Payne 1978). Adult females lay eggs inside buds in early summer. Eggs hatch soon after, but larvae remain inactive. Larval growth begins at budbreak the following spring, when larvae induce rapid formation of conspicuous stem, petiole, or leaf galls. These galls provide the developing wasp protection throughout the larval and pupal stages. Adults emerge from galls in the early summer and locate new chestnut shoots to lay eggs for the next generation (Payne 1978). Once the wasps emerge, galls become woody and dry, and can remain on the tree for several years. Small leaves remain attached to the galls during the winter; these leaves are highly visible and are useful in detecting infestations.

Shortly after the gall wasp was first reported in North America, the parasitic wasps *Torymus sinensis*, *T. tubicola*, and *Megastigmus* spp., which had successfully suppressed damaging gall wasp populations in Japan (Murakami and others 1980, 1994; Moriya and others 1989, 2003), were introduced into gall wasp-infested orchards in Byron, GA (Payne 1978). Within a short period, gall wasp populations in central Georgia declined and the incidence of galling dropped to acceptable levels. *Torymus sinensis* has moved to a limited extent with expanding gall wasp populations in eastern North America, and several native parasitoids, apparently recruited from native oak gallmakers, have adapted to exploit the gall wasp (Cooper and Rieske 2007). *T. sinensis* and the native *Ormyrus labotus* occur in frequencies great enough to influence gall wasp

populations, but the extent to which these natural enemies are able to keep up with expanding gall wasp populations is unknown (Cooper and Rieske 2007, 2009). Several additional parasitoid species have been identified, as well as a potentially pathogenic fungus that causes external lesions that may cause mortality and aid in regulating gall wasp populations (Cooper and Rieske 2007). Interestingly, galling by the nonnative gall wasp may facilitate populations of the native lesser chestnut weevil, *Curculio sayi* (Coleoptera: Curculionidae), through an enhanced resource base, as weevils utilize galls as a resource (Cooper and Rieske 2007).

As we document the expansion of the gall wasp's range in eastern North America, we have been investigating evolving ecological associations, population regulators, and impacts. We have been working to characterize associates of the gall wasp in its invaded North American range to more fully understand what factors regulate gall wasp populations. Here we report on relative abundance of and interactions between the native parasitoid, *O. labotus*, and the introduced parasitoid, *T. sinensis*, assess their effects on populations of the Asian chestnut gall wasp, and evaluate how surrounding vegetation may affect natural enemy occurrence and activity. Our specific objectives were to: 1) evaluate natural enemy recruitment as the ACGW expands its geographic range; 2) determine the primary mortality factor(s) for ACGW in North America; and 3) evaluate the extent to which surrounding vegetation influences natural enemies.

METHODS

We collected galls from four distinct geographic locations: naturally occurring American chestnuts in a forest (Bowling Green, KY), Chinese chestnut, *C. mollissima*, in a landscape setting (Broadview Heights, OH), Chinese chestnut and hybrids in an orchard setting (Hiram, OH), and American and Chinese chestnut and their associated hybrids in a plantation (Meadowview, VA). At each collection site, surrounding vegetation was characterized to evaluate the extent to which associated vegetation can influence natural enemy occurrence and gall wasp mortality. Vegetation data were collected in accordance with the protocols of the U.S. Department of Agriculture (USDA), Forest Service's Natural Resource Information System and its Field Sampled Vegetation Module (FSVeg) and associated Common Stand Exam procedures (USDA Forest Service 2003). The height, diameter, and basal area of all woody vegetation were measured. Canopy structure, ground surface cover, slope, slope position, elevation, and aspect were assessed in each plot. In each 0.04-ha plot, all large trees (≥ 12.7 cm diameter at 1.5 m high; d.b.h.) were measured. Each whole plot contained five 0.004-ha subplots in which saplings (< 12.7 cm d.b.h.) and larger shrubs (≥ 137 cm height) were measured, as well as five 0.0004-ha subplots for seedlings, smaller shrubs (< 137 cm height), and woody vines. The 0.0004-ha subplots were nested within the 0.004-ha subplots, which were located at the center point of each whole plot and 7.8 m from the center in each cardinal direction (Coleman and others 2008). Collection sites were then characterized as "landscape," containing few *Castanea* of the same species and age within a diverse landscape; "orchard," containing few *Castanea* species or varieties and of relatively uniform age; "plantation," containing several to many *Castanea* species and their hybrids, of mixed age; and finally, "natural forest," containing lingering American chestnut with an abundant oak component and of mixed age.

Galls were collected approximately monthly April-August to evaluate wasp development and gall associates, with a single January collection date. In Meadowview (plantation), two shoots per tree, each containing 20-30 galls per tree, were collected from five American and five Chinese chestnut trees. In Bowling Green ("natural forest"), three shoots, each containing 20-30 galls, were collected from relic American chestnut infected with the blight fungus. The Bowling Green site consisted of only a single infested tree, so collections were replicated by shoot. In Hiram (orchard), two shoots per tree, each containing 20-30 galls per tree, were

collected from five Chinese chestnut trees. In Broadview Heights (“landscape”), three shoots per tree, each with approximately 20-30 galls, were collected from three Chinese chestnut trees. Shoots were clipped at the second apical growth scar to ensure only 2 years of growth. Clipped shoots were bagged and kept cool during transport to the laboratory, where current growth was separated from the previous years’ growth to delineate gall age. The fresh weight and volume of collected galls were measured. A subsample (10 percent) was dissected; the number of chambers per gall was counted and the thickness of the protective schlerenchyma layer surrounding larval chambers was measured. The contents of each were identified as gall former, parasitoid, fungus, or empty chamber.

Larval hymenopteran parasitoids are morphologically indistinguishable, so we used a molecular approach to identify the two most numerous parasitoids of the gall wasp in North America, the native *O. labotus*, and the introduced *T. sinensis* (Cooper and Rieske 2007). DNA was extracted from parasitoid larvae, and DNA primers were used to amplify the ribosomal ITS2 region for each wasp. Species-specific size variation in ITS2 PCR products was determined by direct sequencing of DNA extracted from known *D. kuriphilus*, *O. labotus*, and *T. sinensis* adults collected and identified previously (Cooper 2007). Each unknown larval parasitoid was identified by molecular comparison with adults of known specimens.

RESULTS

Our collections confirmed that parasitism accounts for the greatest amount of gall wasp mortality (Table 1) (Cooper and Rieske 2007). Using a molecular approach, we were able to identify parasitoids of the ACGW that were previously indistinguishable. The most abundant parasitoid species present was *T. sinensis*, which was introduced into the United States in the mid-1970s for gall wasp suppression (Payne 1978). *T. sinensis* was present at each collection site, but was most abundant at orchard and plantation settings, which had the greatest relative density of chestnut. The native parasitoid *O. labotus*, which historically utilizes oak gallmakers, was most abundant within chestnut galls in a forested setting containing a significant oak component. Presumably the oaks provide an abundance of closely related gallmaking hosts that historically sustain *O. labotus* populations (Cooper and Rieske 2007, 2009). The majority of nearctic *Ormyrus* are oak-associates (Hanson 1992).

Fungi within the gall interior also caused mortality (Table 1), as did failure of adults to emerge for undetermined reasons. Additionally, feeding by the lesser chestnut weevil and black lesions on the gall exterior

Table 1.—Incidence of factors influencing populations of the Asian chestnut gall wasp, April-August, across four sites in eastern North America.

Site	Setting	Chestnut present	Parasitization (%)		Interior fungus ^a	Weevil feeding ^b	Exterior lesions ^c
			<i>T. sinensis</i> (introduced)	<i>O. labotus</i> (native)			
Meadowview, VA	Plantation	Am., Chi., hybrids	65.5	25.3	3.6	0.3	0.17
Bowling Green, KY	Forest	American	19.3	50.5	1.6	0.04	0.10
Broadview Hts., OH	Landscape	Chinese	95	0	2.1	0.12	0.27
Hiram, OH	Orchard	Chinese, hybrids	90	10	1.0	0	0.11

^aPercent chambers affected

^bAverage number of feeding scars per gall

^cAverage number of fungal lesions per gall

caused by an unidentified fungus, were positively correlated with gall wasp mortality (Table 1). In spite of their co-occurrence with gall mortality, we cannot be certain whether this relationship is causative or simply correlative.

DISCUSSION

We evaluated ecological interactions and mortality agents impacting populations of the ACGW as it expands its geographic range in eastern North America. Parasitization accounted for the majority of gall wasp mortality; we found relatively high rates at all our collection sites. The prevalence of *T. sinensis* suggests that this introduced parasitoid is moving with expanding gall wasp populations in eastern North America, likely in infested plant material. *T. sinensis* is univoltine; adults emerge in early spring and parasitize newly developing galls. *T. sinensis* occurs in galls throughout the year and overwinters in dead galls as pupae and larvae (Payne 1978, 1981; Cooper and Rieske 2009). It is a generalist parasitoid of cynipid gall wasps in China, but acts as a specialist on Asian chestnut gall wasp in Japan, and possibly North America, where *D. kuriphilus* galls are abundant in surviving chestnut (Stone and others 2002). This synchrony with the ACGW life cycle contributes to the effectiveness of *T. sinensis* as a parasitoid of the gall wasp, and may explain why *T. sinensis* appears to act as a specialist on the Asian chestnut gall wasp in North America.

The second most abundant parasitoid, *O. labotus*, is a native species previously confined to oak gallmakers (Cooper and Rieske 2007). Our results suggest that this parasitoid is effectively making a host jump from oak gallmakers to utilize the novel resource provided by the chestnut gallmaker. *O. labotus* produces multiple generations per year and exploits more hosts, including cynipid gall wasps and cecidomyiid gall midges (flies), than do other known *Ormyrus* species (Hanson 1992). *O. labotus* was present in chestnut galls only in late spring and early summer, indicating that this native parasitoid requires additional gallmaking hosts for subsequent generations through the remaining year (Cooper and Rieske 2009).

The Asian chestnut gall wasp is a relatively new host for *O. labotus* (Cooper and Rieske 2007). Our data suggest an evolving competitive interaction between the native *O. labotus* and the introduced *T. sinensis*; this evolving competitive interaction can be influenced by habitat type. *T. sinensis* is the primary parasitoid where chestnut is numerically dominant, whereas *T. sinensis* and *O. labotus* are both common in the more vegetatively diverse study sites. *O. labotus* appears to utilize the chestnut gall wasp and its introduced parasitoid, *T. sinensis*, as hosts within chestnut galls. This finding is consistent with our previous work suggesting potential antagonistic interactions between the native and introduced parasitoids (Cooper and Rieske 2007).

Our results provide important insight into ecological interactions between two important parasitoids of *D. kuriphilus*, and provide evidence for a native parasitoid's host shift to an invasive gall wasp and its introduced parasitoid. Collectively, this study improves our understanding of interactions between native and introduced gall parasitoids and could lead to improved management of *D. kuriphilus* in forest chestnut plantings and cultivated chestnut.

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PREDICTING THE SPATIAL DISTRIBUTION OF *LONICERA JAPONICA*, BASED ON SPECIES OCCURRENCE DATA FROM TWO WATERSHEDS IN WESTERN KENTUCKY AND TENNESSEE

Dongjiao Liu, Hao Jiang, Robin Zhang, and Kate S. He¹

Abstract.—The spatial distribution of most invasive plants is poorly documented and studied. This project examined and compared the spatial distribution of a successful invasive plant, Japanese honeysuckle (*Lonicera japonica*), in two similar-sized but ecologically distinct watersheds in western Kentucky (Ledbetter Creek) and western Tennessee (Panther Creek). The occurrence data of Japanese honeysuckle and seven environmental variables were collected and measured from 533 random plots at the two watersheds. A spatial logistic regression model was developed to identify the factors that contribute most to the spread of this invasive plant. Our results showed that the spatial distribution of this invasive plant was different at the two watersheds. The Ledbetter Creek watershed, which had heavier anthropogenic disturbances, had a much more frequent occurrence of Japanese honeysuckle than did the forested Panther Creek Watershed. The spatial regression model indicated that elevation was significantly correlated with the spatial distribution of Japanese honeysuckle at the Ledbetter Creek Watershed. In contrast, distance from the main road, soil moisture, light intensity, and plant species richness of each plot were all significant factors in relation to the spatial distribution of invasive species at the Panther Creek Watershed.

INTRODUCTION

Exotic plant species are also called alien plants, non-indigenous plants, nonnative plants, and introduced plants in the literature. Invasions by exotic plants can change the niches of native species in the communities, alter the structure and function of ecosystems, and disrupt the evolutionary processes (D'Antonio and Vitousek 1992, Mack and others 2000, Levine and others 2003). Invasive plants have invaded approximately 700,000 ha of natural communities in the United States, at an annual expense of more than \$137 billion (Pimentel and others 2000).

Studies that emphasize specific sites or habitat types have provided a general model of the distribution of invasive plants and have measured their impacts on natural communities. Additional studies have shown that forested regions have been impacted to a lesser extent than other communities (Mooney and others 1986). Management and control of biological invasion have been a great challenge for researchers and the general public. The geography of native habitat, including the availability of water sources, elevation, slope and aspect, and human disturbances are important factors in the process of plant invasion (Gillham and others 2004). The spatial distribution of invasive plants, however, is poorly documented (Dark 2004). We know the general regions where an invasive plant may be found, but detailed maps are usually unavailable.

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Several recent studies have demonstrated the importance of using statistical techniques that incorporate space on some level when assessing species distribution. For instance, Nielsen and others (2008) found that using logistic regression models and incorporating a spatial element into the model output improved their ability to predict the distribution and impact of invasive plant species.

The objectives of this project are: (1) to provide the first geographic characterization of the invasive plant, Japanese honeysuckle (*Lonicera japonica* [Thunb.]), in two distinct watersheds; and (2) to develop test models to identify factors contributing most to the spread of this invader. We hypothesized that the spatial distribution of Japanese honeysuckle is site-specific and that intensive anthropogenic disturbance causes greater invasion.

STUDY AREA

A watershed provides a natural unit for the study of an ecosystem. This study compared the spatial distribution and pattern of Japanese honeysuckle in two watersheds, Ledbetter Creek, in western Kentucky, and Panther Creek, in Tennessee, utilizing geospatial techniques. These two watersheds are similar in size and are roughly 30 km apart (Fig. 1). Both streams empty into the Kentucky Lake Reservoir. Ledbetter Creek is located on the west side of Kentucky Lake. Its watershed contains forest, cropland, grassland, and small communities that cover an area of about 24 km². The recently opened highway KY80 cuts through the watershed on the north, which will likely bring more disturbance and change to the area. Panther Creek, an almost completely forested watershed, is located on the east side of the Kentucky Lake Reservoir within the Land Between the Lakes National Recreational Area. Since human disturbance is considered a major factor in plant invasion (Alston and Richardson 2006), we expected to report considerable differences in plant invasion between the agriculture-based Ledbetter Creek drainage area and the forested Panther Creek watershed.

METHODS

SPECIES SELECTION

Japanese honeysuckle, a perennial woody vine, was selected as the target invasive species for this study for the following three reasons. (1) Japanese honeysuckle has been a highly successful invader since its introduction to the United States from eastern Asia in the 1800s as an ornamental and forage plant. It is very common in the southeastern states and is now found in 45 states (Miller 2003). (2) Japanese honeysuckle has had an enormous negative impact on native ecosystems (Dillenburg and others 1993a,b). (3) Very little information on the spatial distribution of Japanese honeysuckle has been collected in the study area.

FIELD DATA COLLECTION

Field data collection was conducted between June and December 2008. To avoid pseudo-replication in the data collection, 63 random sampling points were assigned to two watersheds based on the aerial maps. We established 32 sampling points at the Ledbetter Creek Watershed and 31 sampling points at the Panther Creek Watershed. Each sampling point contained 10 to 20 plots, each 1-m x 1-m in size. The plots were distributed approximately 15 m apart along the transect lines. Each transect length was determined by measuring the distance from the sampling point to the main road. We collected data from 210 plots within the Ledbetter Creek Watershed and 323 plots within the Panther Creek Watershed. We measured the following six variables, which are most likely to constrain the geographic distribution of the species: species richness per plot, distance from the main road, elevation, soil moisture, soil pH, and light intensity. Light intensity measures

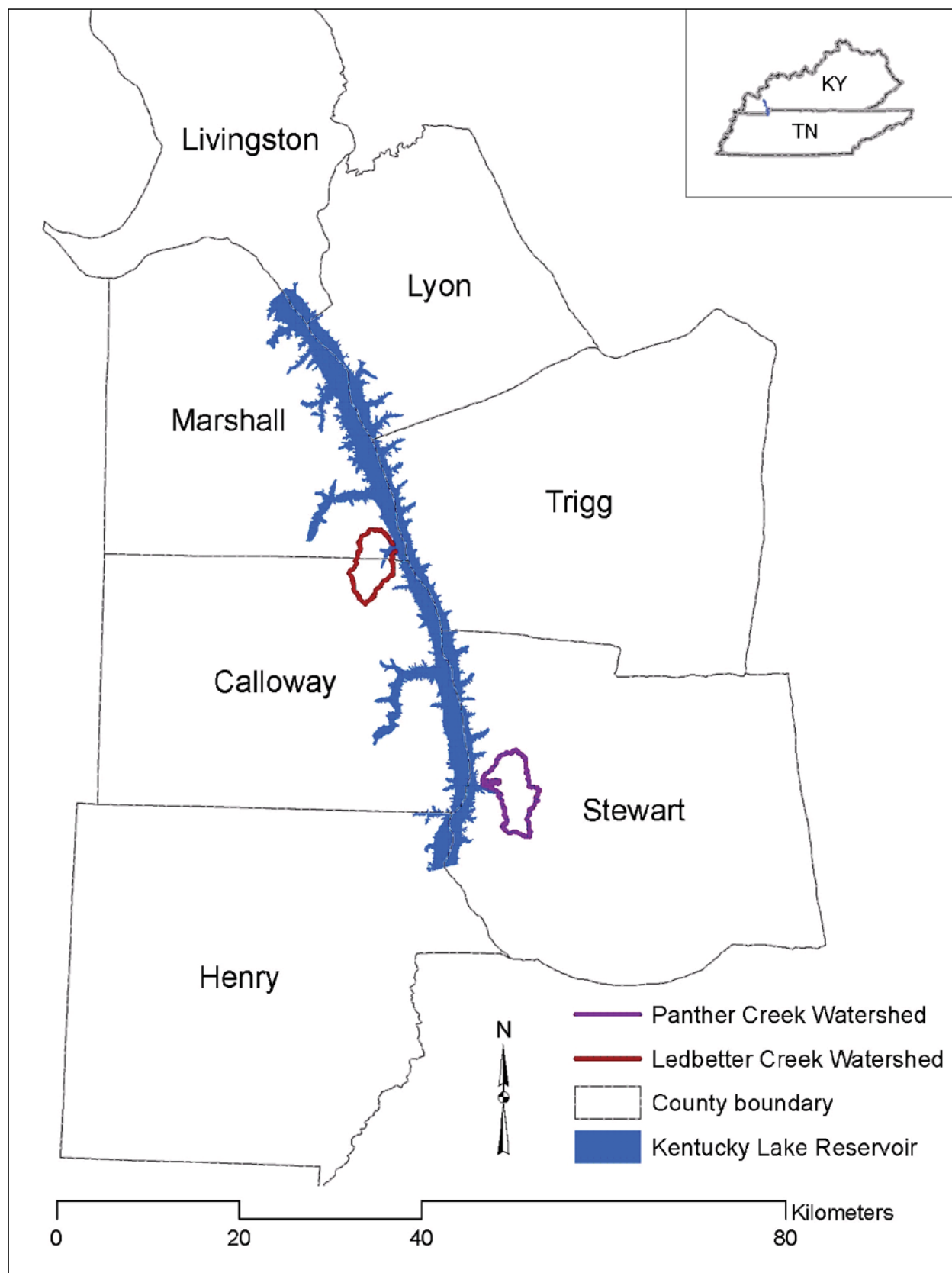


Figure 1.—The study area, showing the location and boundaries of each watershed.

were made early in the morning and late in the afternoon with an EA 30 Light Meter (Extech Instruments Corp., Waltham, MA). Distance from the main road was measured with a meter tape, and elevation data were obtained with a Garmin GPS 76 Global Positioning System (Garmin, Olathe, KS). Soil moisture and soil pH measurements were determined by a Kelway Soil Meter (Kel Instruments Co., Inc., Wyckoff, NJ). Last, Japanese honeysuckle occurrence data (1-presence, 0-absence) were recorded at each plot of both watersheds. Occurrence data were used to compare the distribution of Japanese honeysuckle within the two watersheds.

STATISTICAL ANALYSES

Logistic regression models were developed at plot level to identify contributing factors in the region using SYSTAT 12.0 (SYSTAT Software, Inc., Chicago, IL). We used logistic regression models because of the robust nature of the models with binary data (Smith 1994) and their good performance in predicting the distribution of invasives at different spatial scales (Evangelista and others 2008, Nielsen and others 2008). The data for light intensity and distance to the main road were log-transformed before the analyses. All other residuals were normally distributed, so no transformations were necessary. To build the landscape-level predictive models, we extracted five explanatory variables from a GIS database, Kentucky Lake Geographic Information System (MARC Associates 1998) using ArcMap (ESRI, Redlands, CA). These variables were distance from the main road, elevation, depth of water table, slope, and aspect. The occurrence data (1, 0) of Japanese honeysuckle at the landscape level were determined based on the plot-level occurrence data. We recorded “1” if Japanese honeysuckle covered more than 50 percent of the plot, and “0” if it covered less than 50 percent of the plot.

RESULTS

The occurrence of Japanese honeysuckle clearly differed between the watersheds. The invasive plant was found on 87.1 percent of the plots in Ledbetter Creek Watershed ($n = 183$, based on a total of 210 plots). In contrast, Japanese honeysuckle occurred on only 30.7 percent of the plots in the Panther Creek Watershed ($n = 100$, based on 323 plots).

Factors contributing to occurrence also differed significantly between the two watersheds. At the plot level, the results of logistic regression models indicated that elevation was the only significant factor ($p < 0.05$) at Ledbetter Creek Watershed (Table 1). For Panther Creek Watershed, the significant factors determined by the logistic regression model were distance from the main road, soil moisture, light intensity, and species richness (Table 2).

The geographic factors that may contribute to the potential spread of Japanese honeysuckle differed as well between the two watersheds. At the landscape level, the logistic model indicated that slope was the only marginally significant factor at Ledbetter Creek (Table 3). Elevation and soil water depth were significantly correlated with the spatial distribution of this invasive species at the Panther Creek Watershed (Table 4).

DISCUSSION

Specific habitat type and intensive anthropogenic disturbance have been hypothesized to account for the different spatial distribution of Japanese honeysuckle between two distinct watersheds. We predicted that the geographic conditions of habitats and heavy anthropogenic disturbance could cause a greater occurrence of Japanese honeysuckle because both factors have been found to play a significant role in plant invasion (Gillham and others 2004, Alston and Richardson 2006). The results of our study showed that the occurrence

Table 1.—Results of logistic regression model at Ledbetter Creek at plot level.

Parameter	Estimate	Standard Error	Z	p-value	95% Confidence Interval	
					Lower	Upper
Constant	-38.300	15.644	-2.448	0.014	-68.961	-7.639
Distance to road	0.180	0.126	1.427	0.154	-0.067	0.426
Elevation	4.477	1.804	2.482	0.013	0.942	8.012
Soil pH	1.792	1.366	1.312	0.190	-0.886	4.470
Soil moisture	-0.002	0.016	-0.152	0.880	-0.034	0.029
Light intensity	-0.010	0.009	-1.078	0.281	-0.028	0.008
Richness	-0.107	0.133	-0.802	0.422	-0.368	0.154

Table 2.—Results of logistic regression model at Panther Creek at plot level.

Parameter	Estimate	Standard Error	Z	p-value	95% Confidence Interval	
					Lower	Upper
Constant	-3.204	9.159	-0.350	0.726	-21.155	14.747
Distance to road	-0.360	0.107	-3.376	0.001	-0.569	-0.151
Elevation	-0.085	0.626	-0.136	0.892	-1.313	1.143
Soil pH	0.412	1.106	0.372	0.710	-1.756	2.579
Soil moisture	0.062	0.014	4.467	0.000	0.035	0.090
Light intensity	-0.073	0.028	-2.652	0.008	-0.127	-0.019
Richness	0.536	0.124	4.313	0.000	0.293	0.780

Table 3.—Results of logistic regression model at Ledbetter Creek at landscape level.

Parameter	Estimate	Standard Error	Z	p-value	95% Confidence Interval	
					Lower	Upper
Constant	-96.822	51.003	-1.898	0.058	-196.786	3.141
Elevation	18.758	10.331	1.816	0.069	-1.490	39.007
Aspect	0.190	0.462	0.411	0.681	-0.716	1.095
Slope	0.120	0.064	1.877	0.060	-0.005	0.244
Distance to road	0.074	0.376	0.198	0.843	-0.663	0.812
Water depth	0.132	0.297	0.446	0.656	-0.450	0.714

Table 4.—Results of logistic regression model at Panther Creek at landscape level.

Parameter	Estimate	Standard Error	Z	p-value	95% Confidence Interval	
					Lower	Upper
Constant	76.678	32.284	2.368	0.018	13.207	140.150
Elevation	-14.635	6.535	-2.239	0.025	-27.444	-1.826
Aspect	0.063	0.441	0.142	0.887	-0.802	0.927
Slope	-0.110	0.061	-1.792	0.073	-0.229	0.010
Distance to road	0.522	0.389	1.341	0.180	-0.241	1.284
Water depth	-0.641	0.301	-2.132	0.033	-1.231	-0.052

of Japanese honeysuckle is heavily associated with anthropogenic disturbances in general. However, it provided little evidence that the distribution of Japanese honeysuckle is determined by habitat conditions when there is prevalent anthropogenic disturbance. The degree of disturbance better accounts for the observed differences in the distribution of Japanese honeysuckle between the two watersheds.

We concluded that if a watershed is associated with heavy human interference, then the factors that usually determine the habitat conditions, such as soil moisture, light intensity, and species richness do not play a significant role in relation to the spatial spread of invasive species. The predominant influence of human activity is apparent in the Ledbetter Creek Watershed, where there is ongoing human interference from farming, habitation, and frequent traffic. On the contrary, if the watershed had less exposure to human activities, then factors such as distance from the main road, soil moisture, light intensity, and species richness are all significantly related to the spatial distribution of the invasive species. The results from the Panther Creek drainage area, an almost completely forested watershed, are in perfect agreement with our conclusions. Furthermore, our modeling results indicate that invasion risk is much higher in areas with constant anthropogenic disturbance.

Our study provides the first geographic characterization of Japanese honeysuckle in the region. Moreover, we identified factors associated with successful invasion and evaluated the future risk of invasion at both watersheds. The results of this study should be very useful when recommendations for future land-use practices and conservation policies are considered. This study approach also may be applicable to the study of invasive plants in other watersheds or other geographic regions.

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A REVIEW OF IMPACTS BY INVASIVE EXOTIC PLANTS ON FOREST ECOSYSTEM SERVICES

Kevin Devine and Songlin Fei¹

Abstract.—Many of our forest ecosystems are at risk due to the invasion of exotic invasive plant species. Invasive plant species pose numerous threats to ecosystems by decreasing biodiversity, deteriorating ecosystem processes, and degrading ecosystem services. Literature on Kentucky's most invasive exotic plant species was examined to understand their potential impacts on natural and semi-natural systems. This review suggests that many of the targeted species possess the ability to alter ecosystem processes and functions. However, the impacts on these services are still emerging, are often vague, and lack empirical data. Thus, further research on species-level impacts and the mechanisms facilitating the impacts of exotic plants on ecosystem services is sorely needed.

INTRODUCTION

Ecosystem services, defined as the complex interplay of conditions and natural cycles through which ecosystems sustain and fulfill humanity (Daily and others 2001), are essential to human well-being. Forest ecosystems play a vital role in performing a suite of processes and functions that produce resources for consumption and other uses. For example, forests play a critical role in delivering drinking water to more than 180 million people in the United States and support nutrient exchanges necessary for biological communities (Smail and Lewis 2009). Forests provide the pathways for water capture, transport, and evaporation as it circulates throughout various systems at multiple scales (Hornberger and others 1998). Forest-derived processes also serve integral functions in determining plant and animal community compositions by influencing light levels, soil conditions, nutrient availability, and physical structures (Boettcher and Kalisz 1990, Vitousek and others 1997, Smail and Lewis 2009). Furthermore, forests play globally important roles in influencing other large-scale ecosystem patterns and processes such as the carbon and nutrient cycles (Bowman and others 2009).

Infiltration of invasive exotic species (IES) into forested environments may disrupt forest ecosystem processes and functions by displacing native species (Collier and others 2002, Huebner 2003). In addition, species that become community dominants may alter environmental conditions and resource availability of market and non-market products and services (Gordon 1998). This disruption is accomplished when IES differ from native biota in specific species traits such as nutrient cycling, chemistry, and phenology (Ehrenfeld 2003). However, not all invading species are equal in their ability to alter larger-scale processes. It seems reasonable that IES with the ability to become community dominants will contribute the greatest alterations to existing ecosystem services (Gordon 1998). In view of the multitude of IES entering forest ecosystems, identifying which species pose the greatest threats is important when considering effective management schemes.

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Incorporating the ecosystem services approach into conservation management is becoming increasingly prevalent through better understanding of the connections between ecosystems and their associated products and processes (Patterson and Coelho 2009). While much effort has gone into the quantitative valuation of these goods and services, knowledge gaps persist in understanding how IES impact ecosystem services (Pejchar and Mooney 2009). A comprehensive analysis of existing literature may provide insight into how much we understand the impacts caused by IES on ecosystem services in eastern forests. Such an exercise could help ecologists, social scientists, and natural resource managers to discover hidden taxes on ecosystem services and could assist in developing effective IES management strategies (Webster and others 2006, Pejchar and Mooney 2009). In this paper we examined the extent and prevalence of the impact of IES on ecosystem services.

METHODS

To accomplish this goal, we synthesized peer-reviewed data on IES impacts on three targeted ecosystem services (regeneration, nutrient dynamics, and hydrology) and provided examples from the literature for extended discussion. Due to the large number of species that exist in the eastern forests, a comprehensive review of all IES requires tremendous time and effort. Thus we reviewed manuscripts concerning the 26 most detrimental IES, as identified by the Kentucky Exotic Pest Plant Council (KY-EPPC) (Table 1). These species are listed as the most severe threat to Kentucky because they possess the characteristics of invasive species and displace native vegetation. Many of these 26 species are documented as invasive in states containing central hardwood forests. Other species, such as *Lonicera japonica*, *L. maackii*, and *Alliaria petiolata* have become highly invasive in both disturbed areas and closed-canopy forests throughout much of the United States (Nuzzo 1999). We conducted literature searches on all 26 species to examine evidence for their ability or potential to alter ecosystem services. Examples of other species impacting targeted forest ecosystem services throughout different regions of the world were also provided.

Table 1.—Most severe invasive plants in Kentucky and their documented impacts on ecosystem processes.

Scientific Name	Common Name	Invasive Species Effects on Ecosystem Processes		
		Regeneration	Nutrient Dynamics	Hydrology
<i>Ailanthus altissima</i>	Tree-of-heaven	(Huebner 2003); (Gomez-Aparicio and Canham 2008)	(Gomez-Aparicio and Canham 2008)	--
<i>Alliaria petiolata</i>	Garlic mustard	(Hochstedler and others 2007; Meekins and McCarthy 1999; Stinson and others 2006)	(Rodgers and others 2008)	--
<i>Carduus nutans</i>	Musk thistle	(Floyd and others 2006)	--	--
<i>Celastrus orbiculata</i>	Oriental bittersweet	--	--	--
<i>Conium maculatum</i>	Poison hemlock	--	--	--
<i>Coronilla varia</i>	Crown vetch	(Brown 1995)	--	--

(Table 1 continued on next page)

(Table 1 continued)

Scientific Name	Common Name	Invasive Species Effects on Ecosystem Processes		
		Regeneration	Nutrient Dynamics	Hydrology
<i>Dioscorea oppositifolia</i>	Chinese yam	--	--	--
<i>Elaeagnus umbellata</i>	Autumn olive	(Orr and others 2005)	(Goldstein and others 2009; Orr and others 2005)	--
<i>Euonymus alatus</i>	Burning bush	--	--	--
<i>Euonymus fortunei</i>	Winter creeper	--	--	--
<i>Festuca arundinacea</i>	Kentucky 31 fescue	(Spyreas and others 2001; Tunnell and others 2004)	--	(Self-Davis and others 2003)
<i>Lespedeza cuneata</i>	Sericea lespedeza	(Brandon and others 2004)	--	--
<i>Ligustrum sinense</i>	Privet	(Morris and others 2002)	--	--
<i>Lonicera japonica</i>	Japanese honeysuckle	(Dillenburg and others 1993; Yurkonis and Meiners 2004)	--	--
<i>Lonicera maackii</i>	Bush honeysuckle	(Cipollini and Dorning 2008; Flory and Clay 2009; Hutchinson and Vankat 1997; Meiners 2007)	(Collier and others 2002; Demars and Boerner 1997)	--
<i>Lythrum salicaria</i>	Purple loosestrife	(Blossey and others 2001)	(Gutknecht and others 2006)	--
<i>Melilotus alba</i>	White sweet clover	(Wolf and others 2004)	(Wolf and others 2004)	--
<i>Melilotus officinalis</i>	Yellow sweet clover	(Van Riper and Larson 2009)	--	--
<i>Microstegium vimineum</i>	Japanese grass	(Adams and Engelhardt 2009; Baiser and others 2008; Horton and Neufeld 1998; Oswalt and others 2007)	(Ehrenfeld and others 2001; McGrath and Binkley 2009)	--
<i>Miscanthus sinensis</i>	Chinese silver grass	--	--	--
<i>Phragmites australis</i>	Common reed	--	--	--
<i>Polygonum cuspidatum</i>	Japanese knotweed	--	--	--
<i>Pueraria lobata</i>	Kudzu	--	--	--
<i>Rosa multiflora</i>	Multiflora rose	--	--	--
<i>Sorghum halapense</i>	Johnson grass	--	--	--
<i>Stellaria media</i>	Chickweed	--	--	--

RESULTS

REGENERATION

Sustainable timber production in the eastern United States depends upon continued regeneration of high-value tree species. Survival of tree seedlings is a necessary component because current seedling stock must mature to economically viable timber. Many native woody species are dependent upon mutualistic relationships with native plants, animals, and fungi for their growth and survival (Callaway and others 2008). Natural regeneration regimes are known to be altered by invading plant species that utilize various mechanisms to dominate understory environments and outcompete desired species for available resources (Hutchinson and Vankat 1997, Stinson and others 2006). Moreover, invading species may change disturbance regimes or resource supply rates, resulting in large, even cascading impacts (Adams and Engelhardt 2009).

Overall, 50 percent of the species listed as a severe threat by the KY EPPC have been documented to threaten regeneration through direct resource competition, altering resource availability, or a suite of other mechanisms (Table 1). For example, *Rosa multiflora* and *L. maackii* were shown to invade forest edges, openings, and disturbed understories by outcompeting native woody species, resulting in simplified biological communities (Meiners 2007). *L. maackii* was proven to have influenced landscape-level species composition by invading along forest-edge environments throughout a wide range of light environments (Hutchinson and Vankat 1998). In another study, results suggest that *Elaeagnus umbellata* utilizes allelopathy as a mechanism to impact the regeneration of several common successional species such as *Acer saccharinum*, *Populus deltoides*, and *Platanus occidentalis* (Orr and others 2005). This experiment showed the emergence of all three native tree species was suppressed when aqueous extracts derived from soil, leaf litter, and live leaves of *E. umbellata* were introduced. In Southern forests, *Microstegium vimineum* was shown to reduce native woody species regeneration by invading disturbed environments in moist forests (Oswalt and others 2007). In this study, regression models showed *M. vimineum* percent cover had a strong negative relationship with native woody species density. This species was shown to successfully invade areas with both low and high light conditions (Horton and Neufeld 1998) and responded positively to several harvest treatments (Oswalt and others 2007), making it a direct threat to woody species regeneration throughout the central hardwoods region and beyond.

Successful invasion into mesic temperate forests also appears to be facilitated by disequilibrium in native communities or the ability of invasive plants to disrupt existing ecological associations among native species (Stinson and others 2006). Most vascular plants form mycorrhizal associations with arbuscular mycorrhizal fungi (AMF), on which they are highly dependent (Smith and Read 2008). Invasive plants not dependent on such associations can invade sites and degrade AMF densities in the soil. *A. petiolata*, a nonmycorrhizal plant, was shown to suppress the growth of native tree seedlings by impairing native mycorrhizal mutualisms (Smith and Read 2008). This particular study provided empirical evidence that dominant native hardwood species, *A. saccharum*, *A. rubrum*, and *Faxinus americana* showed reduced growth and survival rates when grown in soils where *A. petiolata* had invaded.

While these examples illustrate that invasive plants can inhibit regeneration of several native woody species, knowledge gaps exist for many invasive exotic plants regarding their influence on regeneration processes. Research is needed on the following questions: 1) Which mechanisms yield the greatest alteration to community structure and regeneration regimes; 2) Which species exhibit what type of mechanism; and 3) Which species are capable of causing a large-scale ecosystem change?

NUTRIENT DYNAMICS

In total, only 25 percent of the species have been documented to alter nutrient dynamics (Table 1). Nutrient availability to primary producers is a key ecosystem function which appears to be sensitive to changes in the composition of plant communities (Mack and D'Antonio 2003). Invading species possessing specific traits capable of altering soil nitrogen (N) pools, as well as a suite of other soil conditions, are likely to influence soil processes (Corbin and D'Antonio 2004). Alterations to nutrient dynamics within soils may further drive species composition changes and affect net primary production, producing dramatic changes in biodiversity. Adverse ecological impacts on nutrient dynamics arise when IES occupy large areas or occur in high densities. Certain traits of exotic plants may cause alterations to the soil-based community or ecosystem-level processes by augmenting atmospheric N fixation or increasing rates of N losses (Corbin and D'Antonio 2004). Such soil alterations may favor fast-growing invasive species throughout a variety of environments. For instance, soils directly beneath *M. vimineum* were shown to contain elevated levels of pH and nitrification rates when compared to soils beneath the native *Vaccinium pallidum* (Ehrenfeld and others 2001). In this study, leaf litter from *M. vimineum* was shown to decompose at a rate different from that of surrounding native species, causing reduced N immobilization. *M. vimineum* populations were also shown to utilize low light levels to enhance carbon gain while combining small biomasses of N-rich roots and N-poor litter to change soil surface conditions (Horton and Neufeld 1998). Rodgers and others (2008) also suggested IES enhance invasibility by creating positive feedback cycles through soil alterations. By comparing various species compositions and soil horizon developments at sites in five northeastern forests, Rodgers and others (2008) reported *A. petiolata* consistently and significantly elevated N, phosphorus (P), calcium, and magnesium availability, and soil pH, in invaded plots. These apparent traits may be an important competitive advantage within undisturbed communities throughout deciduous forests, enabling ecosystem-level modifications in community structure.

Nitrogen-fixing is another trait that allows IES to invade and potentially alter ecosystem processes and services. The ability of *E. umbellate* to produce nonpoint source pollution by enhancing stream nitrate levels was documented by Goldstein and others (2009). This particular study examined the relationship between *E. umbellate* cover in a watershed and stream quality. *E. umbellate* is an N-fixing invasive shrub that has become naturalized in the eastern United States. Percentage of *E. umbellate* cover is positively correlated with mean stream nitrate concentrations. *E. umbellate* reached a critical density threshold in a watershed in southern Illinois, allowing it to produce watershed-scale impacts. This examination provides supporting evidence that ecosystem-level impacts from invasive exotic plant species are possible if species are present in critical abundance. Wolf and others (2004) observed lower nitrogen availability and mineralization in soil patches invaded by the N-fixing exotic *Melilotus alba* in montane grasslands in Colorado. Although this study was not conducted in eastern forests, these findings illustrate the potential impacts of N-fixing exotics.

Other studies provide additional support to the few examples identified for the targeted species in this study. For example, a study in New Zealand suggested that novel interactions between an invasive shrub, *Coriaria arborea*, and soil symbionts in floodplain systems produced soil alterations favorable to invaders and simplified communities (Bellingham and others 2009). The invasive shrubs were shown to alter soil N:P ratios, favoring invasion and limiting P available to native species. In another example, Jordan and others (2008) observed how three species of invasive grasses—*Euphorbia esula*, *Bromus inermis*, and *Agropyron cristatum*—modified soil microbiota by further invasion by conspecifics in the Great Plains region of North America.

Concern is growing over the impacts caused by IES on nutrient dynamics in soils. These examples offer

insight into how IES alter nutrient dynamics important for ecosystem function. Given the challenges that IES pose to ecosystems, it is important to investigate: 1) Which IES are capable of producing ecosystem-level changes in nutrient dynamics; 2) How long do these impacts last; and 3) will soil alterations be sustained even after the removal of IES?

HYDROLOGY

Forest hydrology cycles produce many diverse goods and services through a combination of multiple complex inputs and conditions (Hornberger and others 1998). This supporting ecosystem service often has no direct market value yet is vital to ecosystems worldwide (Wilson and Carpenter 1999). Examples of native plant species that alter hydrology through changes in evaporation rates (Breshears and others 1998), surface runoff (Lacey and others 1989), and water table levels (Jackson and others 2000) exist throughout the literature.

However, the existing studies that documented the impact of IES on hydrology are very limited. Out of our 26 targeted species, only *Festuca arundinacea* was documented for its impact on hydrologic systems. In this study *F. arundinacea* was compared with four other species and was shown to be more effective at reducing runoff volumes and increasing infiltration. Investigations into how IES may impact hydrology are uncommon, and those that do exist are mostly focused on areas dryer than eastern forests. For example, catchments invaded by *Eucalyptus* species in South Africa recorded flow reductions up to 22 percent, severely impacting downstream industries dependent on this resource (Le Maitre and others 2002). *Eucalyptus* tree species have higher net evaporation rates than natives in this region and extract more groundwater. Quantifying the water loss to South African industries justified spending millions of dollars on invasive management controls (Le Maitre and others 2002).

Since invasive herbaceous and woody plants often have different maximum rooting depths, they may influence underground water resources differently (Jackson and others 2000). Herbaceous plants generally have much shallower rooting depths than shrubs or trees in eastern forests (Canadell and others 1996). Therefore, herbaceous species invading forests could potentially have an effect on forest hydrological regimes if such species become prominent over large areas. Gaining specific knowledge of whether Kentucky's most severe invasive plant species can alter water resources will help to quantify their impacts on this resource. As invasive species expand in abundance and distribution, this information may become increasingly important in light of increases in future water resource demands.

Given the abundant stream channels and adjacent riparian environments found throughout the central hardwoods region, understanding what impacts IES may exert on these systems is crucial. Natural riparian zones are characterized as diverse, dynamic, and complex biological interfaces (Mulla 2003). They serve a myriad of roles as ecosystem regulators by controlling energy and material fluxes and by maintaining critical habitat (Naiman and Decamps 1997). Furthermore, riparian zones may be sensitive to environmental and biological change. While none of the targeted species are documented to disrupt these systems, they could potentially alter riparian processes and subsequent riparian functions. Several knowledge gaps need to be addressed regarding IES impacts on hydrology: 1) Which IES exhibit significant differences in water consumption; 2) Do IES with different water utilization rates produce measurable alterations to hydrologic regimes; and 3) Do IES alter riparian processes and functions?

DISCUSSION

Although numerous studies have attributed exotic plant invasions to losses in community structure and dominance, fewer report alterations to key ecosystem functions and processes. Identifying the mechanisms from which these impacts arise provides information for predicting which species are most likely to exert large impacts (Wilgen and others 1996, Levine and others 2003). Overall, our findings suggest insufficient data were available for many invasive species important to the central hardwoods region. Invasions to forest ecosystems may directly or indirectly alter the ability of forests to carry out important functions, leading to changes in plant community composition and structure that ultimately alter or disrupt ecosystem services.

This review revealed a suite of mechanisms by which invasive plants degrade ecosystem services by altering biotic and abiotic conditions. Forests in eastern North America are being rapidly invaded by myriad IES, which disrupt native understory and canopy species. Without clear empirical tests of the importance of their effects, the evidence for justifying management control efforts on many species may be insufficient. Further investigation into the mechanisms and densities necessary for ecosystem-level change is critical to understanding the impacts on ecosystem services and producing justifiable IES management schemes.

Invasive exotic species have been shown to alter, disrupt, and degrade many ecosystem services throughout the eastern United States and across the world. By invading intact forests, riparian zones, or disturbed habitats, they have the ability to alter the environmental and biological conditions around them, often leading to further invasions. As consumption of natural resources continues to rise, understanding how to minimize the expanding impacts of IES on resource development will be critical. Resources such as timber, water, food, and fibers are dependent on the natural conditions where they are produced.

Research assessing the impacts of IES on ecosystem services is still emerging and our understanding of the impacts to some ecosystem services is not fully developed. Further research on predicting factors that influence invasion, comparing invader and native traits, and evaluating environmental impacts is needed (Pejchar and Mooney 2009). Moreover, research is needed to assess large-scale replacement of plant functional types (e.g., herbaceous versus woody, N-fixing versus non-fixing species), which has the potential to alter regeneration, soil nutrient dynamics, and hydrology in forest ecosystems.

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PREPARING FOR HEMLOCK WOOLLY ADELGID IN OHIO: COMMUNITIES ASSOCIATED WITH HEMLOCK-DOMINATED RAVINES OF OHIO'S UNGLACIATED ALLEGHENY PLATEAU

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Abstract.—Hemlock woolly adelgid (HWA) is an invasive insect causing widespread mortality in eastern hemlock (*Tsuga canadensis* [L.] Carr; hereafter “hemlock”) throughout eastern forests. Hemlock is a foundation species, regulating ecosystem structure and function (e.g., microclimate, nutrient cycling). Across the central and southern Appalachians, hemlock tends to dominate ravine and riparian forests; thus its loss will have dramatic effects across ecosystem boundaries. Prior to invasion by HWA, we are collecting data on hemlock in southeastern Ohio. We found hemlock associated with short, steep slopes regardless of aspect. Hemlock is dominant at lower slope positions adjacent to streams, where few other woody species are found in either the overstory or sapling layers. At upper slope positions, increases in diversity indices are small. Nonmetric multi-dimensional scaling analyses indicate additional species (deciduous hardwoods) form predictable associations along environmental gradients within ravines. Thus, at upper slope positions, communities of oaks and other deciduous hardwoods similar to other ecosystems of the region may develop should widespread hemlock mortality occur. At the same time, riparian forests will experience nearly complete shifts in species composition and, thus, function. Hemlock mortality remains a serious challenge for the Central Hardwoods region, where hemlock-dominated ravine systems are important recreation and tourism sites.

INTRODUCTION

Over the past two centuries, the forests of eastern North America have changed dramatically as introduced pests and pathogens have reshaped forest ecosystem structure and function (Lovett and others 2006). Some of the species lost were considered foundation species (Ellison and others 2005), as they were the dominant species in an ecosystem in terms of both abundance and influence. The loss of such species has likely had significant ramifications for the stability of ecosystem processes. In particular, foundation species are thought to structure exchanges of energy through a limited number of strong interactions (Ellison and others 2005). American chestnut (*Castanea dentata* [Marsh. Borkh.]) once filled such a foundational role across more than a million square hectares of eastern forests, but was eliminated by chestnut blight (*Cryphonectria parasitica* [Murr.] Barr) in the early 20th century (Anagnostakis 1987). As chestnut declined, eastern hemlock (*Tsuga canadensis* [L.] Carr; hereafter “hemlock”) replaced it as the foundation species in many of these forest ecosystems, particularly in riparian communities of the central and southern Appalachians (Elliott and Swank 2008). Yet hemlock may soon also be eliminated as another introduced pest, the hemlock woolly adelgid (*Adelges tsugae*; HWA), is causing widespread hemlock mortality throughout an expanding portion of its range.

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The full impact of hemlock mortality is not completely understood, but initial studies indicate forest dynamics including species composition, diversity, and nutrient and energy exchanges will be affected. Hemlock stands support a unique suite of species, and therefore contribute to landscape or beta and gamma diversity (Snyder and others 2002, Tingley and others 2002, Ross and others 2003). Changes following hemlock mortality will likely be dependent on the post-hemlock community dynamics and therefore may be region-specific (Orwig and Foster 1998, Jenkins and others 1999, Kizlinski and others 2002, Eschtruth and others 2006, Stadler and others 2006, Ford and Vose 2007, Nuckolls and others 2008, Orwig and others 2008). Whereas most efforts are currently focused on finding biological controls for the current and future spread of HWA along the invasion front (Onken and Reardon 2008), data are needed to initiate restoration and management planning strategies for forests impacted by HWA. To inform these plans, we are collecting baseline data on these hemlock forest ecosystems in southeastern Ohio prior to HWA invasion.

The objective of this investigation was to document the current community composition in southeastern Ohio hemlock ravine forest ecosystems. These data would allow us to test the following hypotheses:

1. Hemlock is particularly dominant immediately adjacent to small streams, which indicates hemlock mortality will have a direct impact across forest and stream ecosystems.
2. Species that co-occur with hemlock are structured along physiographic gradients; thus post-hemlock forest dynamics will differ across these gradients.

STUDY AREAS

Study areas are located within the unglaciated Allegheny Plateau physiographic province of southeastern Ohio (Brockman 1998) and characterized by sandstone bedrock that forms deep valleys and cliffs. This region supports the majority of the hemlock stands within Ohio (Black and Mack 1976, Prasad and others 2007). Our study areas are part of the Ironton and Shawnee-Mississippian Plateau physiographic subsections (Brockman 1998). We sampled three sites within Lake Katharine State Nature Preserve in Jackson County, a 817-ha preserve with limited areas of hiking trails. In Hocking County, five sample sites were located within Sheick Hollow State Nature Preserve and the Hocking State Forest. Sheick Hollow is a 61-ha preserve adjacent to the state forest and accessible by permit only, while Hocking State Forest contains 3,924 ha managed for multiple uses, although hemlock timber is not extracted. Hocking and Jackson Counties have continental climates with cold, snowy winters (0 °C average) and warm, humid summers (21.6 °C average) (Kerr 1983, Lemaster and Gilmore 1989). Across both counties, an average rainfall of approximately 102 cm is distributed evenly throughout the year. Hocking County is characterized by sedimentary bedrock of the Mississippian and Pennsylvanian Systems. In the Mississippian system, the Logan formation of sandstone, shale, and conglomerate overlays the Cuyahoga formation of Cuyahoga Shale and Blackhand Sandstone (Lemaster and Gilmore 1989). The bedrock of Jackson County is similarly porous, formed by Sharon conglomerate and Pottsville sandstone (Beatley 1959, Runkle and Whitney 1987). Soils in the upland and slope portions of Lake Katharine are well drained hapludults, mainly the Clymer silt loam formed from sandstone residuum and Rigley sandy loam formed by colluvium at the base of slopes. Orville fluvaquents occur in the floodplains of some small coves (Kerr 1983, Runkle and Whitney 1987). In Hocking County, Lemaster and Gilmore (1989) describe the predominant soils of Hocking State Forest as part of the Cedar Falls-rock outcrop complex with 40- to 70-percent slopes. Cedar Falls is the predominant soil series, a steep and well drained soil. All sites were selected in valley/ravine riparian areas within second-growth forests with little evidence of recent human disturbance and a dominance of hemlock (approximately 50 percent or greater of the total overstory basal area).

METHODS

FIELD METHODS

At each of the eight study sites, we established three transects parallel to the stream at 10, 30, and 50 m from the stream bank. The side of the ravine chosen for sampling was determined haphazardly, in some cases based on logistical considerations due to the location of cliffs. In each transect, we used a series of five 100-m² circular plots (5.62-m radius) for a total of 15 plots per study site. Within each circular plot, we recorded basic physiographical data, slope percent (using a clinometer), slope shape, slope position, and aspect. All species and diameter at breast height (d.b.h) of the woody vegetation (all stems >2.5 cm d.b.h.) were recorded. Stems between 2.5 cm and 10 cm d.b.h. were classified as saplings, while those >10.0 cm were classified as overstory. The five subplots along each transect were combined for data analyses.

STATISTICAL ANALYSES

All statistical analyses were performed using R version 2.10.0 (online documentation available at <http://www.r-project.org/>). For the overstory layer (>10 cm d.b.h.), analyses are based on relative basal area; for the sapling (2.5-10 cm d.b.h.) layer, stem counts are used. We calculated metrics of heat load and direct incident radiation for each plots using latitude, slope percent, and aspect in Eqn. 1 from McCune and Keon (2002).

To understand the change in community characteristics across our transects going upslope from the stream, we compared measures of species richness (S), Shannon's diversity (H'), and Pielou's evenness (J') using an analysis of variance. Prior to analyses, we ensured that each metric (S, H', and J') fit the model assumptions, using a Shapiro-Wilk test for normality and Levene's test for homogeneity of variance. Skewness and kurtosis were examined with the D'Agostino and Bonett-Seier tests, respectively. Analyses of variance were then used for all three metrics (S, H', J') and significant results were examined in detail with Tukey's honestly significant difference. These metrics were calculated once for the overstory and once for the sapling layers.

To examine the environmental gradients structuring hemlock ravine communities in Ohio, we used multivariate techniques. To extract measures of community composition, we subjected the overstory and sapling layers to nonmetric multidimensional scaling analysis (NMDS). For the purposes of analysis, species appearing on fewer than 5 percent of subplots were excluded (six subplots out of a total of 120). NMDS is an unconstrained ordination method regarded as highly robust for community data (Minchin 1987). We used the metaMDS procedure with a Bray-Curtis distance matrix using the vegan package for community ecology (Dixon and Palmer 2003, <http://finzi.psych.upenn.edu/R/library/vegan/html/metaMDS.html>). The metaMDS procedure automatically transforms data with large ranges using square root and Wisconsin double standardization. The metaMDS procedure uses an iterative approach with random starts to insure against entrapment by local minima while converging on a solution that minimizes stress. The procedure also rotates the final solution so that the first axis explains the most variance. To examine associations with specific factors, environmental vectors determined by principal components analysis (PCA) were overlaid on the community NMDS diagram using the envfit procedure, also part of vegan (Dixon and Palmer 2003, <http://finzi.psych.upenn.edu/R/library/vegan/html/envfit.html>). We allowed NMDS iterations to continue until the convergence on a global solution. Categorical variables including slope shape and slope position were coded as dummy variables.

To examine community structure along physiographic gradients without the dominant influence of hemlock, we constructed an additional NMDS. Relative basal area was re-calculated without hemlock; then an NMDS solution was calculated with an overlay of environmental PCA axes as above. This analysis is intended only to examine gradients that may have been obscured by the presence of hemlock. It is not intended as a simulation of hemlock mortality, as modeling this complex disturbance is beyond the scope of this investigation.

RESULTS

Comparisons of species richness and diversity indicated that hemlock is dominant throughout our study ravine and riparian areas (Fig. 1). In the overstory, species richness measures indicate that hemlock is particularly dominant at lower slope positions, where few other species are found. However, hemlock seems to exert a greater influence than we expected at upslope positions. Although additional species occur, evenness remains low and exhibits a slightly decreasing trend upslope as additional infrequent species occur. Thus, upslope gains in Shannon's diversity are minimal.

In the overstory, NMDS axis one was associated with slope shape, slope position, heat load, and the middle (30-m) transect (Fig. 2). Axis two was more strongly associated with the lower (10-m) and upper (50-m) transects, slope percent, and incident radiation. The ordination diagram places hemlock near the origin of the environmental gradients, suggesting a ubiquitous distribution of this foundation species in steep ravines. Lower slope positions are characterized by species such as sweet birch (*Betula lenta* L.), American beech (*Fagus grandifolia* Ehrh.), and yellow-poplar (*Liriodendron tulipifera* L.). Moving up the slope, species including sugar

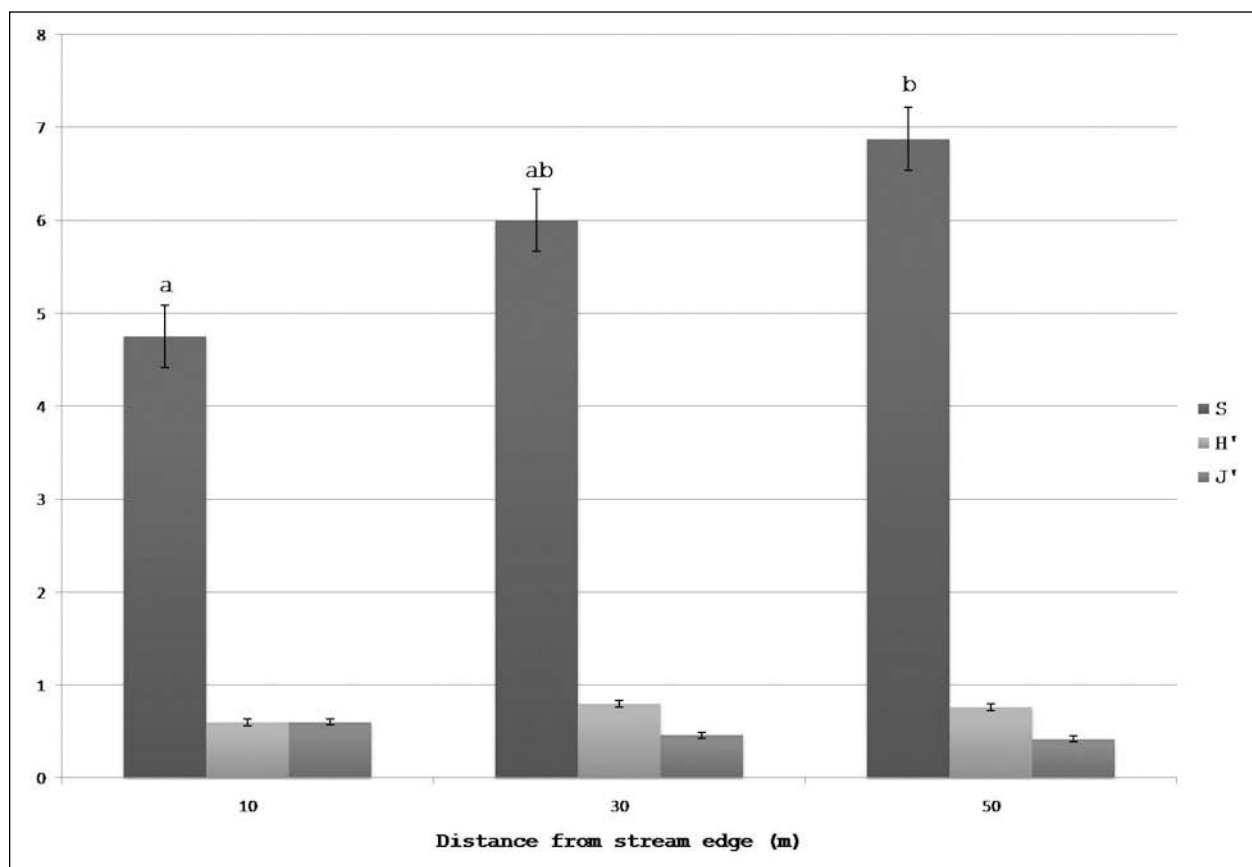


Figure 1.—Mean (± 1 SE) overstory species diversity. S refers to species richness, H' to Shannon diversity, J' to Pielou's evenness. Only species richness was determined to be different at $\alpha = 0.05$. Letters represent Tukey's HSD separation of the means.

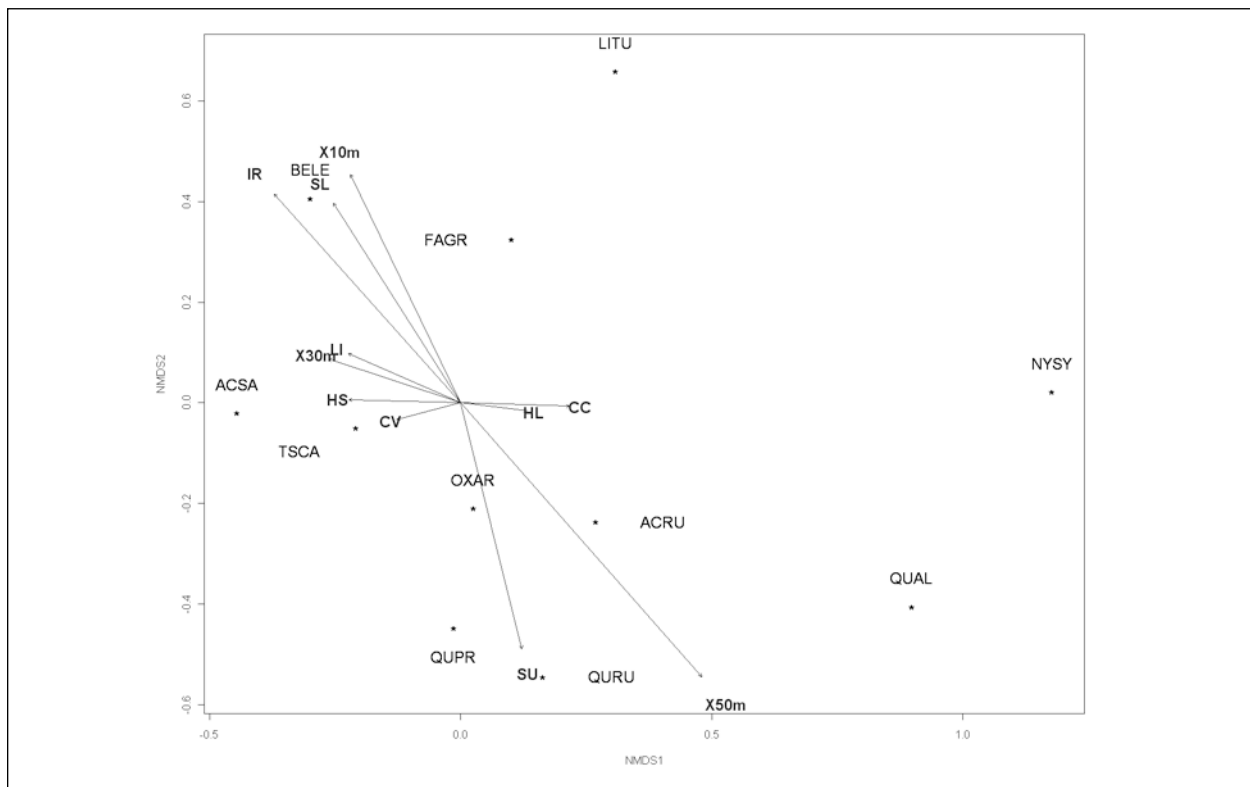


Figure 2.—Nonmetric multidimensional scaling (NMDS) ordination for overstory species with environmental principal components axes. The distance matrix for the NMDS is Bray-Curtis, based on relative basal area. Species are coded as follows: ACRU – red maple, ACSA – sugar maple, BELE – sweet birch, FAGR – American beech, LITU – yellow-poplar, NYSY – blackgum, OXAR – sourwood, QUAL – white oak, QUPR – chestnut oak, QURU – red oak, TSCA – hemlock.

maple (*Acer saccharum* Marsh.), red maple (*A. rubrum* L.), and white oak (*Quercus alba* L.) become more abundant, with red oak (*Q. rubra* L.) and chestnut oak (*Q. prinus* L.) on summits.

When the overstory was reanalyzed without the dominant influence of hemlock, environmental gradients exerted a stronger influence on community composition (Fig. 3). In this case, NMDS axis one was driven by the upper and lower transects and incident radiation. Heat loading and the 30-m distance had a greater influence on the second axis. As in the previous ordination, sweet birch and yellow-poplar were associated with lower and steeper slopes, while white and red oaks and blackgum (*Nyssa sylvatica* Marsh.) were found at upper slope positions. In this analysis, American beech and chestnut oak were separated from other species, but were not strongly associated with the environmental vectors. These two species may be influenced by metrics not included, such as soil chemistry or moisture, which are known to be influential in the region (Runkle and Whitney 1987, Hutchinson and others 1999, Small and McCarthy 2005).

The sapling layer was species-poor, and some sample transects did not have any non-hemlock saplings. As species occurred infrequently in the sapling layer, we completed an NMDS with all species to examine possible environmental gradients (Fig. 4). Species including sourwood (*Oxydendrum arboreum* [L.] DC.), umbrella magnolia (*Magnolia tripetala* L.), blackgum, red maple, and black walnut (*Juglans nigra* L.) grouped on convex slopes, associated with axis one. As in the overstory layer, hemlock's ubiquity was apparent from its position near the origin. Slippery elm (*Ulmus rubra* Muh.), hornbeam (*Carpinus caroliniana* Walter), and sugar maple grouped together on summits and convex slopes. The short environmental vectors indicate sapling distribution may be driven by a factor not included in the analysis, or there were too few saplings to indicate a clear pattern.

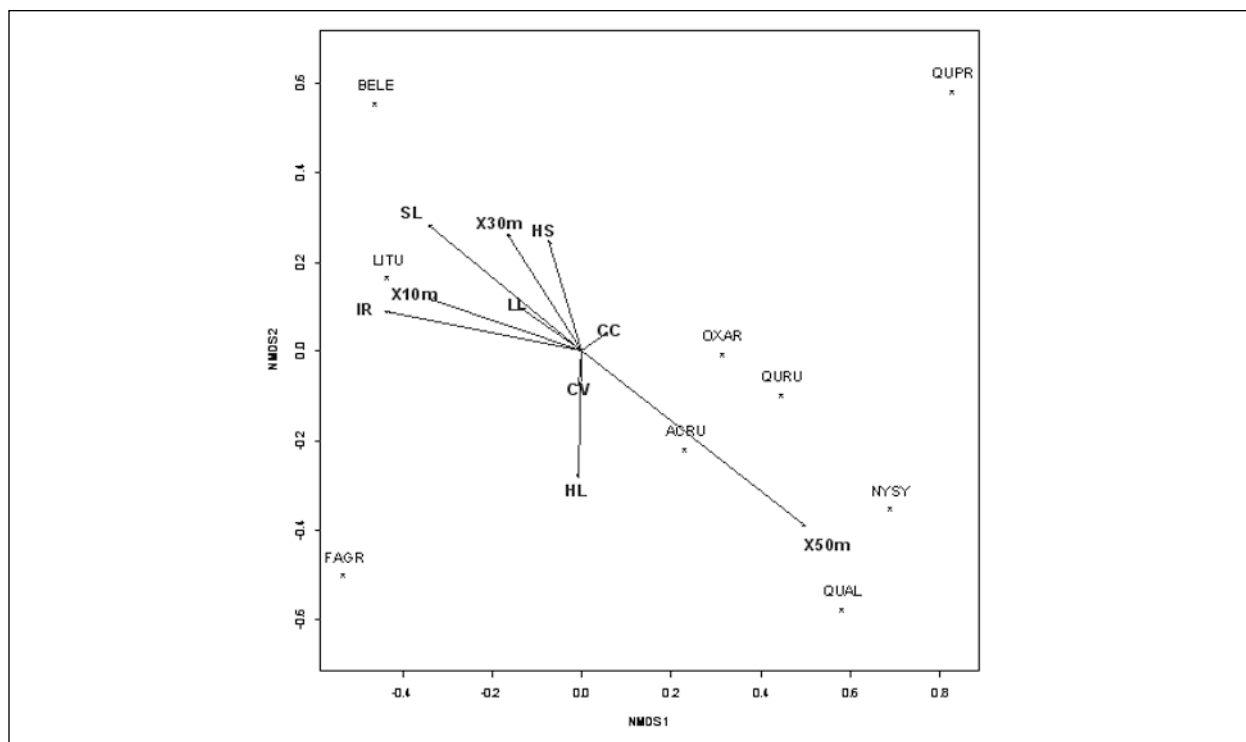


Figure 3.—Nonmetric multidimensional scaling (NMDS) ordination for overstory species with hemlock removed. Environmental vectors are principal components axes. The distance matrix for the NMDS is Bray-Curtis, based on relative basal area. Species are coded as follows: ACRU – red maple, ACSA – sugar maple, BELE – sweet birch, FAGR – American beech, LITU – yellow-poplar, NYSY – blackgum, OXAR – sourwood, QUAL – white oak, QUPR – chestnut oak, QURU – red oak.

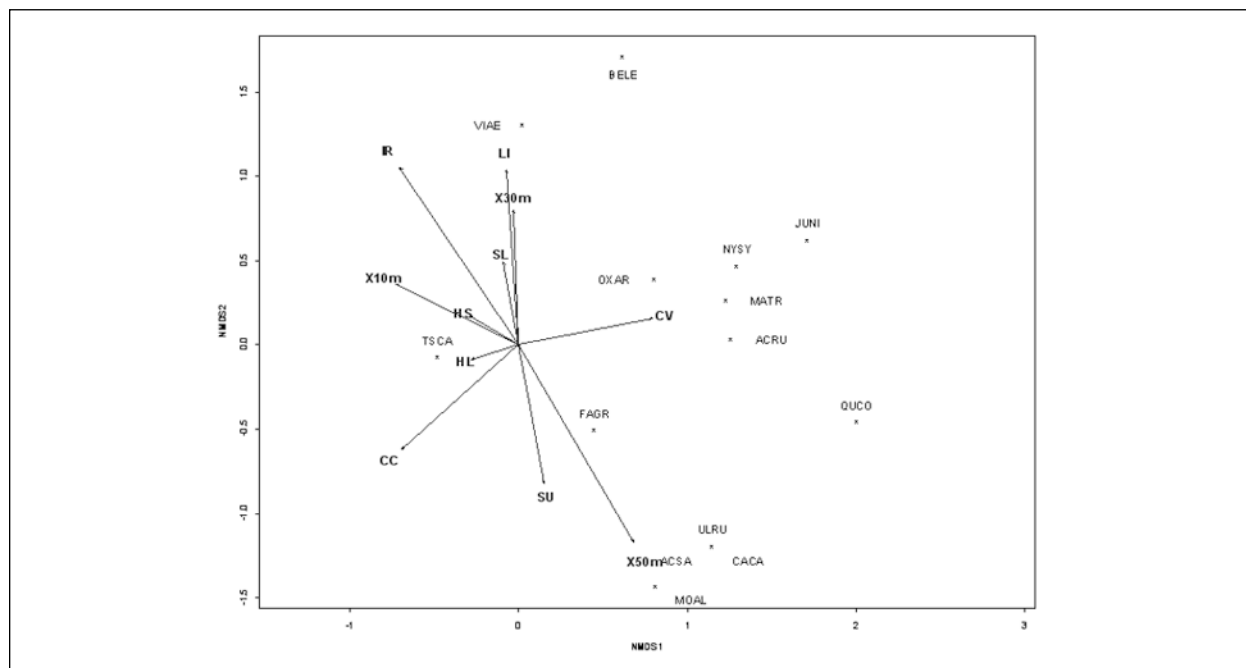


Figure 4.—Nonmetric multidimensional scaling (NMDS) ordination for sapling species with environmental principal components axes. The distance matrix for the NMDS is Bray-Curtis, based on stem counts. Species are coded as follows: ACRU – red maple, ACSA – sugar maple, BELE – sweet birch, CACA – hornbeam, FAGR – American beech, JUNI – black walnut, MATR – umbrella magnolia, MOAL – mulberry (*Morus alba* L.)*, NYSY – blackgum, OXAR – sourwood, QUCO – scarlet oak (*Quercus coccinea* Munchh.), TSCA – hemlock, ULRU – slippery elm, VIAE – wild grape (*Vitis aestivalis* Michx.).

*invasive

DISCUSSION

Throughout Ohio, hemlock is not broadly distributed; in the specialized habitats where it does occur, however, it dominates both the overstory and sapling layers, regardless of aspect. As few other species occur in either the overstory or sapling layers of these steep riparian ravines, the arrival of HWA would be a major disturbance in key headwater areas of the Ohio River watershed. Successional patterns that result following hemlock mortality will differ moving upslope away from first- and second-order streams as environmental gradients drive the re-initiation of deciduous hardwood communities.

The successional patterns we can expect for Ohio and central hardwood forest ecosystems where hemlock is a dominant overstory species will be unique and illustrative of the variance of the impact of HWA. Pollen records indicate that historic hemlock declines resulted in a transition to birch, followed by maple and oaks in the southern Appalachians (Allison and others 1986). These large-scale patterns from pollen records provide a general model that can be refined on regional and local scales for management purposes. For example, following the historic pattern, studies indicate that hemlock will likely be replaced by sweet birch in the Northeast, where hemlock occurs across the landscape (Orwig and Foster 1998, Kizlinski and others 2002). In the Connecticut Arboretum, Small and others (2005) found that hemlock ravines shifted toward a mixed hardwood overstory with a more developed understory.

The future composition and structure of ravine and riparian forests is not as clear for the central and southern portions of the hemlock's range (Ellison and others 2005). We found sweet birch sporadically, generally at lower slope positions. Birch was rarely found in the current understory. However, our study may underestimate the role of birch, as all birch at Lake Katharine were identified as yellow birch, in agreement with Runkle and Whitney (1987). For analysis purposes, the species were treated separately and yellow birch was later removed from analyses as very infrequent (present on less than 5 percent of all plots). Thus, the future influence of sweet birch and/or ecologically similar yellow birch will depend on its ability to disperse and germinate more widely throughout the forest. Birch is known as an opportunistic species well adapted to disturbances from canopy gaps created by hemlock mortality (Orwig and Foster 1998, Catovsky and Bazzaz 2000). At the same time, birch is a shade-intolerant, short-lived species that would not be expected to remain dominant after one generation (Ford and Vose 2007). Shade-tolerant, later successional mesophytic species well suited to the cooler microclimate of ravines, including sugar maple and basswood (*Tilia americana* L.), were infrequent and absent (respectively) from our plots. Runkle and Whitney (1987) suggest the low pH and poor soil quality may exclude these species. In addition to birch, yellow-poplar may also expand (Ford and Vose 2007), particularly as it is already more abundant at lower slope positions.

Additional data such as soil properties may be particularly illustrative to refine future successional trajectories (e.g., Boerner 2006). The influence of basic physiographic variables was particularly evident when the dominant influence of hemlock was removed from the overstory ordination. These gradients could be compared to known gradients within southern Ohio (e.g., Goebel and Hix 1997, Hix and Percy 1997, Hutchinson and others 1999, Small and McCarthy 2005), particularly with additional environmental data. Several studies indicate that nitrogen and soil moisture may be particularly influential in the region (Runkle and Whitney 1987, Hutchinson and others 1999, Small and McCarthy 2005). While such studies provide a solid foundation for the environment-vegetation relationships in the unglaciated Allegheny Plateau, hemlock-dominated ravines are not represented, and thus, the interaction of the unique physiographic setting and soil properties on the occurrence of species besides hemlock merits additional inquiry. The environmental

gradients measured as part of this study provide valuable insight, and we plan to build on this foundation with further analyses. In particular, future analyses of soil properties within the ravines will allow exploration of how varied soil properties are within 50 m of these streams and among ravines of the unglaciated Allegheny Plateau. It is reasonable to expect we will identify decreasing soil moisture moving upslope as an additional factor driving the transition from mesic species such as sugar maple and beech to increasingly dry-mesic and xeric oaks (Fig. 2).

The role of hemlock as a foundation species may be particularly important in the central and southern Appalachians, including the Central Hardwoods region. Hemlock dominates headwater riparian areas from ravine bottoms upslope, in many cases to the summit of short hills. Hemlock ravines contribute a unique community to landscape diversity. In riparian areas, hemlock is more likely to be replaced by species such as birch and yellow-poplar. These fast-growing, deciduous species that produce higher quality, faster-decaying leaf litter than hemlock will likely alter the cycling of nutrients and energy across riparian and headwater streams (Kominoski and others 2007, Ball and others 2008). At upper slope positions, if oak and hickory species are able to expand, species composition and function may not shift quite as far. This outcome may indicate more resilience to the loss of hemlock. Yet the future overstory will depend in part on the response of the seedling and sapling layer (Collins and Carson 2004), which is currently sparse. In a region with significant recreation and tourism centered in hemlock-dominated ravines, work to further understand hemlock ecosystem ecology and continue refining successional predictions is critical before the arrival of HWA.

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

THE EXTENT OF SELECTED NONNATIVE INVASIVE PLANTS ON SOUTHERN FOREST LANDS

Sonja N. Oswalt and Christopher M. Oswalt¹

Abstract.—Studies suggest that the southern United States is an area of primary concern with regards to the spread of nonnative invasive plant species. Recent data show that species such as Japanese honeysuckle (*Lonicera japonica*) and Nepalese browntop (*Microstegium vimineum*) are invading forests and displacing native species throughout the southern United States. Monitoring on large spatial scales is among the most important mechanisms for detecting the spread of nonnative species. Better assessments of ongoing biological invasions are a primary research priority in the Southeast. As one method for addressing this need, a federal-state partnership initiated a survey of 33 invasive plant taxa in 2001 on all southern forest ownerships. Presence information and estimates of cover for these invasives are collected as part of the U.S. Forest Service, Southern Research Station, Forest Inventory and Analysis (FIA) program. Within the FIA program, the Southern Invasive Plant Indicator Program monitors and reports on plant invasions within southern forests; it also develops tools to detect emerging invasions and potentially threatening invasive plant populations. Results from data collected through the FIA program in the southern United States from 2001 to 2008 are described in this paper. Current distribution maps for selected nonnative invasive plants are presented, along with a discussion of the potential impacts of these species on native forests of the southern United States.

INTRODUCTION

Foresters and ecologists have noted the spread of nonnative invasive species onto United States forest land for decades. Billions of dollars have been dedicated to researching the physiological mechanisms of invasions, the environmental impacts of invasion, and site qualities that facilitate invasion, and to pursuing management and control efforts yearly, yet it is unclear how successful control programs have been, to date (Pimentel and others 2005). For example, Pimentel and others (2005) estimate expenditures of 3 to 6 million dollars annually on attempts to control the spread of the invasive tree *Melaleuca quinquenervia* in Florida, and Webster and others (2006) stated that more than 17,000 man-hours were spent treating invasive species over a 10-year period in the Great Smoky Mountains National Park in Tennessee and North Carolina. Annual costs of efforts to control nonnative plant species (including species that do not occur on forest land) nationally are estimated to be as high as \$25 billion (Pimentel and others 2005).

Despite soaring costs and inestimable environmental impacts, nonnative invasive species continue to spread across managed and natural ecoregions, including forest land (Miller 2003). Southern forests are of particular concern because extended growing seasons and high site productivity, combined with relatively high disturbance rates and increasing forest land parcellization, may facilitate the growth and spread of hundreds of

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potentially noxious plants. Of the 31 terrestrial plants included on “100 of the World’s Worst Invasive Alien Species” (Lowe and others 2000), 23 occur in the southern states. The U.S. Department of Agriculture, Forest Service (USFS) recognizes 53 noxious species that are deemed currently problematic and widespread enough on southern U.S. forest land to warrant constant monitoring through the USFS Southern Research Station (SRS) Forest Inventory and Analysis (FIA) program (USFS 2007).

Working with state forestry agents throughout the South, SRS-FIA began monitoring nonnative invasive plant species (NNIPS) in 2001 in response to a growing desire to track potential forest health threats on U.S. forest land. No other program in the United States provides a mechanism for monitoring the spread of common invasive species across both public and private lands on a regularly updated basis. The survey of invasive plant species was added to the timber resource surveys conducted since the 1930s. The NNIPS selected for survey are regionally recognized exotic pest trees, shrubs, vines, grasses, canes, forbs, and ferns known to invade interior forest stands and forest edges, canopy gaps, or stream-sides. SRS-FIA’s Southern Invasive Plant Indicator Program (SIPIP) uses NNIPS presence information and estimates of cover to monitor and report on plant invasions within southern forests. Taking a proactive approach, SIPIP also develops tools to detect emerging invasions and potentially problematic invasive plant populations. This paper describes current results from data collected through the FIA program in the southern United States from 2001 to 2008. The paper shows the current distribution of selected nonnative invasive plants, along with discussing the potential impacts of these species on Southern forests.

STUDY AREA

The Southern Research Station FIA program collects data on public and private land in 13 southern states and U.S. Territories: Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, Texas, the U.S. Virgin Islands, and Puerto Rico. This paper includes data collected from 2001 to 2007 in all of the above except west Texas, Oklahoma, the U.S. Virgin Islands, and Puerto Rico. Collection dates for each state were: Alabama (2001 to 2005), Arkansas (2000 to 2005), Florida (2002 to 2007), Georgia (2005 to 2007), Kentucky (2000 to 2004), Louisiana (2001 to 2005), Mississippi (2006), North Carolina (2003 to 2007), South Carolina (2002 to 2006), Tennessee (2000 to 2004), east Texas (2004 to 2008), and Virginia (2002 to 2007).

METHODS

Field methods for the basic FIA program are well documented and interested users may refer to the SRS National Core Field Guide (USFS 2007) or to Bechtold and Patterson (2005) for detailed methodology. Briefly, the FIA program utilizes a nationally consistent systematic random sampling technique to locate sampling plots at three different geographic scales, referred to as “Phases.” Phase 1 assigns points to land-use strata to determine the amount of forest land in a sample area while reducing sample variance by grouping like units (Bechtold and Patterson 2005). Phase 2, a subset of the Phase 1 points, consists of field-visit plots at a scale of roughly one plot to every 6,000 acres. Data collected on Phase 2 plots include all FIA National Variables, along with regional variables assigned to answer specific monitoring needs. The NNIPS survey is included among the regionally collected Phase 2 variables for the southern United States. Phase 3 plots are a subset of Phase 2 plots and receive additional, intensive sampling for certain forest health variables not discussed in this paper.

Phase 2 plots consist of four 1/24-acre subplots located approximately 120 feet apart at 120-, 240-, and 360-degree azimuths from a central subplot. In addition to a suite of mensuration variables, SRS-FIA collects information on nonnative plants on each subplot. Up to four nonnative invasive plants from a predetermined list of species known to be problematic in the South are recorded on the forested portion of each subplot (Table 1). In addition to presence/absence of up to four species, field personnel are instructed to select a range of abundance (in the form of percent cover) of each nonnative invasive plant recorded on the subplot. Percent cover categories are: Trace <01 percent; 1-10 percent; 11-50 percent; 51-90 percent; 91-100 percent. In the winter, personnel are instructed to treat percent cover values as though plants were in leaf-on condition. SRS-FIA collects regular mensuration data (e.g., height, diameter at breast height) on any NNIPS trees found on each subplot if the species is part of the SRS-FIA tree species measurement list. For a complete list of trees measured by SRS-FIA, see the Forest Inventory and Analysis SRS Core Field Guide: Volume 1: Field Data Collection Procedures for Phase 2 Plots (USFS 2007). Values from the field are transmitted to the SRS-FIA office, where they are processed through the National Information Management System.

Distribution maps for this paper were created by grouping all inventoried invasive species into their respective life-forms, and plotting presence/absence of an invasive species in a Phase 2 plot by life-form using a geographic information system. We also identified the overall number of Phase 2 plots in which each invasive species occurred to examine the extent of invasion across the South. A total of 33,534 forested plots were sampled.

RESULTS FROM THE SOUTHERN INVASIVE PLANT INDICATOR PROGRAM

The SRS-FIA program includes 12 invasive trees on the SIPIP survey list. Six of those trees are inventoried only in Florida. Invasive trees, as a life-form, were spread fairly evenly across the southern United States, with some dense concentrations along the Gulf Coast in east Texas, Louisiana, and Mississippi, where Chinese tallowtree (*Triadica sebifera*) is particularly problematic (Fig. 1a, Fig. 2). Chinese tallowtree was detected in more plots than any other invasive tree species, followed by tree-of-heaven (*Ailanthus altissima*), mimosa (*Albizia julibrissin*), and Chinaberry (*Melia azedarach*) (Fig. 2).

Shrubs are more densely concentrated across the South than are trees, particularly from the eastern bank of the Mississippi River across to the Atlantic Coastal Plain (Fig. 1b). Chinese and European privets (*Ligustrum* spp.) are by far the most common invasive shrubs, becoming fairly ubiquitous on forest land from Tennessee and Arkansas south to northern Florida (Fig. 3). Nonnative roses (*Rosa* spp.), particularly multiflora rose (*Rosa multiflora*), are problematic from Tennessee north through Kentucky and into western Virginia, while Brazilian pepper (*Schinus terebinthifolius*) is the most common invasive shrub in the Florida peninsula.

Vines are the most common invasive life-form detected by SRS-FIA in the southern United States and are present throughout the entire southeastern forested landscape north of Florida (Fig. 1c). Japanese honeysuckle (*Lonicera japonica*) is the most widespread invasive vine in forests in the entire Southeast, and occupies 55 times the number of FIA plots as the next most common invasive vine, kudzu (*Pueraria montana*, Fig. 4). In fact, Japanese honeysuckle is the single most common invasive species across all life forms in the Southeast.

Invasive grasses occupy fewer FIA plots than trees, shrubs, or vines, but are still commonplace in forests, particularly in Tennessee, Kentucky, and Virginia (Fig. 1d). The most commonly detected grasses on forest land were tall fescue (*Lolium arundinaceum*) and Nepalese browntop (*Microstegium vimineum*) (Fig. 5).

Table 1.—Southern Research Station Forest Inventory and Analysis (SRS-FIA) Nonnative Invasive Plants monitoring list. Each species is monitored on all standard SRS-FIA field plots. Species are grouped by life-forms, and were compiled from State and Federal noxious plant lists.

Common Name	Scientific Name
Trees	
Australian pine	<i>Casuarina equisetifolia</i>
Camphor tree	<i>Cinnamomum camphora</i>
Carrotwood	<i>Cupaniopsis anacardioides</i>
Chinaberry	<i>Melia azedarach</i>
Chinese tallowtree	<i>Triadica sebifera</i> , <i>Sapium sebiferum</i>
Java plum	<i>Syzygium cumini</i>
Melaleuca	<i>Melaleuca quinquenervia</i>
Mimosa, silk tree	<i>Albizia julibrissin</i>
Princesstree, Royal Paulownia	<i>Paulownia tomentosa</i>
Russian olive	<i>Elaeagnus angustifolia</i>
Schefflera	<i>Schefflera actinophylla</i>
Tree-of-heaven	<i>Ailanthus altissima</i>
Shrubs	
Autumn olive	<i>Elaeagnus umbellata</i>
Bush honeysuckles	<i>Lonicera</i> spp.
Chinese / European privet	<i>Ligustrum sinense</i> / <i>L. vulgare</i>
Coral ardisia	<i>Ardisia crenata</i>
Japanese / glossy privet	<i>Ligustrum japonicum</i> / <i>L. lucidum</i>
Lantana	<i>Lantana camara</i>
Nonnative roses	<i>Rosa</i> spp.
Sacred bamboo, nandina	<i>Nandina domestica</i>
Silverthorn, thorny olive	<i>Elaeagnus pungens</i>
Winged burning bush	<i>Euonymus alata</i>
Vines	
Cat's-claw vine	<i>Macfadyena unguis-cati</i>
Chinese / Japanese wisteria	<i>Wisteria sinensis</i> / <i>W. floribunda</i>
English ivy	<i>Hedera helix</i>
Japanese honeysuckle	<i>Lonicera japonica</i>
Kudzu	<i>Pueraria Montana</i> var. <i>lobata</i>
Nonnative vincas, periwinkles	<i>Vinca minor</i> / <i>V. major</i>
Nonnative yams	<i>Dioscorea bulbifera</i> / <i>D. oppositifolia</i>
Oriental or Asian bittersweet	<i>Celastrus orbiculatus</i>
Rosary pea	<i>Abrus precatorius</i>
Skunk vine	<i>Paederia foetida</i>
Wintercreeper	<i>Euonymus fortunei</i>
Grasses	
Chinese silvergrass	<i>Miscanthus sinensis</i>
Cogongrass	<i>Imperata cylindrica</i>
Giant reed	<i>Arundo donax</i>
Napier grass	<i>Pennisetum purpureum</i>
Nepalese browntop	<i>Microstegium vimineum</i>
Nonnative bamboos	<i>Phyllostachys</i> spp. / <i>Bambus</i> spp.
Tall fescue	<i>Lolium arundinaceum</i>
Herbs	
Chinese lespedeza	<i>Lespedeza cuneata</i>
Garlic mustard	<i>Alliaria petiolata</i>
Hairy indigo	<i>Indigofera hirsuta</i>
Shrubby lespedeza	<i>Lespedeza bicolor</i>
Tropical soda apple	<i>Solanum viarum</i>
Ferns	
Japanese climbing fern	<i>Lygodium japonicum</i>
Old world climbing fern	<i>Lygodium microphyllum</i>
Sword fern	<i>Nephrolepis cordifolia</i>

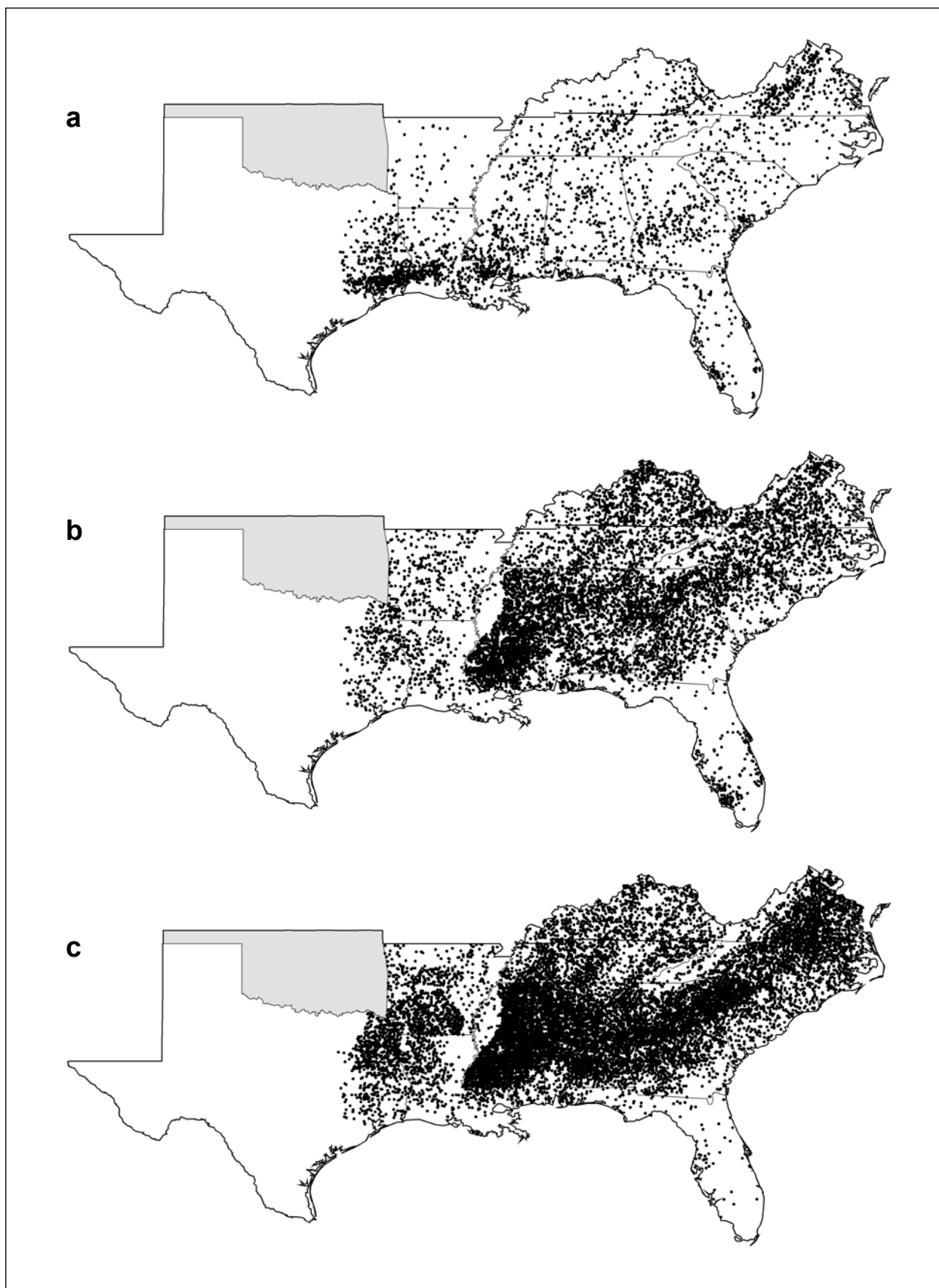


Figure 1.—Forest Inventory and Analysis plots containing invasive species in six life-forms from the FIA invasive species list in the southern United States (plot locations are approximate): (a) Trees, (b) Shrubs, (c) Vines, (d) Grasses, (e) Herbs, (f) Ferns. (Figure 1 continued on next page.)

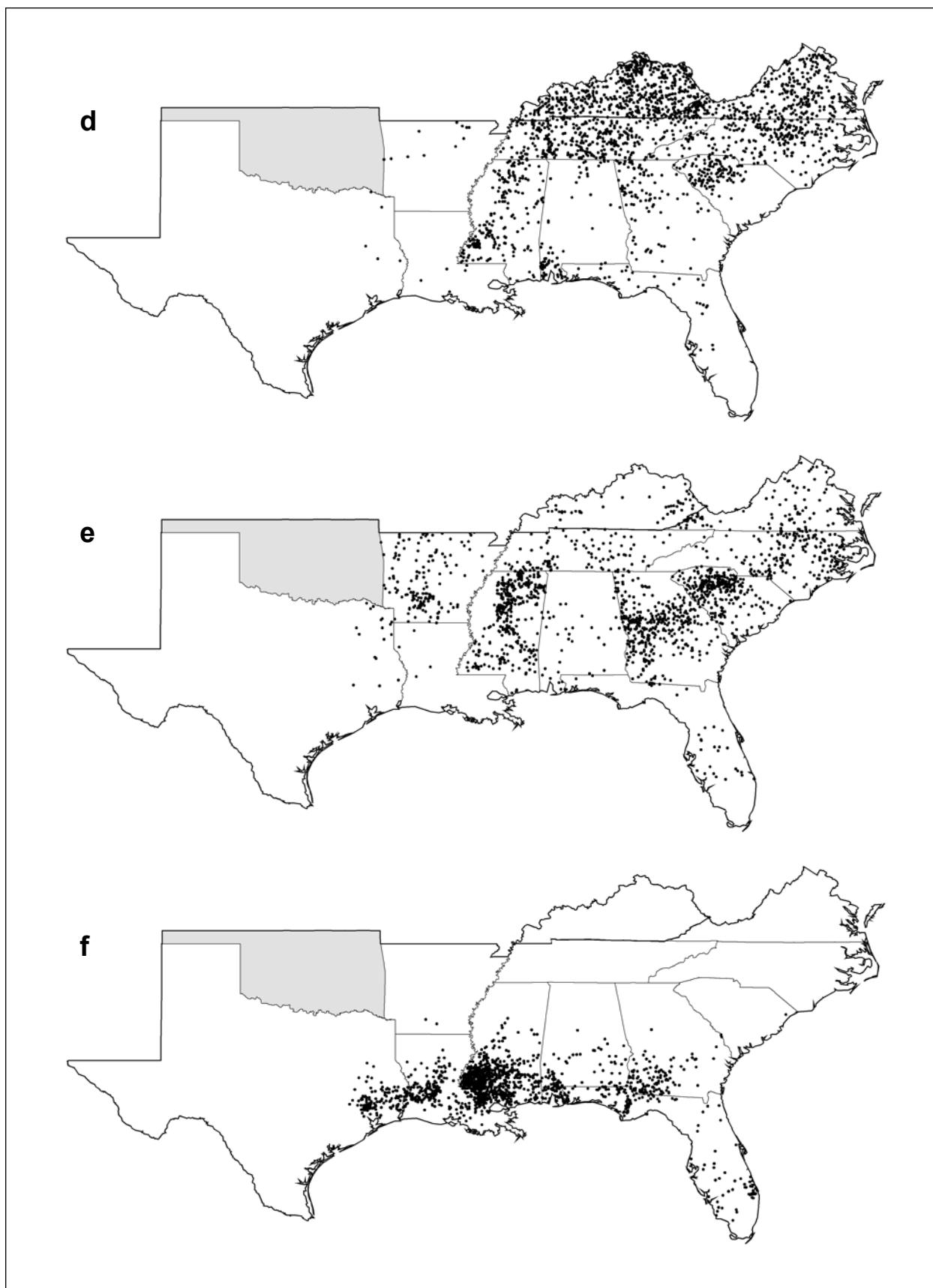


Figure 1 (continued).—Forest Inventory and Analysis plots containing invasive species in six life-forms from the FIA invasive species list in the southern United States (plot locations are approximate): (a) Trees, (b) Shrubs, (c) Vines, (d) Grasses, (e) Herbs, (f) Ferns.

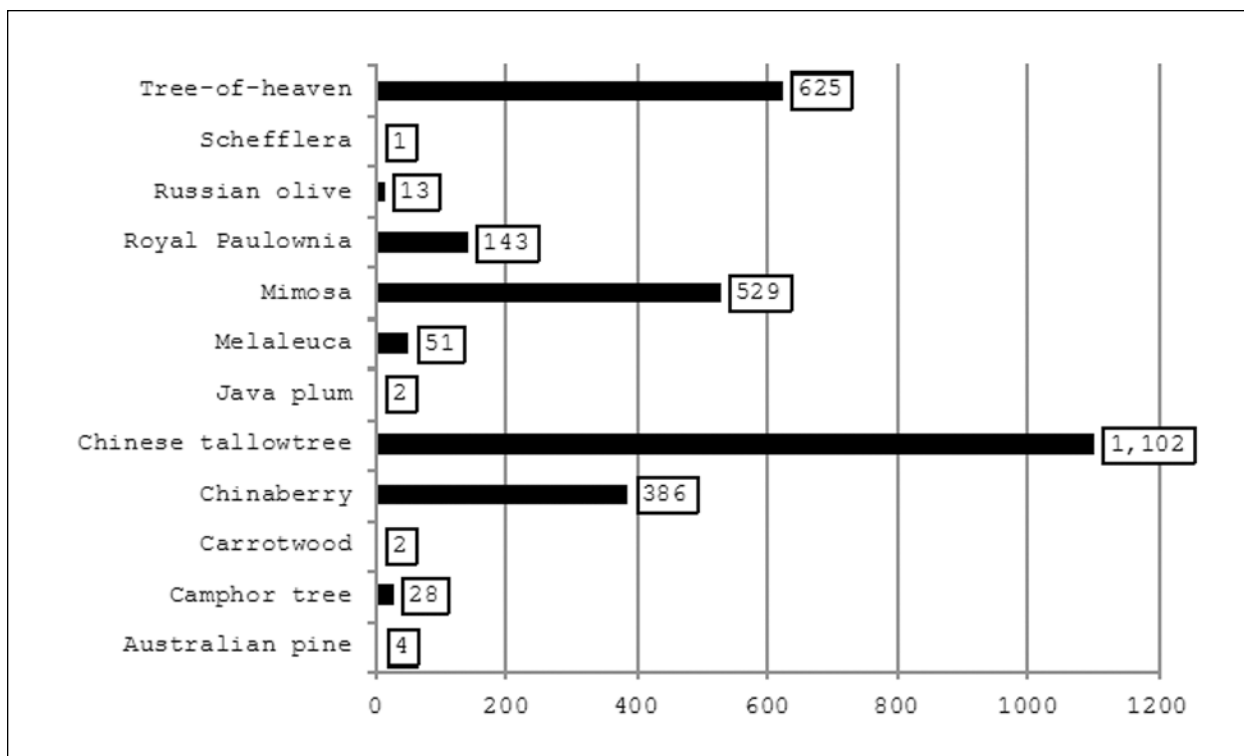


Figure 2.—Number of FIA plots containing invasive trees by species name. Plot counts are given at the end of each bar.

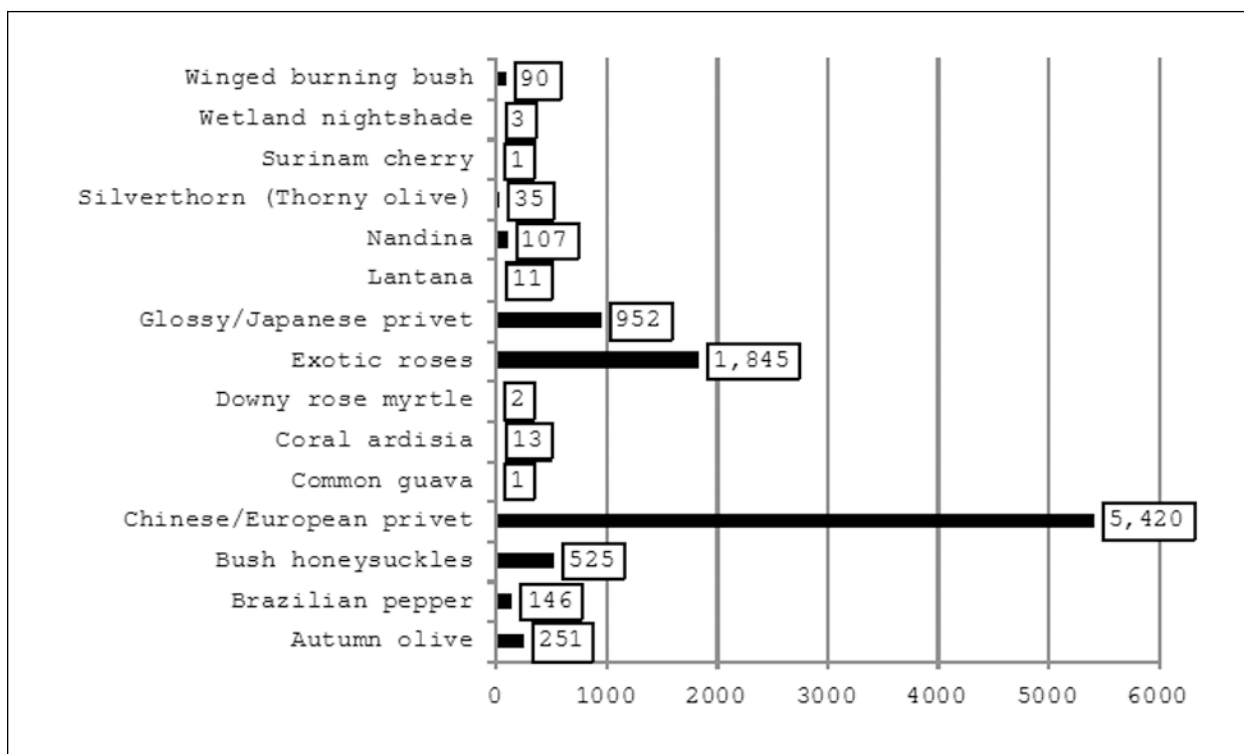


Figure 3.—Number of FIA plots containing invasive shrubs by species name. Plot counts are given at the end of each bar.

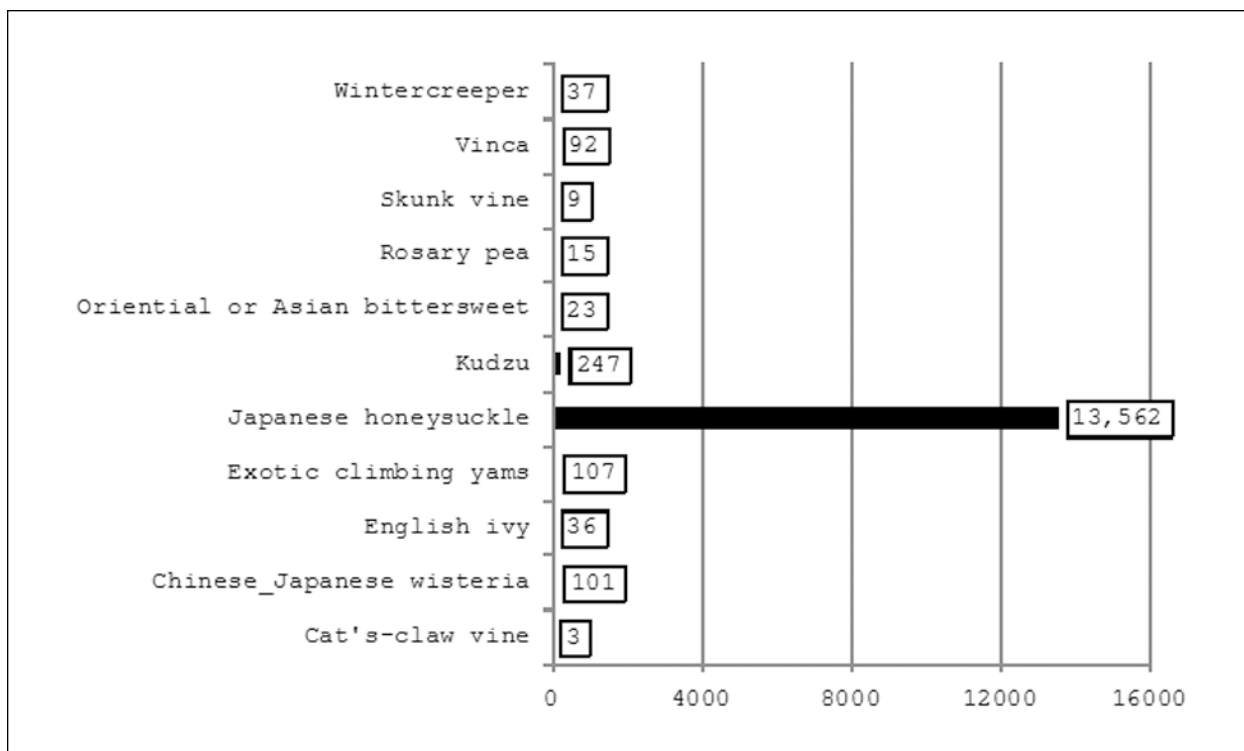


Figure 4.—Number of FIA plots containing invasive vines by species name. Plot counts are given at the end of each bar.

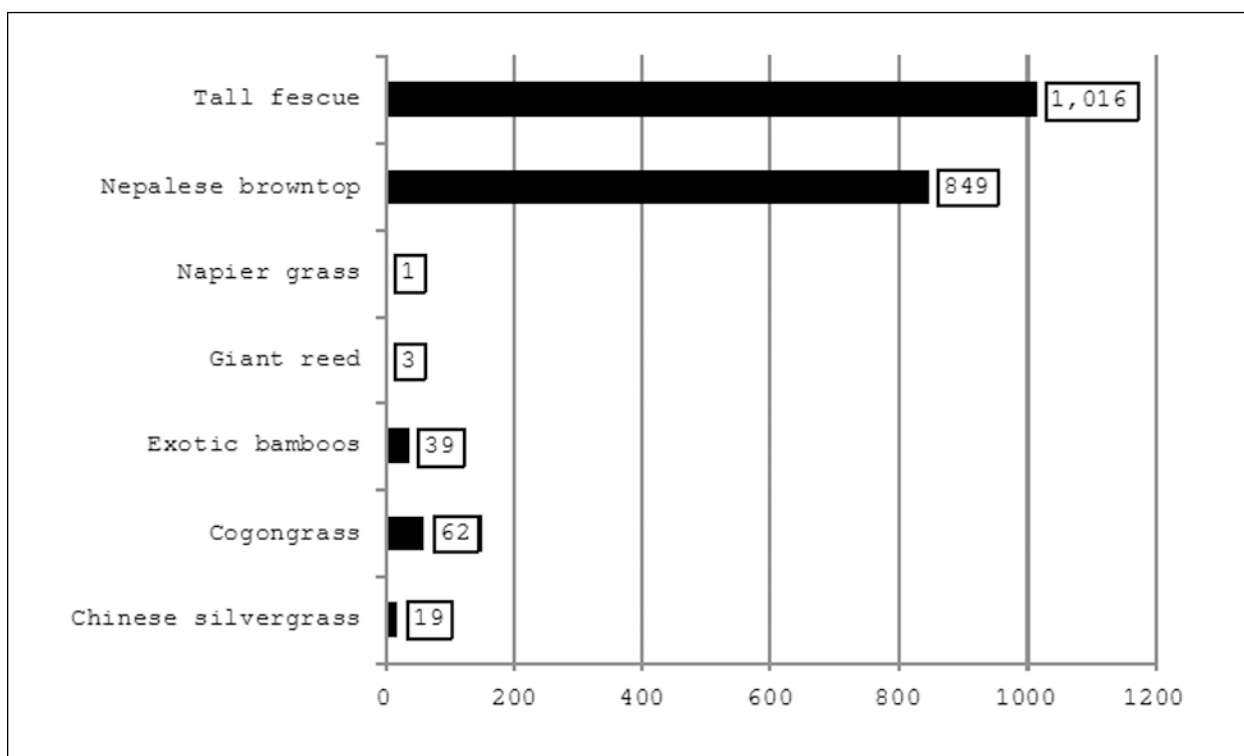


Figure 5.—Number of FIA plots containing invasive grasses by species name. Plot counts are given at the end of each bar.

Invasive herbs were not uniformly distributed across the South. Spatial aggregations tended to follow particular landscape features like roads or rivers (Fig. 1e). Lespedezas (*Lespedeza bicolor* and *L. cuneata*) were the herbs most commonly detected in the South, though garlic mustard (*Alliaria petiolata*) detection increased in the northernmost reaches of Kentucky and Virginia (Fig. 6).

Most nonnative invasive ferns do not survive the winter in areas outside the subtropical south. Therefore, it is not surprising that invasive ferns were most likely to be detected in forested plots of the Deep South (Fig. 1f). Japanese climbing fern (*Lygodium japonicum*) is the only species SRS-FIA monitors on forest land outside of Florida and not surprisingly, it was the most commonly detected invasive fern (Fig. 7).

OVERVIEW OF ONLINE PRODUCTS

The SIPIP team has utilized the SRS-FIA data to populate two publicly accessible online products, to date. The Nonnative Invasive Plant data tool (http://srsfia2.fs.fed.us/nonnative_invasive/Southern_Nonnative_Invasives.htm) provides user access to the raw subplot data collected in the field. The data have been compiled by state and species of interest, but has not been expanded to population levels. Users may select individual states or species to view tables of total species by subplot counts by county. Further drill-down selections allow users to access complete tables with state, county, plot, subplot, species, percent cover values, and control numbers used to link the NNIPS data with FIA field data obtained from other online sources. The raw tables provide subplot proportion information that users can apply to FIA plot data to expand NNIPS populations to large areas.

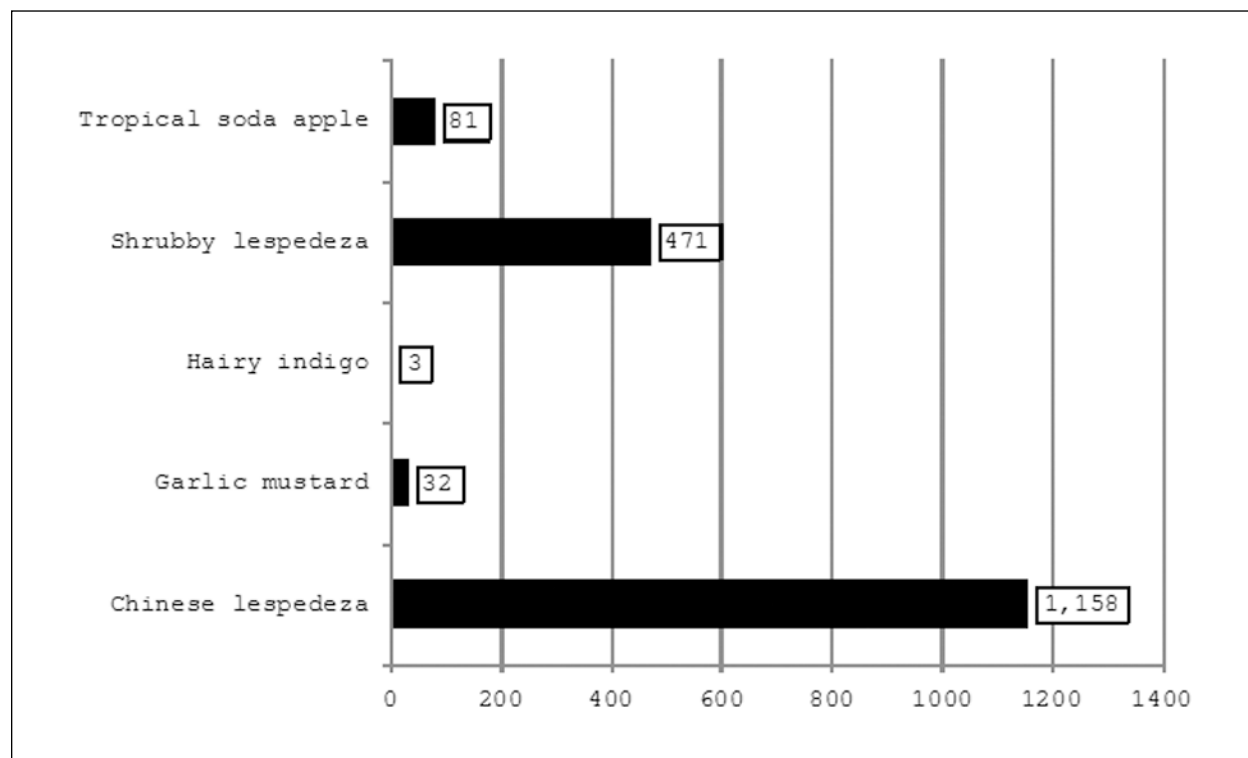


Figure 6.—Number of FIA plots containing invasive herbs by species name. Plot counts are given at the end of each bar.

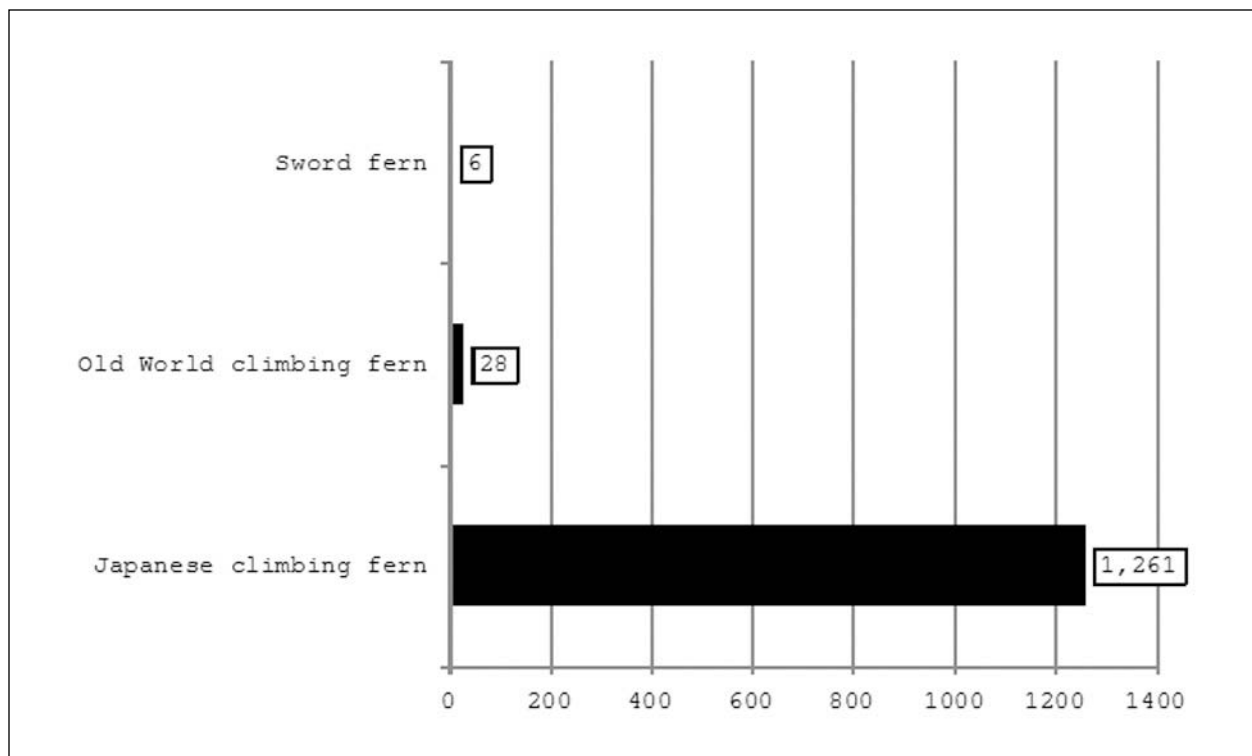


Figure 7.—Number of FIA plots containing invasive ferns by species name. Plot counts are given at the end of each bar.

In addition to the NNIPS data tool, the SIPIP team has created a series of Southern Invasive Plants Maps, which are southwide depictions of NNIPS detected in the most recent survey period available for each state (http://srsfia2.fs.fed.us/data_center/data_mapping.shtml). Species are grouped by life-form, and users may select a species to view choropleth maps that depict the percentage of forested subplots containing that particular species in each county. These maps are useful for visualizing the extent and degree of infestation of a particular species across multiple states.

IMPLICATIONS

The latest surveys conducted by SRS-FIA illustrate how widespread nonnative plant invasions are on forests throughout the South. Japanese honeysuckle, originally introduced as an ornamental plant in the 1800s (and still sold and planted as an ornamental and for deer browse) is the most common of all forest-invasive nonnative species detected by SRS-FIA in the southern United States. Though the leaves, fruit, and nectar from Japanese honeysuckle are used by many wildlife species and insects, the species poses a threat to native plants and trees through competition, and may lower species diversity in areas where it has become established (Dillenburg and others 1993, Yurkonis and others 2005).

Chinese tallowtree, the most common tree species detected on SRS-FIA plots, encroaches on native coastal prairies and freshwater marshes in many southern states, including east Texas, Louisiana, and Mississippi, threatening conversion to near-monoculture forest land and disrupting entire ecosystems (Bruce and others 1995). A recent study conducted by Oswalt (in press) documented a Chinese tallowtree population expansion of more than 500 percent in Louisiana from 1991 to 2005, with similar increases of 445 percent in Mississippi from 1994 to 2006 and 174 percent in east Texas.

Chinese and European privets, both introduced as ornamental species and now widespread in forests throughout the Southeast, have the potential to severely impact forest understory species diversity and limit tree regeneration in invaded sites (Merriam and Feil 2002). While privets were once touted as important winter food for white-tailed deer in some areas (Stroymayer and others 1998), few (if any) studies exist that compare the relative value of privet as habitat and forage to the value of native species that evolved in conjunction with the wildlife species inhabiting the Southeast (Graham 2002). Furthermore, no studies appear to document the impacts of privet establishment in forests on habitat specialists who utilize native understory structure and components.

Like many nonnative invasive species, the impacts of the most common nonnative forest-invasive herb, Chinese lespedeza, are not well documented. Deep taproots and allelopathic characteristics enable the species to compete successfully against native species (Stevens 2002). However, most studies of the species have been conducted in grasslands and other open spaces, where it is most likely to colonize and become highly invasive. Thus, further research regarding the impacts of the species in forested conditions is necessary.

The second most common forest-invasive nonnative grass, Nepalese browntop, has been shown to impact species diversity and native woody species density in hardwood forests in the Southeast (Oswalt and others 2007). Similarly, Adams and Engelhardt (2009) found that Nepalese browntop impacted plant community structure in a way that could have long-term repercussions for site diversity.

Nonnative invasive ferns are problematic primarily in the Deep South. The most common species detected in forest areas by SRS-FIA, Japanese climbing fern, has the ability to grow over plants and trees much like kudzu, smothering native species and reducing plant diversity (Minogue and others 2009).

All of the nonnative species mentioned in detail above are continuing to spread across the southeastern United States, and much research is still needed to understand the full impact they are having on forest ecosystems and the wildlife associated with those systems. The NNIPS data and mapping tools identified, described, and exemplified in this paper provide a mechanism for monitoring these species through time and space. In addition, users have the ability to link the data to other SRS-FIA plot-level data, which could provide researchers with tools necessary to better understand the impacts of nonnative invasive species at a large scale.

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

SCREENING BUTTERNUT AND BUTTERNUT HYBRIDS FOR RESISTANCE TO BUTTERNUT CANKER

J.R. McKenna, M.E. Ostry, and K. Woeste¹

Abstract.—Butternut (*Juglans cinerea*) is being killed throughout its native range by the fungus *Sirococcus clavigignenti-juglandacearum* (*Scj*). In recent years, many disease-free trees have been determined to be complex hybrids with an admixture of Japanese walnut (*J. ailantifolia*). We challenged 5-year-old trees from two progeny tests with *Scj* in 2008. The first test (northern Indiana), planted in 2003, had 37 diverse families (n=319). Thirty-two of these seedling families were derived from a grafted orchard. Five additional families were collected from hybrid trees. The second test (southern Indiana), planted in 2004, had 12 pure butternut half-sib families collected from a woodlot with: 4 resistant, 4 moderately resistant, 4 susceptible, and 1 resistant hybrid families (n=213). Resistance ratings were based on the disease status of the mother trees when the seed was harvested in the fall of 2002. Eleven black walnut (*J. nigra*) trees were also included. In early fall of 2008, trees were inoculated with two strains of *Scj* obtained from branch cankers on trees in two locations in Indiana. The trees were scored 8 months after inoculation for canker incidence and severity. Some trees in the first test were naturally infected by *Scj* and resulting canker incidence and severity were recorded. Butternut hybrid families were more resistant to natural infection than the pure butternut families. Eight months after inoculation, canker incidence and severity varied significantly among butternut hybrid families and *Scj* strain but not among pure butternut families.

INTRODUCTION

Butternut (*Juglans cinerea*) was not historically a dominant species in eastern deciduous forests, or a species that produced a large volume of timber. However, it has been locally important and very valuable for its edible nuts and its wood (Kellogg 1919). Currently, butternut is being killed throughout its native range by the butternut canker disease first reported from Wisconsin more than 40 years ago (Renlund 1971). The causal agent of the disease is a fungus, *Sirococcus clavigignenti-juglandacearum* (*Scj*), which was described as a new species by Nair and others in 1979. The knowledge we have at present indicates it began causing disease in the southern part of butternut's range (Anderson and LaMadeleine 1978). A 1976 survey of butternut in Wisconsin revealed that 31 percent were diseased and 9 percent were dead, and, by 1992, 92 percent of the trees were diseased and 27 percent were dead (Carlson and Guthmiller 1993).

Sticky conidia (asexual spores) of *Scj* can be carried by insects (Katovich and Ostry 1998, Halik and Bergdahl 2002), birds, and, presumably, small mammals into the crowns of trees, where spore germination and infection of branches occur. Infection of boles and buttress roots results from spores spread in rainsplash from the diseased branches (Tisserat and Kuntz 1983). Trees are killed as multiple cankers coalesce and girdle the

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bole. Cankers caused by *Scj* are easily recognized and the pathogen is easy to culture. A number of factors strongly indicate that *Scj* is an exotic pathogen. Detection reports and dissections of trees have shown the disease was not present in North America much more than 60 years ago; no sexual stage of the fungus has been identified, and randomly amplified polymorphic DNA (RAPD) markers of the fungus revealed no appreciable genetic variation (Furnier and others 1999). Natural resistance is rare as most native butternut has died or is dying across its range, further suggesting that the pathogen has been recently introduced. Additionally, *Scj* has not been reported as a pathogen elsewhere in the world. When the fungus was introduced and from where are still not known.

Resistance to *Scj* in butternut is fundamental for the conservation of the species (Michler and others 2006). No specific mechanism of resistance has been identified, and we do not know the number and type of genes that may be involved. However, it has been observed repeatedly that various butternut hybrids and other *Juglans* species across the landscape are less affected by the disease compared to pure butternut (Ostry 1997). Apparent differences between yard-grown butternut hybrids and forest-grown pure butternuts may be the result of hybrid vigor, reduced competition for light and water, or *Scj* resistance.

Orchard and others (1982) screened a variety of *Juglans* species and hybrids with *Scj* using artificial stem inoculations. They observed that all *Juglans* species and hybrids were susceptible and mean canker lengths for Japanese walnut (*J. ailantifolia*) were smaller than *J. × bixbyi* (butternut × Japanese walnut, known as a “buart” F1 hybrid), Persian walnut (*J. regia*), and black walnut (*J. nigra*). These authors and others have noted that stem inoculations through artificial wounds may bypass natural resistance mechanisms, as black walnut was only rarely infected under natural conditions (Orchard and others 1982, Ostry and Moore 2007, and others). It is possible that *Scj* infects black walnut more commonly than previously believed (Ostry and others 1997), but that colonization is usually limited to small twigs, which has a negligible effect on tree health (K. Broders, personal communication).

For several decades, numerous healthy butternut trees have been identified in the wild where other butternuts have been killed by butternut canker disease. Many such healthy trees have been selected and have been clonally propagated to provide potentially resistant germplasm for future breeding and reintroduction. At present, a robust and effective screening method is necessary to select and sort out *Scj*-resistant sources from those susceptible to *Scj* (Michler and others 2006). Ostry and Moore (2008) investigated artificial inoculation methods with grafted ramets of a number of these selected trees with the goal of developing a satisfactory screening method to facilitate breeding canker-resistant butternut. Similar to Orchard and others (1982), Ostry and Moore (2008) were able to infect every clonal accession they tested. However, significant differences in canker incidence and mean canker length among the clones were obtained.

Although hybrids of butternut with Japanese walnut have been known to exist for some time (Bixby 1919), we are just beginning to understand how common hybridization has become. Complicating the matter further is that *Juglans* species are self-compatible (Rink and others 1989, Busov and others 2002, Bai and others 2006), and that buart hybrids are highly fruitful, making them unique among *Juglans* F1 interspecific hybrids, which otherwise yield less than each parental species or may be almost completely sterile. It is uncertain how many healthy “butternuts” surviving today are instead complex hybrids such as F2 hybrids, F1 buart backcrossed to butternut (Hoban and others 2008, Ross-Davis and others 2008, Hoban and others 2009), or even F2 hybrids backcrossed to butternut. If sufficient resistance to *Scj* is not present within butternut, fertile

hybrids would be useful for back-cross breeding, not unlike the breeding program under way with American chestnut (Orchard and others 1982, Schlarbaum 1997, Michler and others 2006).

We present preliminary results of two tests designed to screen the relative resistance of selected open-pollinated seedling families derived from an orchard of putative resistant clones and families from a natural stand where healthy trees have persisted to maturity. We also compared the relative resistance of hybrid butternut families.

MATERIAL AND METHODS

PLANT MATERIAL AND PLOT DESIGN

Seeds for both tests were collected in the fall of 2002 and grown in two test plantations. The first test was established at Purdue University's Martell Forest, West Lafayette, IN. For this plantation (Breeding Block), seed was collected from 6- to 10-year-old grafted trees in an orchard composed of putative resistant and susceptible butternuts from across the species' range (Table 1). Several offspring of the mother trees later proved to be butternut hybrids as were several open-pollinated half-sib families from highly resistant mother trees in northern Indiana (see Hybrid Determination). Seed was stratified in a cooler over the winter, sprouted in the greenhouse in April 2003, and planted in the field the following month. Ten seeds per family were planted as two five-tree plots at a spacing of 3.7 m between rows and 1.8 m within rows.

For the second test plantation (Resistance Block), seeds were collected from 12 mother trees in a 16-ha woodlot in southern Wisconsin near the town of Whitewater. This mixed hardwood stand is a natural population unique in that butternut regenerated here 50-60 years ago with a density approaching 250 trees per ha in areas, and trees exhibit both a wide range of bark color and a wide range of disease severity (Ostry and Woeste, 2004). We selected four trees from each of three categories—resistant, moderately resistant, and susceptible—based on their health status in the field (Table 1). The “resistant” female parents had been monitored and rated as canker-free and healthy for more than 20 years. As a resistant check lot, we included seeds from the ortet of a patented putative butternut, ‘Bountiful,’ which was growing in Vera, MO. At the time the seeds were collected, we suspected that ‘Bountiful’ was a hybrid, and confirmed our suspicion with DNA markers (described below) soon afterward. The seeds for this test were stratified and seedlings were grown in 7.6-liter containers in 2003 and then were planted as dormant seedlings in the spring of 2004 at the Southeastern Purdue Agricultural Center in Butlerville, IN. Trees in the plantation were planted on a 2.4-m × 2.4-m spacing as a randomized complete block. There were 18 blocks with a single tree per family in each block. For several butternut families, fewer than 18 seedlings of acceptable quality were alive by the time of planting. To achieve a full stocking rate in the plantation, black walnut seedlings were planted as filler trees and were subsequently inoculated along with the butternut.

HYBRID DETERMINATION

The phenotype of each seedling was rated based on morphological characteristics. Two independent observers rated each seedling on a three-point scale: 2 = pure *J. cinerea*; 1 = pure and hybrid traits mixed; and 0 = hybrid using the methods described by Woeste and others (2009). F1 hybrid accessions were easily recognized by their shell characteristics and were excluded from the block. Thus, any remaining hybrids could be more complex F2's or backcrossed hybrids. In the former case, we were able to confirm that some of the families were hybrids using a chloroplast DNA marker based on a sequence polymorphism in the TRNF-L intergenic

Table 1.—Geographic origin of source material for the Breeding Block and Resistance Block (bold) screening.

Type	HTIRC Acc. # ²	Ostry Acc # ³	Origin/Location ⁴	Canker Rating ⁵	Observed ⁶
Hybrid Butternut					
	# 696	-	Vera, MO	None (R)	2003/2005
	# 702	-	New Paris, IN	None	2003/2009
	# 704	-	Plymouth, IN	None	2002
	# 706	-	New Paris, IN	2 cankers	2003/2009
	# 707	-	Brimfield, IN	None	2003/2009
	# 708	-	Steuben Co., IN	None	2002
	# 710	OS-# 7	Madison, WI	None	1990
	# 711	OS-# 8	Madison, WI	None	1990
	# 731	OS-# 91	Clover Lick, WV	None	1996
	# 732	OS-# 92	Loudon, NH	None	1996
	# 734	OS-#128	Sanford, ME	None	1995
	# 735	OS-#129	Sanford, ME	None	1995
	# 748	OS-#184	Chequam., NF, WI	-	1996
	# 750	OS-#191	Ankeny, IA	Healthy	1996
Butternut					
	# 709	OS-# 6	Caledonia, MN	None	
	# 712	OS-# 10	Arlington, WI	Branch cankers	1993
	# 713	OS-# 14	Rochester, MN	Branch cankers	1994
	# 714	OS-# 16	Rochester, MN	2 cankers	1993
	# 715	OS-# 17	Rochester, MN	1 branch canker	1995
	# 716	OS-# 19	Rochester, MN	None	1997
	# 717	OS-# 20	Whitewater, WI	None (R)	1997/2009
	# 718	OS-# 22	Whitewater, WI	None (R)	1997/2009
	# 722	OS-# 54	Nicolet NF, WI	None	1997
	# 723	OS-# 60	Whitewater, WI	Healthy	1994/2009
	# 724	OS-# 61	Whitewater, WI	Healthy (MR)	1994/2009
	# 725	OS-# 63	Whitewater, WI	Healthy (R)	1994/2009
	# 726	OS-# 67	Mazaska Lake, MN	Healthy	1994
	# 727	OS-# 78	Rochester, MN	Cankered	1991
	# 728	OS-# 85	M. Twain NF, MO	-	1995
	# 730	OS-# 87	M. Twain NF, MO	-	1995
	# 733	OS-# 97	Perch River, NY	Healthy	1994
	# 736	OS-#132	Berlin, VT	-	1995
	# 738	OS-#141	Trade Lake, WI	Healthy	1995
	# 740	OS-#147	Whitewater, WI	Healthy (MR)	1995/2009
	# 741	OS-#148	Whitewater, WI	Healthy	1997/2009
	# 742	OS-#149	Stratford, NH	Healthy	1995
	# 743	OS-#157	PA	-	1995
	# 744	OS-#159	IA	-	1995
	# 746	OS-#171	NY	-	1995
	# 747	OS-#181	Bark River, MI	Healthy	1996
	# 751	OS-#401	Whitewater, WI	Cankered/dead(S)	2002/2009
	# 752	OS-#403	Whitewater, WI	Cankered/dead(S)	2002/2009
	# 757	OS-#414	Whitewater, WI	Healthy	2002/2009
	# 767	OS-#423	Whitewater, WI	Cankered/dead(S)	2002/2009
	# 769	OS-#426	Whitewater, WI	Cankered (MR)	2002/2009
	# 772	OS-#430	Whitewater, WI	Cankered/dead(S)	2002/2009
	# 773	OS-#431	Whitewater, WI	Cankered (MR)	2002/2009

² Hardwood Tree Improvement & Regeneration Center, USDA FS Northern Research/Purdue University, accession number.

³ Ostry, USDA Forest Service North Central accession number.

⁴ Location where the original scion material (plain) or seed (bold) originated.

⁵ Canker rating of the original ortet or seed tree (bold) in the field. S = susceptible, MR = moderately resistant, and R = resistant selections for the resistance test.

⁶ Most recent year(s) when canker status of each ortet or seed tree was last evaluated.

spacer (Aradhya and others 2006, Woeste, unpublished data). The Japanese walnut haplotype of the TRNF-L sequence polymorphism is recognized by the restriction enzyme *Mbo*II. In practice, most butternut hybrids can be identified using this cleaved amplified polymorphic sequence (CAPS) marker to determine whether a suspect tree carries the chloroplast of Japanese walnut or butternut. Because *Juglans* chloroplasts are inherited maternally, maternal siblings of a seedling containing a Japanese walnut chloroplast will contain the same chloroplast, irrespective of their male parent. A second marker (ITS/*Bs*IE) is a nuclear (CAPS) marker based on polymorphism in the ITS sequence. This marker is useful for detecting F1 hybrids or cases where a hybrid has pollinated a pure butternut (Woeste, unpublished data).

FUNGAL STRAINS AND CULTURE

We had no information regarding different levels of aggressiveness among Indiana *Scj* strains in our collection. We used a strain isolated from the Breeding Block at the Martell Forest designated as *Scj* IN-1375-4A and a second strain from more than 100 km away in the Hoosier National Forest of southern Indiana, designated *Scj* IN-1378-3 and collected in August 2008. Inoculum was prepared from sporulating 2-month-old cultures of these two *Scj* strains grown on malt agar at 20 °C in the dark.

INOCULATION

In late September and early October 2008, when the trees are most susceptible to infection (Ostry and Moore 2007, 2008), all trees with at least a 1.0-m tall, relatively clear trunk and with a minimum diameter at breast height (d.b.h.) of 17 mm, were inoculated in both plantations. Four 6-mm diameter holes were drilled through the bark and just into the cambium of the main trunk of each tree and each of the four wounds was separated by at least 20 cm analogous to the method developed for screening chestnut to the chestnut blight (*Cryphonectria parasitica*) fungus (Anagnostakis 1992). A 6-mm diameter plug of fungus and agar was inserted into the hole, with the fungal hyphae facing the cambium. Each inoculated wound was wrapped with one layer of masking tape. *Scj* IN-1375-4A was consistently inoculated into the upper two holes and *Scj* IN-1378-3 into the bottom two holes, following the method for chestnut screening (Anagnostakis 1992). This systematic approach is useful to keep track of each strain easily over time. Previous studies with different *Scj* strains randomly inoculated along branches in the greenhouse showed differences in aggressiveness (Ostry, unpublished data), suggesting strain rather than branch position was a significant factor in canker development. For trees with less than 1.0 m of clear trunk and a d.b.h less than 17 mm, only two wounds were made, each with a single inoculation of each strain.

CANKER EVALUATION

In the Breeding Block, natural canker infection began in 2006, when the seedlings were in their third growing season. *Scj* was isolated from several samples of natural cankers in August 2008, confirming *Scj* as the causal agent. Each tree was evaluated and rated for cumulative natural canker incidence and severity in November of 2009 using a subjective scale. Incidence was rated as 0 to 3, where 0 = no natural cankers; 1 = 1 or 2 cankers; 2 = 3 to 5 cankers, and 3 = 6 or more. Severity was assigned to each tree based on the average relative size of the cankers, where 1 = small ~30 × 10 mm (length × width); 2 = medium ~60 × 20 mm; and 3 = large ~100 × 25 mm sized cankers.

Responses to inoculations were evaluated after 8 months in May 2009. The external vertical length and maximum horizontal width of each canker were recorded. Each individual artificial inoculation site was treated as an experimental unit. To achieve balanced data for statistical evaluations, compensating for dead,

missing, or small suppressed trees, we considered all of the upper and lower inoculations per *Scj* strain as two replications. Incidence data were determined simply as the number of inoculations per family that produced a measurable canker beyond non-inoculated wounded controls. Because half or fewer of our inoculations produced a canker, instances where no canker resulted were removed from the dataset for comparison of canker size among genotypes and *Scj* strains. Hence, for both tests, both canker incidence and canker size were associated with the three main fixed effects: *Scj* strain, family, and butternut type—hybrid or pure. Canker incidences and severity (external length) were subject to 2-way ANOVA (Excel 2007, Microsoft Corp., Seattle, WA). Least squares means separation tests for canker size on hybrid families were performed using Fisher's least significant difference procedure. Differences between butternut hybrids and pure butternut in canker development from natural infection were analyzed using Chi-square (SAS Version 9.2, SAS Institute, Cary, NC, 2008).

RESULTS

BREEDING BLOCK

The correlation between canker length and width was highly significant ($P \leq 0.0001$, $r = 0.56$). Given this correlation, we considered only vertical canker length (parallel with the axis of the trunk) in the analyses. Inoculated butternut hybrids and pure butternuts had equal canker lengths when the data were pooled over all families and both *Scj* strains, (ANOVA $P \leq 0.16$.) When butternut hybrid and pure butternut families were analyzed separately, butternut hybrid families differed significantly ($P \leq 0.02$) for canker size, while canker size on pure butternut families was not significantly different.

The *Scj* strain used was a highly significant source of variation for canker length (ANOVA $P \leq 0.0001$). *Scj* IN-1375-4A produced larger cankers (ANOVA $P \leq 0.007$) on both butternut hybrids and pure butternut than did *Scj* IN-1378-3 (Table 2). Inoculation with *Scj* IN-1375-4A led to nearly twice the number of cankers per family compared to *Scj* IN-1378-3 (Table 3). The canker incidence was remarkably consistent between butternut hybrids and butternut. After inoculation with *Scj* IN-1375-4A, the incidence of infection in butternut hybrids and pure butternut was 55 percent and 50 percent, respectively. Inoculations with *Scj* IN-1378-3 resulted in a canker incidence of 26 percent for both types. Family effects were not significant for canker incidence. Incidence ranged from 10 percent to 85 percent for *Scj* IN-1375-4A inoculations and 0 percent to 50 percent for *Scj* IN-1378-3 inoculations for both types.

Cankers were significantly smaller (12 percent) on hybrid families than on pure butternut families, with an overall canker length of 63 mm vs. 71 mm, respectively (Table 2). After inoculation with *Scj* IN-1375-4A, the ranges of canker lengths were 33 mm to 92 mm for hybrid families and 51 mm to 94 mm for butternut families. After inoculation with *Scj* IN-1378-3, the ranges of canker lengths were 19 mm to 92 mm for hybrids and 23 mm to 94 mm for butternut. The range data did not include two families that made no cankers when inoculated with *Scj* IN-1375-4A. Cankers produced by *Scj* IN-1378-3 were more variable in size than those produced by the other strain. The coefficients of variation (CV) for canker length for the hybrids and butternut inoculated with *Scj* IN-1375-4A were 0.28 and 0.17, respectively. By contrast, butternut hybrids and butternut inoculated with *Scj* IN-1378-3 had CV's of 0.47 and 0.41, respectively.

Table 2.—Length of cankers 8 months after inoculation of Breeding Block hybrid butternut and butternut families with two *Scj* strains, West Lafayette, IN.

Type	Family	Sdlgs (no.)	Inoc. (no.)	Canker Length (mm)		Avg.
				Scj Strain		
				IN-1375-4A	IN-1378-3	
Hybrid Butternut						
	# 706	3	6	32.6 a	18.5 ab	25.6
	# 707	8	20	46.9 ab	00.0 a	23.5
	# 750	10	20	50.2 ab	68.3 c	59.3
	# 748	7	12	58.5 b	52.0 c	55.3
	# 711	10	18	55.1 b	91.8 d	73.5
	# 735	10	20	74.9 c	59.5 c	67.2
	# 710	10	20	75.9 cd	63.3 c	69.6
	# 732	9	18	90.3 cd	32.0 b	61.2
	# 704	9	18	78.8 cd	70.1 cd	74.5
	# 702	6	12	79.3 cd	69.8 cd	74.6
	# 708	8	13	86.1 cd	71.5 cd	78.8
	# 734	10	18	93.2 cd	88.5 d	90.9
	# 731	10	20	95.3 d	27.7 b	61.5
Sum /Mean		110	215	70.5**7	54.8**	62.7
SD				19.9	27.5	19.4
Butternut						
	# 709	10	20	78.8	23.3	51.1
	# 712	10	16	77.2	52.5	64.8
	# 713	10	16	67.6	37.3	52.4
	# 714	9	18	62.5	93.0	77.7
	# 715	7	16	62.1	68.8	65.4
	# 716	9	18	65.7	60.9	63.3
	# 717	9	16	92.0	89.8	90.9
	# 718	10	20	89.5	88.4	89.0
	# 722	10	20	93.8	46.1	69.9
	# 723	8	16	81.4	88.8	85.1
	# 726	9	15	93.3	87.7	90.5
	# 727	10	16	57.9	102.0	80.0
	# 728	8	14	50.6	41.7	46.1
	# 730	10	20	78.7	81.5	80.1
	# 733	10	18	76.1	99.9	88.0
	# 736	10	20	68.5	67.5	68.0
	# 738	9	14	59.4	0.0	29.7
	# 741	10	20	73.3	55.5	64.4
	# 742	7	14	68.5	91.8	80.1
	# 743	10	18	93.3	56.9	75.1
	# 744	9	18	84.3	60.0	72.1
	# 746	10	20	86.7	89.7	88.2
	# 747	10	20	73.1	42.2	57.6
Sum/Mean		122	403	75.4*	66.3*	70.9
SD				12.6	26.7	16.0

⁷ *Scj* strain significantly affected canker length, $P \leq 0.007$ for both butternut hybrid and butternut families. Among families, significant difference occurred only for hybrid families. Means followed by different letters are significant at $P \leq 0.02$ by Fisher's protected least significant difference = 19.7 mm.

Table 3.—Canker incidence 8 months after inoculation of hybrid butternut and butternut families with two *Scj* strains in the Breeding Block, West Lafayette, IN.

Type	Family	Sdlgs (no.)	Canker Incidence by <i>Scj</i> Strain (%)	
			IN-1375-4A	IN-1378-3
Hybrid Butternut	# 702	6	58	25
	# 704	9	67	28
	# 706	3	50	17
	# 707	8	35	0
	# 708	8	77	31
	# 710	10	55	35
	# 711	10	50	22
	# 731	10	45	15
	# 732	9	50	6
	# 734	10	56	50
	# 735	10	85	35
	# 748	7	58	33
	# 750	10	30	45
	Sum/mean		110	55***
SD			15	14
Butternut	# 709	10	60	25
	# 712	10	31	13
	# 713	10	44	13
	# 714	9	67	33
	# 715	7	19	19
	# 716	9	67	33
	# 717	9	56	44
	# 718	10	65	40
	# 722	10	30	20
	# 723	8	69	25
	# 726	9	47	27
	# 727	10	56	13
	# 728	8	43	43
	# 730	10	65	30
	# 733	10	61	39
	# 736	10	10	35
	# 738	9	43	0
	# 741	10	60	25
	# 742	7	43	21
	# 743	10	67	39
	# 744	9	22	22
	# 746	10	85	35
	# 747	10	35	15
	Sum/mean		214	50***
SD			19	11

⁸ *Scj* strain significantly affected canker incidence $P \leq 1.0 \times 10^{-6}$.

NATURAL INFECTION

Butternut hybrids and pure butternut trees differed significantly (Chi-square value with 1 degree of freedom = 25.4 $P \leq 0.0001$) for incidence of canker development resulting from natural infection by *Scj* (Table 4). Cankers developed on 6 out of 133 butternut hybrid trees over the last 3 years compared to 58 out of 227 pure butternut trees. No significant difference in disease severity among butternut hybrid or pure butternut families was observed. Because families were planted in only two groups of five contiguous seedlings, we did not have enough statistical power, given the low incidence of canker development resulting from natural infection, to analyze the data for family effects.

RESISTANCE BLOCK

In the Resistance Block, only one butternut hybrid family (#696) was included. *Scj* strains were significantly different in terms of canker incidence (ANOVA, $P \leq 0.0001$). *Scj* IN-1375-4A produced cankers 39 percent of the time vs. 15 percent with *Scj* IN-1378-3 (Table 5). Canker development on pure butternut families did not differ significantly. In response to inoculation with *Scj* IN-1378-3, trees among the butternut hybrid family #696 and black walnut trees had a canker incidence of 3 percent and 4 percent, respectively.

The correlation for canker length and width was similar to that in the Breeding Block and was highly significant ($P \leq 0.002$; $r = 0.55$). We analyzed canker severity by canker length alone. Once again, we found *Scj* strain to be a highly significant source of variation for canker size (ANOVA $P \leq 0.0006$); *Scj* IN-1375-4A produced larger cankers (Table 6). There was no difference, however, among pure butternut families in canker size. The hybrid family #696 and black walnut had canker lengths similar to pure butternut families when inoculated with *Scj* IN-1375-4A. When inoculated with *Scj* IN-1378-3, these two genotypes produced smaller canker lengths compared to pure butternut (Table 6).

DISCUSSION

Despite past reports that the *Scj* pathogen has little to no genetic variation (Furnier and others, 1999), the present studies, along with those of Ostry and Moore (2007, 2008) demonstrate that *Scj* strain significantly affects canker size. Inoculation of *Scj* IN-1375-4A induced more cankers, and in most cases, larger cankers than the other strain, and may indicate that *Scj* strains vary in aggressiveness. An elegant system utilizing two strains, one highly aggressive and the other much less aggressive, was developed for screening American chestnut for susceptibility to chestnut blight (Anagnostakis 1992). Seedlings that developed large cankers when inoculated with the weaker strain of chestnut blight are rogued out quickly as they have no resistance. Trees that show resistance to the weaker strain are then evaluated for their reaction to the more aggressive strain, retained, and evaluated more carefully. Just how much *Scj* strains vary in aggressiveness needs to be investigated further and future screening should include a highly aggressive strain.

Artificial inoculation of trees with *Scj* did not reproduce results seen in the field under natural conditions. In the Breeding Block, butternut hybrid families with a low incidence of butternut canker from natural infections were readily infected when artificially inoculated through stem wounds. In the Resistance Block, incidence of canker was similar among black walnut and the #696 butternut hybrid family. Additionally, there was no difference in canker incidence among diverse families selected in the native stand in Whitewater, WI, and only marginal differences in canker size among inoculated butternut hybrid families in the Breeding Block. This result strongly suggests that results of artificial stem inoculations do not reflect natural resistance observed over decades in the field. Such an inoculation method apparently does not allow the expression of resistance mechanism(s).

Table 4.—Incidence of butternut canker from natural infection on butternut hybrids and pure butternut families as of the sixth growing season in the Breeding Block, West Lafayette, IN.

Type	Family	Sdgs (no.)	Canker Incidence (%)	Avg. Severity ⁹	Avg. Size ¹⁰
Hybrid Butternut					
	# 702	7	0	-	-
	# 704	10	0	-	-
	# 706	5	0	-	-
	# 707	10	0	-	-
	# 708	9	0	-	-
	# 710	10	20	1.0	3
	# 711	10	10	1.0	1
	# 731	10	0	-	-
	# 732	10	0	-	-
	# 734	10	0	-	-
	# 735	10	10	1.0	3
	# 748	10	10	1.0	1
	# 750	10	10	1.0	3
Sum/mean		121	5*** ¹¹	1.0	2.1
SD			7	0.0	1.0
Butternut					
	# 709	10	40	1.3	2.3
	# 712	10	20	3.0	3.0
	# 713	10	20	1.5	1.0
	# 714	10	40	2.0	2.3
	# 715	8	50	2.5	2.0
	# 716	10	40	2.0	2.0
	# 717	10	30	1.3	1.3
	# 718	10	20	1.5	2.5
	# 722	10	40	1.3	1.5
	# 723	10	10	1.0	1.0
	# 726	10	10	1.0	1.0
	# 727	10	40	1.5	1.5
	# 728	10	20	1.0	2.0
	# 730	10	10	1.0	2.0
	# 733	10	0	-	-
	# 736	10	0	-	-
	# 738	9	0	-	-
	# 741	10	20	1.5	1.5
	# 742	10	20	1.0	2.5
	# 743	10	30	1.3	2.0
	# 744	10	80	2.0	2.0
	# 746	10	50	1.2	2.2
	# 747	10	10	1.0	3.0
Sum/mean		227	26	1.5	1.9

⁹ Average severity based on ratings: 1 = 1 or 2 cankers; 2 = 3 to 5 cankers, and 3 = 6 or more.

¹⁰ Average size class of cankers where 1 = small ~30 × 10 mm; 2 = medium ~60 × 20 mm; and 3 = large ~100 × 25 mm.

¹¹ Hybrid butternut has a highly significant lower incidence of natural infections, $P \leq 0.0001$ by chi-square.

Table 5.—Canker incidence 8 months after inoculation of butternut families in the Resistance Block, Butlerville, IN.

HTIRC ACC #	Type	Rating ¹²	Sdls. Inoc. (no.)	Canker Incidence (%)	
				<i>Scj</i> Strain	
				IN-1375-4A	IN-1378-3
# 717	Butternut	R	18	43	20
# 718	Butternut	R	17	24	12
# 725	Butternut	R	16	37	20
# 757	Butternut	R	16	59	26
Means				41*** ¹³	20
# 724	Butternut	MR	17	29	21
# 740	Butternut	MR	15	33	30
# 769	Butternut	MR	12	42	8
# 773	Butternut	MR	18	50	13
Means				39	18
# 751	Butternut	S	18	32	18
# 752	Butternut	S	15	44	14
# 767	Butternut	S	11	37	15
# 772	Butternut	S	16	44	13
Means				39	15
# 696 ¹⁴	Hybrid Butternut	R	17	46	3
BW ¹⁵	Black walnut	R	11	32	5
Means				39	4

¹² Rating based on the canker status and health of each seed tree in the forest stand near Whitewater, WI. Trees were designated in fall 2002 as (R) resistant; (MR) moderately resistant; (S) susceptible. Revaluation of the stand in November 2009 showed (R) trees remaining healthy, (MR's) in decline, and (S's) dead or dying.

¹³ *Scj* strains are significantly different, $P \leq 0.2.5 \times 10^{-5}$.

¹⁴ Half-sib family of the hybrid butternut cultivar 'Bountiful.'

¹⁵ Black walnut seedlings added to complete blocks where butternut families were absent at the time of planting.

The suggestion that Japanese walnut offers a source of resistance to *Scj* is supported by our results. The most compelling evidence is that butternut hybrids and pure butternuts showed a significant difference in the frequency of canker development as the result of natural infection in the Breeding Block, but not in the frequency of those produced in response to artificial inoculations.

Despite the considerable variation in these families' genetic background, the incidence and severity of canker resulting from artificial inoculations were similar. Nevertheless, absent a better inoculation method, numerical differences in canker size may still be a useful measure of relative resistance for screening breeding material. Ostry and Moore (2008) found differences in canker length among the grafted ramets they tested ($P \leq 0.05$), similar to differences among hybrid families in the breeding block that we observed ($P \leq 0.02$). Ostry and Moore (2008) also found that canker size ranged greatly within a genotype, which we also observed, and that separation of clones by mean canker length did not result in well defined resistant and susceptible classes. The check group used by Ostry and Moore (2008), which consisted of seedlings presumed to be susceptible, had the largest mean canker length. The two resistant checks in the Resistance Block had ~20-mm smaller canker lengths than all of the pure butternut families.

Table 6.—Length of cankers 8 months after inoculation of butternut families in the Resistance Block, Butlerville, IN.

HTIRC ACC #	Type	Rating ¹⁶	Sdls. Inoc. (no.)	Canker Length (mm)	
				<i>Scj</i> Strain	
				IN-1375-4A	IN-1378-3
# 717	Butternut	R	18	100.6	44.8
# 718	Butternut	R	17	79.9	88.3
# 725	Butternut	R	16	107.6	86.0
# 757	Butternut	R	16	106.1	80.3
	Means			98.5*** ¹⁷	74.9
	SD			12.8	20.3
# 724	Butternut	MR	17	96.9	53.3
# 740	Butternut	MR	15	101.3	56.9
# 769	Butternut	MR	12	77.1	47.5
# 773	Butternut	MR	18	98.0	96.8
	Means			93.3	63.6
	SD			11.0	22.4
# 751	Butternut	S	18	103.1	46.7
# 752	Butternut	S	15	84.3	73.8
# 767	Butternut	S	11	94.8	83.0
# 772	Butternut	S	16	87.5	81.0
	Means			92.4	71.1
	SD			8.3	16.8
# 696 ¹⁸	Hybrid Butternut	R	17	43.0	32.5
BW ¹⁹	Black walnut	R	11	104.7	50.5
	Means			73.8	41.5
	SD			47.5	21.5

¹⁶ Rating based on the canker status and health of each seed tree in the forest stand near Whitewater, WI. Trees were designated in fall 2002 as (R) resistant; (MR) moderately resistant; (S) susceptible. Re-evaluation of the stand in November 2009 showed (R) trees remaining healthy, (MR's) in decline, and (S's) dead or dying.

¹⁷ *Scj* strains are significantly different $P \leq 2.5 \times 10^{-5}$.

¹⁸ Half-sib family of the hybrid butternut cultivar 'Bountiful.'

¹⁹ Black walnut seedlings added to complete blocks where butternut families were absent at the time of planting.

For the Resistance Block, we selected butternut mother trees that exhibited clear differences in resistance in the field. Although we expected their progeny families also to reflect these differences, we found no differences among them despite field observations of the mother trees' level of resistance over 20 years. If our method of stem inoculations did in fact circumvent the tree's resistance mechanism(s), it may not be surprising that we found no differences among these progeny, and furthermore, there may be no basis for inferring the number of genes or gene-action involved in *Scj* resistance in butternut. Resistance to *Scj* in butternut may be a complex quantitative trait and screening more than 10 to 18 seedlings per family may be necessary. This approach needs to be investigated.

Final conclusions cannot be drawn at this time with just 8 months of data. In the present two studies, 12 months after inoculation, some cankers have callused and appear to have compartmentalized the pathogen; this response will need to be factored into future analyses. Conversely, some of the inoculations that were

scored as having no appreciable canker development relative to non-inoculated wounds have produced cankers at 12 months. Most significantly, some trees scored as resistant based upon inoculation studies have been infected by wild inoculum, with significant canker development. Further observations at 20 months and 24 months will be needed to draw a final conclusion on the fate of each inoculation. The rate of canker elongation over time, and the number of new cankers that develop from natural infections now that the disease is established in the plantings, may reveal more family differences in reaction to *Scj*.

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

SILVICULTURE

USING SILVICULTURE TO SUSTAIN UPLAND OAK FORESTS UNDER STRESS ON THE DANIEL BOONE NATIONAL FOREST, KENTUCKY

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Abstract.—We used a large-scale silvicultural assessment designed to examine the efficacy of five stand-level prescriptions in reducing the potential impacts of gypsy moth infestations and oak decline on upland hardwood forests in Kentucky's Daniel Boone National Forest. Prescriptions involved a mix of intermediate stand treatments aimed at increasing residual tree vigor and regeneration treatments aimed at maintaining regeneration diversity. Prescriptions were as follows: (1) shelterwood with reserves, (2) shelterwood with midstory removal through herbicide, (3) B-line thinning, (4) creation of an oak woodland, and (5) untreated control. Thirty stands were chosen, half originally classified as sub-mesic and half classified as sub-xeric oak forest types. Prescriptions were replicated three times on each site type. Moisture classification index classified 24 stands as sub-xeric and 6 as sub-mesic. Stand basal areas were reduced to 22.9, 80.6, and 68.3 square feet per acre, in the shelterwood with reserves, thinning, and oak woodland treatments, producing reductions equivalent to 81, 39, and 52 percent, respectively. Early assessment showed a slight increase in tree vigor as determined by crown cover and position for residual trees in these three treatments. In the oak shelterwood treatment, oak seedlings greater than 1 foot in height increased from 191 to 325 stems per acre following herbicide treatment that targeted non-oak mid-canopy trees such as red maple, yellow-poplar, and blackgum.

INTRODUCTION

The Cumberland Plateau, the southernmost section of the Appalachian Plateau's physiographic province, comprises more than 9 million acres of forest land (Fenneman 1938; Smalley 1979, 1982, 1986). Kentucky's Daniel Boone National Forest (DBNF), London District, lies in the heart of the Cumberland Plateau. On the Plateau uplands, mixed hardwood forests dominate, and oaks are the most widespread genus, with the most common being white oak (*Quercus alba* L.), scarlet oak (*Q. coccinea* Muench.), black oak (*Q. velutina* Lamarck), northern red oak (*Q. rubra* L.), post oak (*Q. stellata* Wang.), and chestnut oak (*Q. montana* L.) (Braun 1950, Hinkle 1989). On the most xeric sites, oaks such as chestnut oak and scarlet oak dominate, and on less xeric sites, northern red oak and white oak are more common. These stands also contain myriad other species, including hickories (*Carya* spp.), maples (*Acer* spp.), yellow-poplar (*Liriodendron tulipifera* L.), sourwood (*Oxydendrum arboreum* DC), sassafras (*Sassafras albidum* [Nutt.]Nees.), and scattered pine (*Pinus* spp.). Stands are considered stressed due to site conditions, management histories, competition between species, and other disturbances. Another potential stressor on Kentucky's forests is the encroachment of the gypsy moth (*Lymantria dispar* L.).

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Gypsy moth currently infests 25 percent of its potential range in eastern North America and is spreading at a rate of 13 miles per year (Liebhold and others 1989, 1992). A national management program was initiated in 2000 to slow the spread of gypsy moth and has reduced the spread rate by 50 percent. It is estimated that gypsy moth will spread to the DBNF over the next 15 to 30 years with trap catches of gypsy moth males occurring in northeastern Kentucky over the last couple of years. In other stands defoliated by gypsy moth, silvicultural treatments that increased tree and crown vigor were found to reduce mortality (Gottschalk and others 1998).

This study examines treatments that DBNF management might use to increase overall stand vigor and to regenerate stands while maintaining a high oak component. Silviculture regimes for oak-dominated stands and for oak regeneration methods have been described in similar systems (Loftis 1983, Sander and others 1984, Loftis 1990a, Dey 1991, Johnson and others 2002, Brose and others 2008). We manipulated stand structure in two primary forest types (sub-xeric and sub-mesic oak forests) to sustain or increase stand vigor, while giving appropriate attention to how such disturbances influence the competitiveness of oak regeneration. This study will assess the alteration in species composition and stand structure with respect to vulnerability to gypsy moth and oak decline. We will also assess the projected regeneration outcomes. As part of this study, researchers will examine the costs, operational performance, and forest floor disturbance impacts of an integrated mechanized forest operation used to implement the treatments. In this paper, we will present preliminary data detailing sub-xeric and sub-mesic site characterization, and changes in forest structure post-treatment, including basal area reduction, relative stand density reduction, species composition changes, and canopy cover and class differences.

METHODS

SITE DESCRIPTION

The study area is located on the Cold Hill Area of the London Ranger District of the DBNF, Kentucky. The treatment stands are located on the Central Escarpment (221 Hb) land-type association, as described in the Land and Resource Management Plan (2004: 1-8). All treatment stands are located on broad ridges. Treatment stands were relatively similar prior to treatment and uniform within stand boundaries, and are best described as upland hardwood forests dominated by oak species. Total basal area ranges from 100 to 120 square feet per acre, relative stand density from 60 to 104 percent, and ages from 70 to 150 years old. The stands have been subjected to various silvicultural treatments, including harvesting and prescribed burning, since the National Forest acquired the land, but the stands are representative of fully stocked upland hardwood forests on the Cumberland Plateau. Stand boundaries were delineated based on several factors, including administrative constraints, proximity to road infrastructure, stand history, ownership boundaries, topography, soil type, and species composition.

TREATMENTS

The field structure for this study was a quasi-randomized design, 2 X 5 factorial with three replications. Replications were not blocked. Stands, which equal treatments, were randomly chosen from a pool of potential stands located across the landscape. Forest types were randomly distributed but represented fixed factors: sub-mesic (site index for upland oaks estimated at 65-80 feet at base age 50) and sub-xeric (site index 50-65 feet) oak forests (Smalley 1986). The other factor was silvicultural treatment with five levels: (1) shelterwood with reserves, (2) oak shelterwood, (3) thinning, (4) oak woodland, and (5) a control. Fifteen

stands from each of the two forest types were randomly assigned a treatment. Thus, 10 treatment conditions were replicated across the study area. The following is a description of each of the silvicultural treatments:

1. **Shelterwood with reserves.** This treatment will leave a residual basal area of 10 to 25 square feet per acre. Residual trees will be selected to promote increased forest health conditions and to improve habitat for wildlife and plant species that benefit from open, low basal area forest conditions. Oak species will be favored. A new stand will regenerate beneath the reserve trees and eventually create a two-aged stand structure.
2. **Oak shelterwood.** This treatment will not initially impact the overstory basal area. All basal area will be removed from the midstory and understory without making canopy gaps in the overstory. Undesirable tree species less than 3 inches diameter at breast height (d.b.h.) will be treated with a thinline basal bark treatment using triclopyr ester. Trees greater than 3 inches d.b.h. in the midstories and understories will be treated with a stem injection method using triclopyr amine. Undesirable tree species include those specifically in competition with oaks, such as red maple (*A. rubrum* L.), yellow-poplar, and trees with unhealthy stems and/or crowns. When sufficient advanced oak regeneration is present, the overwood will be removed to create a new even-aged oak stand (Loftis 1990b).
3. **Thinning.** This treatment will utilize the Gingrich stocking chart to thin to B-level stocking (Gingrich 1967). Reducing tree density will allow the residual trees to take advantage of improved growing conditions. The result should be increased tree vigor, larger crown diameters, continued or improved diameter growth, and increased capacity to survive defoliation. The thinning treatment was marked using crown vigor guidelines as well as stocking goals to match the presalvage thinning prescription (Gottschalk and MacFarlane 1992, Gottschalk 1993).
4. **Oak woodland.** This treatment will be conducted by first thinning to 45 to 70 square feet per acre followed by prescribed burning every 3 to 5 years. White oaks will be favored as residual trees to increase hard mast production and bat habitat. An objective of the treatment will be spatial and vertical heterogeneity. Another objective is to increase the native ground flora of forbs and grasses that depend on fire. Prescribed burns will be operational spring burns conducted by DBNF personnel.
5. **Control.** This treatment will not receive a silvicultural prescription and will be used to compare and evaluate the results of change from the above treatments.

A mechanical tree-length harvesting system was used to harvest all units. The system consisted of a feller buncher, grapple skidder, and a knuckleboom loader. Trees larger than 23 inches d.b.h. were felled with a chainsaw. All limbing and topping were performed with a chainsaw in the stand. Products removed from the stands included hardwood sawtimber and biomass logs. A biomass log was any material greater than 3 inches d.b.h., reasonably straight, and at least 10 feet long, and that did not qualify as a saw log. Harvesting began in November 2007 and was completed for all 18 stands in September 2009. Herbicide treatment for the oak shelterwood was done between October 2008 and March 2009.

DATA COLLECTION

We established twenty 0.1-acre vegetation measurement plots in each stand and measured plots prior to and just after treatment implementation. All plot centers are permanently marked with rebar, flagging, and global positioning system coordinates. We permanently labeled all trees 4.6 inches and greater d.b.h. We measured and recorded tree species, d.b.h., crown condition, tree grade, canopy cover, and tree height. In each plot, we also created a 0.01-acre plot where we enumerated regeneration (trees < 1.5 inches d.b.h.) by species

and height class. We randomly selected five vegetation plots in each unit, and on those plots we tagged five representative seedlings per regeneration plot and recorded species and measured height and basal diameter (immediately above the root collar). Overstory data were collected following one growing season for all stands and regeneration data were collected following one growing season for all stands except those treated in late summer 2009, which were excluded from this analysis. For post-treatment in the oak shelterwood treatment only, we surveyed status, species, and d.b.h. for all stems 1.6 inches and greater on five 0.025-acre plots to assess this treatment, which targeted smaller-diameter stems.

STATISTICS

The combination of the two factors (2 X 5) and three replications will result in 30 treatment stands. The general statistical analysis model for categorical and continuous variables measured in the study is a mixed model analysis of variance for factorial designs. Because treatment stands were chosen from the larger population of stands within the Cumberland Plateau, they will be treated as random within the analysis. Forest type will be treated as fixed. Due to the relatively large area occupied by each treatment stand, the assumption of independence between treatments is warranted. Treatment stands were assigned randomly within each replication and replications were assigned randomly. Statistical significance of the F-value at the 0.05 significance level for a factor or interaction between factors would initiate further analyses. We used Tukey's Honestly Significant Difference (HSD) tests for means separation. All analyses were carried out using SAS® Version 8.01 (SAS Institute Inc., Cary, NC).

RESULTS

SITE CLASSIFICATION

We used a moisture classification index (MCI) devised by McNab and others (2002) to classify each stand as sub-xeric or sub-mesic. The initial classification was done by DBNF personnel based on broad characterization of each stand's topography and edaphology, and through an estimated site index based on limited tree core analysis and soil surveys. McNab and others (2002) classification uses a stand-level species list, assigns a moisture weight to each species listed, and then calculates an average MCI by summing all weights and dividing by the total number of species listed. The MCI scale is from 0 to 4, with a species such as red maple having a weight of 0 (indicative of neither xeric nor mesic conditions), a post oak having a weight of 1 (xeric classification if value is <1.5), and a basswood (*Tilia glabra* Vent.) having a weight of 4 (mesic classification if value is >3.5). Moisture weights from McNab and others' (2002) were used, and species tallied in DBNF stands not listed in their paper were assigned a value by McNab (W.H. McNab, Southern Research Station, U.S. Forest Service, 1577 Brevard Road, Asheville, NC 28806, personal communication). MCI for each of the 30 stands in this study are given in Table 1. In summary, for the 30 stands, the MCI ranged from 1.63 to 2.63, with 24 stands falling within the sub-xeric range (between 1.5 and 2.4) and six stands classified as sub-mesic (between 2.5 and 3.4). Four of the sub-mesic stands had values at 2.5.

OVERSTORY COMPOSITION AND STRUCTURE

We measured 9,032 trees with d.b.h. that ranged from 4.6 to 46 inches. Thirty different species were identified in these stands. There were three *Pinus* species, dominated by shortleaf pine (*P. echinata* Mill.), with a smaller proportion of Virginia pine (*P. virginiana* Mill.) and pitch pine (*P. rigida* Mill.). There were also, on average, four eastern hemlock trees (*Tsuga canadensis* [L.] Carr.) per acre, with the majority in the pole-size class. Other species included upland oaks (chestnut oak, white oak, northern red oak, scarlet oak, southern

Table 1.—Moisture Classification Index (MCI) for each study stand on the Daniel Boone National Forest, KY. Stand numbers followed by an ‘m’ indicate initial mesic designation. Stand numbers followed by an ‘x’ indicate xeric. MCI between 1.5 and 2.4 indicates sub-xeric. MCI between 2.5 and 3.4 indicates sub-mesic. Bold, italicized text indicates stands that differed between initial and MCI classifications.

Treatment	Stand	MCI
Control	1m	2.63
Shelterwood with reserves	2m	1.63
Oak shelterwood	3m	2.33
Oak woodland	4m	2.53
Control	7m	2.35
Oak shelterwood	10m	2.29
Thinning	11m	2.45
Shelterwood with reserves	12m	2.31
Control	13m	2.21
Oak shelterwood	14m	2.54
Oak woodland	15m	2.41
Shelterwood with reserves	16m	2.23
Thinning	17m	2.11
Thinning	18m	2.54
Oak woodland	19m	2.36
Oak woodland	20x	2.47
Thinning	21x	2.58
Oak woodland	22x	2.17
Oak woodland	23x	2.29
Oak shelterwood	24x	2.44
Thinning	25x	2.39
Control	26x	2.33
Oak shelterwood	27x	2.45
Control	28x	2.50
Shelterwood with reserves	29x	2.37
Oak shelterwood	31x	2.33
Shelterwood with reserves	32x	2.31
Thinning	33x	2.12
Control	34x	2.27
Shelterwood with reserves	35x	2.30

red oak [*Q. falcata* Michx.], blackjack oak [*Q. marilandica* Muench.], post oak, and black oak), hickories (shellbark hickory [*C. laciniosa* Schneid.], shagbark hickory [*C. ovata* K. Koch.], and mockernut hickory [*C. tomentosa* Nutt.]), red maple, sourwood, and lesser amounts of species such as yellow-poplar and blackgum (*Nyssa sylvatica* Marsh.). We found no significant differences for basal area (BA, in square feet per acre) ($P = 0.072$) and stems per acre (SPA) ($P = 0.20$) among the five treatments prior to treatment implementation (Table 2). Basal area in these study stands ranged from 103.6 to 120.0 (standard deviation [std] 0.3-9.2), and SPA were 143-159 stems per acre (std 8-20). There were no significant differences for BA ($P = 0.4282$) and SPA ($P = 0.08370$) for stands originally identified as sub-mesic, which had average BA of 113.5 (std 9) and 146 (std 1.5) SPA compared to sub-xeric stands, which had BA of 110.7 (std 7.2) and 156 (std 4) SPA. There were no treatment-by-site interactions.

Table 2.—Basal area (BA, in square feet per acre) and stems per acre (SPA, minimum d.b.h. 4.6 inches) values for all five treatments, pretreatment and post-treatment, on the Daniel Boone National Forest, KY. Values with different letters within the same column are significantly different at the 0.05 level.

	Pretreatment		Post-treatment	
	BA (std)	SPA (std)	BA (std)	SPA (std)
Shelterwood with Reserves	103.6 (0.3) a	149 (13) a	22.9 (1.1) d	19 (2) d
Oak Shelterwood	106.6 (0.4) a	143 (20) a	107.5 (0.9) a	115 (11) b
Thinning	119.3 (9.2) a	150 (11) a	80.6 (7.9) b	61 (7) c
Oak Woodland	120.0 (3.2) a	153 (8) a	68.3 (4.2) c	49 (4) c
Control	110.0 (4.3) a	159 (14) a	118.2 (5.0) a	159 (14) a

Although not necessarily the goal of these treatments, four significantly different BA and SPA regimes post-treatment (Table 2) were created. As expected, control and oak shelterwood stands did not differ because the goal of the oak shelterwood treatment was to target midstory trees without creating canopy gaps. There was significantly greater residual BA in the thinning treatment compared to the oak woodland and shelterwood with reserves treatments; however, residual SPA for the thinning treatment was not significantly different from SPA in the oak woodland. The shelterwood with reserves treatment had significantly less BA and SPA compared to the other four treatments. The change in BA was greatest for the shelterwood with reserves treatment (80.8), followed by the oak woodland (51.8) and the thinning (38.7). SPA change was also greatest for the shelterwood with reserve treatment (SPA change = 130), but the amount of change did not differ from that of the oak woodland treatment (105). The oak woodland SPA change did not differ from that of the thinning (SPA change = 89). Oak BA increased from 70 to 88 percent of the total basal area for the oak woodland treatment, and from 61 to 74 percent in the shelterwood with reserves. Diameter distributions of each treatment, for the cut or herbicided and residual stand, are presented in Figure 1.

In the oak shelterwood treatment, pretreatment and post-treatment BA for the overstory trees did not differ, as the treatment targeted midstory species. We added supplemental plots in these treatments to assess those smaller-diameter trees that were subjected to the herbicide treatment. Using all stems with d.b.h. 1.6 inches or greater, we found BA changed from 100.0 pretreatment to 85.8 post-treatment, and SPA changed from 333 pretreatment to 139 post-treatment. We herbicide-treated, on average, 176 SPA that had an average d.b.h. of 3.0 inches and a range of d.b.h. of 1.6 to 9.1 inches. Of the 176 SPA treated with herbicide, 106 were red maple, 13 were yellow-poplar, and the rest included blackgum, sourwood, sassafras, bigleaf magnolia (*Magnolia macrophylla* Michx.), and serviceberry (*Amelanchier aborea* [Michx. f.] Fern.). The residual stands comprise primarily oaks, hickories, and red maple, with a lesser component of shortleaf pine, yellow-poplar, sourwood, and flowering dogwood (*Cornus florida* L.).

The thinning treatment reduced BA from 119 to 81, and SPA changed from 150 to 61. According to the Gingrich guide (1967), our stands were taken from nearly 100 percent stocked (A-line) to just above 60 percent stocked, B-line, as was the objective (Fig. 2). The majority of the removed stems were taken from the 5.5- to 9.5-in. diameter class (53 SPA), although stems were removed from all diameter classes (Fig. 1.d.). The percentage of total basal area dominated by oak species changed from 54 percent pretreatment to 64 percent post-treatment.

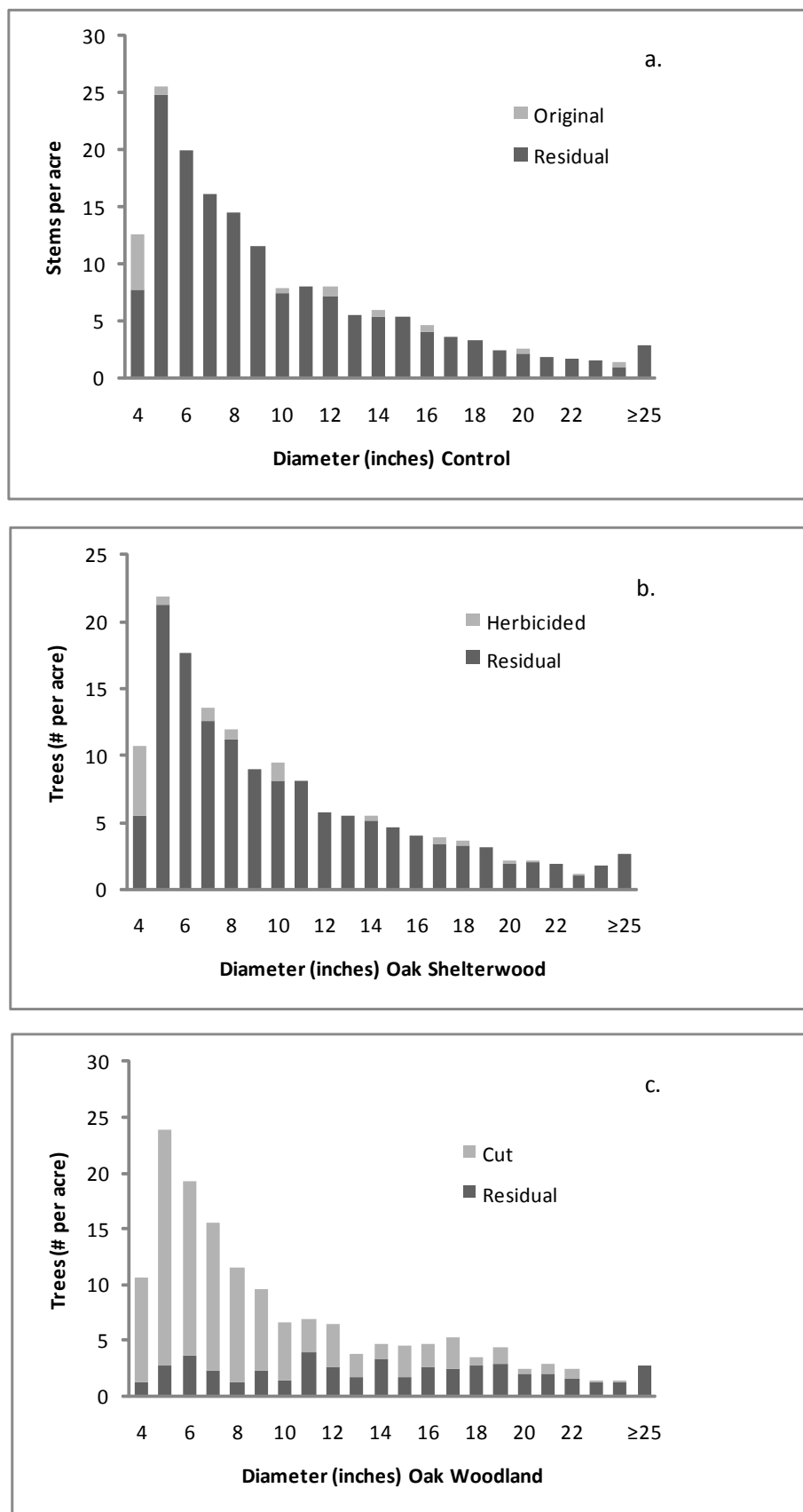


Figure 1.—Treatment diameter distributions for cut and residual trees, Daniel Boone National Forest, KY; lettered graphs depict treatments as: a. Control, b. Oak shelterwood, c. Oak woodland, d. Thinning, and e. Shelterwood with reserves. (Figure 1 continued on next page.)

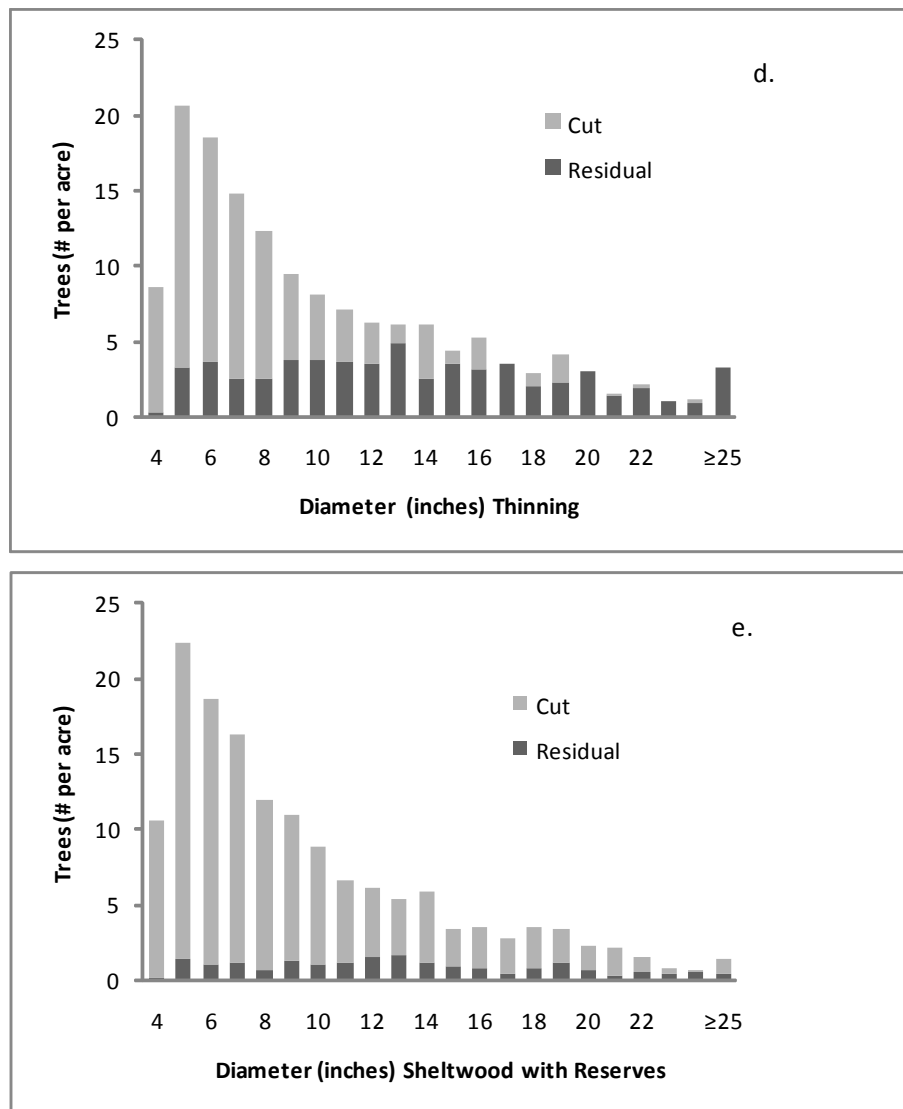


Figure 1 (continued).—Treatment diameter distributions for cut and residual trees, Daniel Boone National Forest, KY; lettered graphs depict treatments as: a. Control, b. Oak shelterwood, c. Oak woodland, d. Thinning, and e. Shelterwood with reserves.

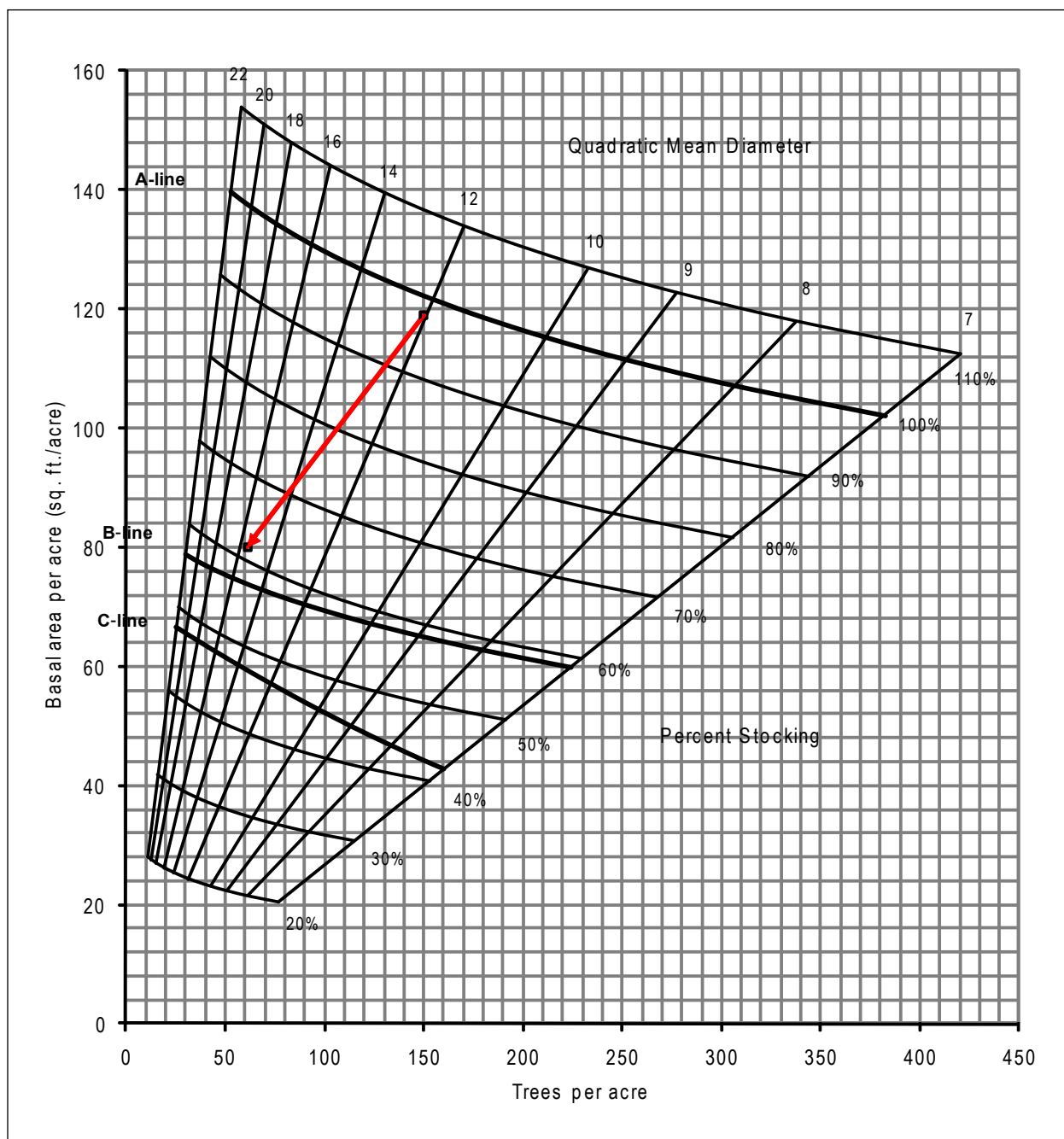


Figure 2.—Plot of stand data from six thinned stands on the Daniel Boone National Forest, KY, superimposed on Gingrich's (1967) stocking guide for central hardwood forest stands. The arrow represents the direction of stocking from pretreatment to post-treatment levels.

VIGOR, CROWN CLASS, TREE CLASS, AND DAMAGE

In the shelterwood with reserves treatment, there were on average two large sawtimber trees per acre (d.b.h. class 23.6 inches and greater), and these trees had a vigor class of 2.0 and a crown class of 3.0 (see Table 3 for vigor ratings and Table 4 for treatment values). The residual stands had only eight large sawtimber trees total; one residual white oak, which was co-dominant with a crown vigor of 1.0, had more than 200 epicormic branches post-treatment and two other residual trees had large bole wounds (600 and 18 1-square inch wounds, respectively). Medium sawtimber trees had few boles with epicormic branches (less than one tree per acre), and little damage was noted.

In the thinning treatment, medium and large sawtimber dominate the residual stand; their vigor class increased slightly, a result of the removal of lower-vigor trees, and these co-dominant trees will now receive more light from the sides (Table 4). Few epicormic branches were noted in these tree classes; less than one large sawtimber tree per acre had an average of four epicormic branches per tree, and three medium sawtimber trees per acre had two epicormics per tree. The majority of the medium sawtimber trees displaying epicormic branching were white oak, black oak, and scarlet oak. Few wounds were found on the lower boles of residual trees; three small sawtimber trees per acre had 2 to 1,000 1-square inch wounds, and less than one tree per acre was wounded in the medium and large sawtimber classes.

The oak woodland residual stands showed an increase in vigor class for all sizes except saplings (Table 4). Incidence of epicormic branching increased from one residual tree per acre to between four and six residual trees per acre in the large, medium, and small sawtimber classes. The trees displaying epicormic branching were again primarily white oak, black oak, and scarlet oak. Residual pole-sized trees had the greatest bole damage, with five SPA damaged; four SPA were damaged in the small sawtimber class, three SPA were damaged in the medium sawtimber class, and less than one SPA was damaged in the large sawtimber class.

REGENERATION

We tallied 52 woody species in the regeneration plots, which included shrubs such as *Vaccinium* spp., *Euonymus* spp., and *Viburnum* spp. Because we tallied only up to 25 stems in any one height class, our sums are not a true total of the stems per acre. On average across all stands, prior to treatment, we tallied 2,624 SPA less than 1 foot tall, 1,403 SPA greater than 1 foot in height but less than or equal to 1.5 inches d.b.h., and 85 stems greater than 1.5 inches d.b.h. For the stands that underwent a harvest, these values changed to 2,002 SPA less than 1 foot tall, 1,720 SPA greater than 1 foot tall and less than 1.5 inches d.b.h., and 50 SPA greater than 1.5 inches d.b.h. For the control and oak shelterwood treatments, seedlings less than 1 foot tall were 2,503 SPA, stems greater than 1 foot tall but less than 1.5 inches d.b.h. were 1,269 SPA, and stems over 1.5 inches d.b.h. were 83 SPA.

In the oak shelterwood treatment, oak seedling SPA were 628 (<1 foot), 191 (>1 foot tall but ≤1.5 inches d.b.h.), and 24 (>1.5 inches d.b.h.) prior to the midstory herbicide treatment, and 594, 325, and 19 SPA for the same classes, respectively, post-treatment. We will continue to follow this increase in the intermediate-sized regeneration because it is this stem size where the treatment response is expected. Red maple stems also slightly increased in this size class, from 463 SPA to 486. Red maple greater than 1 foot tall and less than 1.5 inches d.b.h. also increased in the three harvest treatments by 97 SPA in the thinning (547 SPA total post-treatment), 131 SPA in the oak woodland (621 SPA total post-treatment), and 202 SPA in the shelterwood with reserves (464 SPA total post-treatment). In this same size class, oak SPA increased to 218 in the thinning

Table 3.—Vigor rating based on tree crown and bole condition, adopted from Gottschalk (1993) to assess tree vigor on the Daniel Boone National Forest, KY.

Class	Vigor				
	% dead branches	Crown assessment	Foliage description	Epicormic branching	Stump sprouts
1 - Healthy	0-10	Healthy	Dense; green	None	None
2 - Good	11-25	Good	Density subnormal	Few	None
3 - Fair	26-50	Fair	Density and color subnormal	Some	None
4 - Poor	51-80	Poor	Density, color, and size subnormal	Heavy	None
5 - Very poor	81-100	Very poor, apparently dying	Extremely sparse	Tree living on sprouts	None
6 - Top-killed	100	Tree above ground completely dead	Does not exist	None	Live present
7 - Dead	100	Tree completely dead	Does not exist	None	None

Table 4.—Pretreatment and posttreatment stems per acre (SPA), basal area (BA in ft²/acre), tree vigor, and crown class rating by sawtimber sizes for three treatments (shelterwood with reserves, thinning, and oak woodland) on the Daniel Boone National Forest, KY. A description of vigor classes is given in Table 3.

	Large sawtimber (d.b.h. 23.6 inches and greater)	Medium sawtimber (d.b.h. 17.6-23.5 inches)	Small sawtimber (d.b.h. 11.6-17.5 inches)	Pole timber (d.b.h. 5.6-11.5 inches)	Saplings (d.b.h. 1.8-5.5 inches)
Shelterwood with Reserves					
SPA-pre	2	12	23	32	23
SPA-post	<1	3	5	6	1.0
BA-pre	8.6	27.0	25.2	26.5	3.3
BA-post	2.3	5.9	5.3	2.6	0.2
Vigor class-pre	1.9	1.8	1.7	1.9	2.2
Vigor class-post	1.5	1.3	1.3	1.8	2.1
Crown class*-pre	2.8	2.7	3.2	4.2	4.7
Crown class-post	3.3	2.7	2.8	3.9	4.2
Thinning					
SPA-pre	4	15	28	67	18
SPA-post	4	12	19	17	2
BA-pre	17.7	33.1	21.3	24.2	2.6
BA-post	14.3	26.1	21.8	7.5	0.3
Vigor class-pre	1.5	1.6	1.5	1.8	2.1
Vigor class-post	1.0	1.1	1.1	1.5	1.5
Crown class-pre	2.9	3.0	3.3	4.3	4.7
Crown class-post	2.9	3.0	3.2	4.1	4.7
Oak Woodland					
SPA-pre	4	17	24	70	23
SPA-post	3	11	13	14	3
BA-pre	14.2	38.6	27.3	24.8	3.2
BA-post	12.7	24.7	15.1	5.7	0.4
Vigor class-pre	2.1	2.0	2.0	2.1	2.3
Vigor class-post	1.2	1.2	1.2	1.4	2.2
Crown class-pre	2.8	2.8	3.1	4.1	4.6
Crown class-post	2.9	2.9	3.1	4.1	4.9

*Crown classes are defined as 1-Open Grown, 2-Dominant, 3-Co-dominant, 4-Intermediate, 5-Overtopped

and 206 in the oak woodland treatments. Red maple stems greater than 1.5 inches declined in all three harvest treatments, by 31 SPA in the thinning, 35 SPA in the oak woodland, and 43 SPA in the shelterwood with reserves, resulting in post-treatment SPA of 7, 6, and 2, respectively.

DISCUSSION

Although the study was initially designed to assess forest management response on two site types, our analysis (conducted post-stand selection) showed that the average MCI did not differ among most stands, with the majority classified as sub-xeric. The two stands that had the highest MCI were stands 1 and 28. Stand 1 had a mix of water tupelo (*Nyssa aquatic* L.), hemlock, northern red oak, and yellow-poplar, and had the only sweet birch (*Betula lenta* L.) tallied. All of these species had moisture gradient values between 3.0 and 3.5. Stand 28 had a similar species composition, minus the sweet birch. The other stands with MCI values near 2.5 contained hemlock, yellow-poplar, and flowering dogwood. In these stands, few black walnut (*Juglans nigra* L.) but many bigleaf magnolia trees were present. An examination of soil series, by measurement plot, did not reveal any predominant soil series in sub-xeric or sub-mesic stands, and most series were broadly represented across the study.

Of the primary growth factors for forest trees (site quality, stand density, and species composition), density is more easily affected by management decisions, while species composition is more minimally controlled by management. Thinning can improve average tree quality by removing defective trees or undesirable species. Gingrich's (1967) B-line stocking uses open-grown crown radius to define minimal full stocking. This release should improve future tree quality by maintaining the vigor of individual trees and allowing resources to be directed to the chosen residuals. By achieving the desired residual stocking, we expect the desired residual trees to respond to release.

Crown vigor and crown class most likely have not had enough response time since treatment implementation (1-year post-treatment assessment) to show differences from pretreatment values. However, the management regimes under study favored the retention of higher-vigor trees in all size classes, with minimal bole damage and sporadic epicormic branching. These parameters will be followed as indicators of tree vigor and potential resiliency to defoliation stress.

Although this study employed a variety of silvicultural treatments, all harvesting disturbances should be planned to provide for successful recruitment of desired species. We currently have three scenarios in which the canopy has been subjected to partial harvesting. These individual residual trees will influence the development of the next stand, and we will follow this process over time. In particular, we question whether intolerant and mid-tolerant species can exhibit sufficient height growth to survive and maintain a competitive position (Trimble 1973). Miller and others (2006) showed that low-density residual stands allowed species of all tolerances to compete as a new cohort for years, but over time reserve trees retarded the growth of reproduction in proportion to their distance from the reserve tree. For the shelterwood with reserves treatment, planning is underway to release desirable regeneration using herbicide to control competition within three growing seasons of the initial harvest. For the oak woodland treatment, periodic prescribed fires will greatly influence the regeneration. Once a decision is made to regenerate oak woodland stands, an assessment of the regeneration will be used to prepare the stand for reproduction of desired species.

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INTEGRATING STUDIES IN THE MISSOURI OZARK FOREST ECOSYSTEM PROJECT: STATUS AND OUTLOOK

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Abstract.—The Missouri Ozark Forest Ecosystem Project (MOFEP), which was started in 1989 by the Missouri Department of Conservation, evaluates the effects of forest management practices (even-aged management, uneven-aged management, and no-harvest management) on upland oak-forest components in southern Missouri. MOFEP is a long-term, landscape-level, fully replicated, and multidisciplinary project. More than 30 studies have been implemented in MOFEP. Integration of ecosystem studies has been encouraged since MOFEP was initiated to better understand how different ecosystem components interact with each other. Integration ensures that resource managers also base their decisions for improving resource management on results from a multidisciplinary research process rather than from single disciplinary studies. For MOFEP, the term “integration” has not been well defined and we offer a working definition of integration to facilitate communication and mutual understanding of the term. We provide an overview of progress made thus far and discuss challenges that need to be addressed to make integration more efficient and effective.

INTRODUCTION

Initiated in 1989, the Missouri Ozark Forest Ecosystem Project (MOFEP) is a long-term, landscape-scale experiment to evaluate effects of even-aged, uneven-aged, and no-harvest management on the flora and fauna of oak ecosystems in southern Missouri (Brookshire and others 1997). The impetus for this project was the need to investigate the impacts of forest management at an operational landscape scale on neotropical migrant songbirds as well as to study other forest ecosystem components of concern at the time. MOFEP has become one of the most comprehensive ecological investigations of forest response ever undertaken in upland oak ecosystems and serves as a nationwide model for learning from large-scale manipulations of ecosystems and landscapes. MOFEP is designed to extend through more than one 100-year rotation and it is a multi-investigator experiment of management practices administered by the Missouri Department of Conservation (MDC). Studies include soil characteristics and distribution, below- and aboveground carbon, microclimate, ground flora composition, woody vegetation composition and genetic variation of selected species, coarse woody debris distribution, hard- and soft-mast production, *Armillaria* fungi distribution and ecology, forest bird density and nesting success, herpeto-faunal communities and distribution, small mammal abundance, leaf litter arthropod communities, and abundance of leaf-chewing insects. Today, MOFEP comprises more than 30 studies conducted in cooperation with many agencies and hundreds of scientists and technicians.

The MOFEP study area is located in the Current River and Peck Ranch Conservation Areas in southeastern Missouri. The study area includes nine sites, with sizes ranging from 312 to 514 ha. The management

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practices selected for comparison, and the scale and timeframe of the study, reflect those commonly used by MDC to manage forests. Pretreatment data were collected from 1991-1995, and management treatments were applied in 1996-1997. Post-treatment data were collected after harvest. Although MOFEP has had only one entry harvest, it has provided valuable insights into how no-harvest, even-aged, and uneven-aged forest management influences plants and animals. The overall vision, guidance, oversight, and direction for MOFEP are provided by the MOFEP Steering Committee, which consists of 12 members from the major resource divisions within MDC, public institutions, and academia.

Integration has been discussed since MOFEP inception. It is clear from these discussions that the term “integration” causes a certain amount of concern and confusion among managers, researchers, and policy makers. Integration has been used in many different contexts by resource managers, scientists, and policy makers, making it difficult to know what the term means. Some view integration as something that we already practice, others believe that we have done some integration but more needs to be done, and still others believe that we have not started any integration. This variability in opinions severely limits the MOFEP Steering Committee’s ability to promote the concept. It also causes confusion among scientists when they are trying to collaborate in “integrating” their data, or conduct research that is sufficiently rigorous to provide new knowledge through “integration” and truly inform managers. According to Clark and Stankey (2006), the confusion surrounding the definition of integration is not unusual unless common understanding of integration is reached. Additionally, Hall and others (1997) stress that using words that are well defined, and hence, well understood, facilitates discussion and also facilitates better public communication while minimizing confusion and ambiguity. The variable usage of the term “integration” highlights the need to clarify what integration is in order to reach a common understanding of the term. A definition is essential for facilitating mutual understanding of the term, maximizing efficiency of data collection in the field, and communicating integration needs and progress. Through effective design and analysis of studies of different ecosystem components to determine the interactions among these components, managers are supplied useful information that enables them to understand the impacts of their decisions upon the entire ecosystem. In this paper we suggest a working definition of integration, highlight progress made so far, and discuss some challenges facing integration in MOFEP.

WORKING DEFINITION OF INTEGRATION

Integration, as its name implies, is about bringing two or more things together. In ecology, integration has been defined as combining two or more different areas of understanding or their components into new understanding (Pickett and others 2007). Integration is based on the premise that ecosystem components interact and are interdependent and that no component can function without affecting the whole. Integration focuses on the whole that comprises multiple interrelationships and interactions, rather than on pieces of problems (Clark and Stankey 2006). Integration ensures that resource managers base their decisions on results from a multidisciplinary research process that examines these interrelationships and interactions rather than on single disciplinary results.

We define integration in MOFEP as meshing of data or results from different ecosystem studies across subject or disciplinary boundaries as well as at different temporal and spatial scales. Integration represents a shift from disciplinary to multidisciplinary approaches on how we report and conduct MOFEP. It is about achieving a holistic understanding of how the ecosystem responds to management and provides an opportunity to

facilitate managing for multiple objectives consistent with MDC and public values and concerns. Integration should not be viewed as an end in itself, but rather a means to more effective decisionmaking and enhanced ecosystem benefits.

Integration can and should occur at many scales in MOFEP: fine-scale and broad-scale. Fine-scale represents integration at the plot or stand levels. Broad-scale integration is at a landscape level. There are two types of integration: integration by synthesis, which is usually at a broader landscape scale, and integration by analysis, which must be concerned with elements at the local scale as well as at the broad scale. Integration by synthesis does not aim to generate new primary knowledge, but instead seeks to add value to existing information by collating, evaluating, summarizing, interpreting, and communicating it in a useful form. On the other hand, integration by analyses can generate new primary knowledge through correlation and multi-variable analyses as well as through more complex modeling approaches. These analyses allow interactions (or correlations) between the different ecosystem components to be determined and better understood. These analyses may require more sophisticated and complex statistical approaches, which in turn require more rigorous planning and implementation of different sampling designs so that data are gathered in a manner that permits these analyses. Integration by synthesis and analysis are both important and their application will depend on the target audience and the integration questions.

Integration must be built around well posed questions and it must be developed by all resource managers, scientists, and policy makers, collaborating from the beginning. These integration questions should guide decisions about what studies to integrate, and what methods to use, and identify incentives for integration to work. Involvement of managers from the beginning provides more effective scientific direction and facilitates the use of the research results in understanding the ecosystem and in the forest management decision process.

PROGRESS TOWARD INTEGRATION

The need for integration has long been recognized since MOFEP was initiated in 1989. During this early phase of the project the concern was raised that MOFEP was concentrated on collecting information on individual ecosystem components, and that in the future, collaborative, integrated research was required (see Brookshire and others 1997, Larsen and others 1997). Current and previous MOFEP strategic plans highlight integration as a priority.

The common research sites, common landscape-scale experimental design, and breadth of research studies in MOFEP provide a good opportunity for addressing multidisciplinary questions through integration. MOFEP study results to date have resulted in 180 publications and more than 240 presentations at national and international meetings. Although many of these products have focused mainly on individual ecosystem components, they have contributed to understanding ecosystem components to forest management and have provided a foundation for integration. The limitation of presenting results from individual studies is that they do not capture the complexity of the ecosystem responses. Synthesis documents that bring together the scientific highlights of MOFEP in a fashion useful to resource managers have been developed, providing some insights on the response of the various ecosystem components. Integration by analyses also has been attempted.

INTEGRATION BY SYNTHESIS

One of the biggest challenges facing managers is to take proper account of relevant research findings in making decisions. A major problem for the potential users of research, be they managers, private landowners, or other researchers, is that they find it difficult or impossible to unearth all the relevant evidence, and decide what it means. Everyone therefore depends on good summaries of research to guide their decisionmaking process. Integration of information by synthesis is helpful in guiding this effort.

Scientists have synthesized MOFEP information from published individual disciplinary studies. An example of a synthesis of MOFEP studies is by Kabrick and others (2004). The purpose of their synthesis was to summarize what has been learned through the first 10 years of MOFEP for vegetation, birds, amphibians and reptiles, and small mammals. Their synthesis indicated that relative to the no-harvest management sites, ground flora richness, total vegetative cover, and woody vines increased and legumes decreased after harvesting. There was little difference in ground flora response between even-aged and uneven-aged treatments. No treatment effects were detected on amphibian and reptile abundances, with one exception. American toad abundance declined on all treatments; the steepest declines were observed on no-harvest treatment sites. Small mammal abundance declined on no-harvest sites yet remained the same on even-aged sites. Mature forest songbird abundance, particularly Ovenbirds, decreased and early successional songbird abundance increased in harvested sites. However, neither nest predation nor nest parasitism increased following treatments. Kabrick and others (2004) concluded that forest management objectives, including regeneration, do not appear to conflict with other management objectives, such as sustaining plant and wildlife communities, because harvesting was not necessarily detrimental to them during the first few years following an initial harvest entry into the forested sites.

Gram and others (1997) compiled a summary of pretreatment analysis of variance results found in 12 MOFEP studies (research on the genetics of woody species, snags and down wood, berry-production plants, acorn production, and surface soils; and seven studies of taxonomic groups at various levels [woody vegetation, ground flora, small mammals, leaf-chewing insects, forest interior birds, herpeto-fauna and *Armillaria*]) to present treatment and block effects. They reported that out of 57 variables assessed, treatment effects were significant for only three: ground flora species richness, density of moth caterpillar (*Dichomeris ligulella*) on white oak (*Quercus alba*), and *Rubus enslenii* abundance. They found significant block effects for 19 of 57 variables. These results showed that sites were variable prior to harvesting in 1996-97 and underscored the importance of having pretreatment information in an experiment of this scale.

Several researchers have discussed their findings in the context of other MOFEP studies. For example, Wallendorf and others (2007) integrated their findings on birds with findings from vegetation responses in the discussion section of their paper. Although they did not do integration by analyses, it was useful to explain how the bird responses might be related to vegetation responses.

Researchers have recently consolidated results that have been found following the first harvest entry into MOFEP sites. They use a matrix format that shows the response of various ecosystem components to the three forest management systems. Matrices covering overstory vegetation, ground flora, birds, reptiles, amphibians, small mammals, dung beetles, and oak insect herbivores have been developed. An example of a matrix is given in Table 1. This matrix of “winners” and “losers” has a zero denoting no treatment effect, “+” denoting a positive effect (increase) and “-” denoting a negative effect (decrease) on the species. This matrix gives busy

Table 1.—Impacts of even-aged forest management (EAM) uneven-aged forest management (UAM), and no-harvest management (NHM) on ground vegetation (<1 m in height) 2 years post-harvest. A zero denotes no treatment effect, + denotes a positive effect, and – denotes a negative effect on the species.

Species/item	EAM	UAM	NHM
Tree seedling density	0	0	0
Species richness	+	+	0
Ground cover	+	+	0
Annuals/biennials	+	+	+
Forbs	0	0	0
Graminoids	+	+	0
Legumes	-	-	0
Woody vines	+	+	0
Shrubs	0	0	0
Nonnative species	0	0	0

managers a reference that is brief and to the point and has allowed the researchers to interpret their data for the reader. The matrix gives managers something to work from in the near term and encourages them to dig deeper when they have the time and interest.

These syntheses define current scientific understanding and integrate MOFEP findings in a format that managers and administrators can use without reading the voluminous reports of individual studies. However, the integration by synthesis has been useful in summarizing findings in an easy to understand format but provides little insight into the interactions within and among the ecosystem components. They assume that immediate effects of forest management on the different ecosystem components are additive (i.e., the total is equal to the sum of the parts) and hence ignore interactions among components. On the other hand, integration by analysis allows correlations to be inferred and thus provides a more holistic understanding where the whole is more than summing up the results from the different components.

INTEGRATION BY ANALYSIS

The first integration by analysis study was by Gram and others (1997), who identified potential interactions among individual species by performing correlation analyses (using product moment correlations) on density and relative abundance of 24 species from the seven different taxonomic groups of plants and animals. They found 25 potential interactions and 80 percent of them were between species from different taxonomic groups. For example, shortleaf pine (*Pinus echinata*) and wood thrush (*Hylocichla mustelina*) densities were positively correlated ($r = 0.84$), and sassafras (*Sassafras albidum*) relative abundance was negatively correlated with spotted salamander (*Ambystoma maculatum*) ($r = -0.85$). They concluded that the predominance of intertaxa correlations confirmed the importance of integrating data among taxa because it is likely that some of these relationships influence widespread ecosystem processes. They also correlated the taxonomic groups across the year the land was acquired by MDC to explore potential effects of land-use history on taxa abundance patterns. They found correlations between six groups of taxa, including high positive correlations between woody vegetation and ground flora ($r = 0.95$), and high negative correlation between woody vegetation density and sassafras inbreeding coefficient ($r = -0.99$). These results suggested that patterns of species succession in plant communities may be directly related to land-use history.

The second study was by Gram and Sork (1999), who evaluated 1) the relationship between genetic diversity indices and density (seedlings, small trees, and large trees); and 2) the relationship between genotypic composition and density measures of three species (*Q. alba*, *Carya tomentosa*, and *Sassafras albidum*) using canonical correlation analysis. Canonical correlation analysis is a generalized multiple regression between two groups of variables. In their case, they modeled genetic diversity as the dependent variable group and population density as the independent variable group, and genotypic composition (i.e., the multilocus genotype of each individual per population) as the dependent variable group and population density as the independent variable group. They found that population density was not correlated with genetic diversity in large populations of plant species, but density was associated with genotypic composition of populations. These results indicated that populations with small densities had different genotypes than those with large densities, suggesting that preserving populations with different densities increases the chance of maintaining a variety of genotypes.

The third study was by Gram and Sork (2001), who evaluated 1) the correlation between forest structure variables (basal area, large tree density, medium tree density, small pine density) and individual genotypes for each species using composite variables generated from a canonical correlation analysis; and 2) the relationship between environmental characteristics (soil and aspect) and the distribution of genotypes for three common woody species (*Quercus alba*, *Carya tomentosa*, and *Sassafras albidum*). They tested whether genotypes differed among four soil-type/aspect classes with multivariate analysis of variance (MANOVA), treating individual genotypes as dependent variables. They found significant correlations between mean population genotypic vector and the forest structure vector in *Quercus* ($r = 0.67$), *Carya* ($r = 0.85$), and *Sassafras* ($r = 0.57$). They found that genotypic composition was associated with environmental variation but did not find any differences in genotypes among soil-type/aspect classes in any of the three species.

The fourth study used meta-analyses to determine effects of forest management on animal community diversity after the first entry harvest (Gram and others 2001). Meta-analysis is a statistical approach that facilitates integration of results across a set of studies on multiple species groups. A meta-analysis reports findings in terms of effective sizes, which in the case of this study, were estimated as standardized mean difference for each ecological group. Data used for the analysis consisted of densities or relative abundance for amphibians and reptiles, birds, small mammals, and leaf-chewing insects. They found that following first entry harvest, 1) animal communities showed an overall short-term change in response to even-aged and uneven-aged management; and 2) individual species groups responded differentially to management treatments (Table 2). For example, significant treatment effects for early successional birds were detected in 1997 and 1998, and for toads and forest interior birds in 1998. This analysis provided insight into how this first entry harvest affected multiple species groups.

The fifth study used Bayesian spatial modeling to integrate data (Sun and others 2008, Zang 2008). Sun and others (2008) mapped site index using covariates in the model. The covariates were aspect class, land type association, and soil depth. Results showed that aspect class and soil depth were both significant while land type association was less significant. Total vegetation coverage data were also analyzed with aspect class, land type association, and soil depth as covariates (Sun and others 2008). Results showed that the soil depth covariate was an important factor while the aspect class was less important when modeling the total vegetation coverage. Zang (2008) extended the work by Sun and others (2008) by spatially modeling vegetation coverage by an individual species where excess zeros exist. They developed the new model 'Zero-inflated Bayesian

Table 2.—ANOVA probabilities for treatment effects for ecological groups of species in 1997 (1 year after harvest) and 1998 (2 years after harvest) (Gram and others 2001).

Ecological Group	1997	1998
<i>Ambystoma</i> salamanders	0.603	0.268
<i>Plethodon</i> salamanders	0.917	0.620
Toads	0.086	0.018
Skinks	0.368	0.434
Small snakes	0.786	0.642
<i>Peromyscus</i> species	-	0.106
Forest interior birds	0.480	0.004
Edge/early successional birds	0.007	0.003
Free-feeding caterpillars, black oak	0.673	0.894
Leaf-rolling caterpillars, black oak	0.654	0.577
Free-feeding caterpillars, white oak	1.000	0.776
Leaf-rolling caterpillars, white oak	0.889	0.924

Spatial model' and illustrated its use on *Desmodium nudiflora* and *Cornus florida* (both with zero vegetation cover percent in many plots) to model spatial relationships among the coverage proportions of the two species collected at selected locations and predict the coverage on an unmeasured location.

Recently, researchers at the University of Missouri-St. Louis have been working on the interactions between avian and insect herbivore communities within MOFEP using regression analysis. The study is based on the idea that birds and herbivorous insects interact: birds eat herbivorous insects, and potentially control the population levels of those insects, at least in non-outbreak years. Some of the preliminary results indicate that 1) birds drive down insect populations locally; 2) birds affect insect community structure; and 3) plant species affect bird-insect interactions.

TOOLS FOR INTEGRATION – CONCEPTUAL MODELS

As part of the MOFEP 2006-11 strategic plan, the MOFEP Steering Committee developed five conceptual models: an overarching model, a physical environment submodel (Fig. 1), a human impact submodel, a fauna submodel, and a flora submodel. The overarching model defines the relationships of the primary ecosystem components while each submodel captures the feedbacks among the components in the submodel, and forest structure and composition. The models developed were a consensus of opinion among MOFEP partners on key ecosystem components and their linkages, and help in understanding ecosystem complexity. The conceptual models have been used by the MOFEP Steering Committee to identify individual studies implemented in MOFEP and the research gaps that have been determined. For example, air, water, and wildfire ecosystem components in the physical environment submodel have not been addressed. In the future, the models will be used as a basis for conducting further integration studies because identification of linkages among ecosystem components is a key to designing and conducting syntheses and analyses that provide managers with usable information.

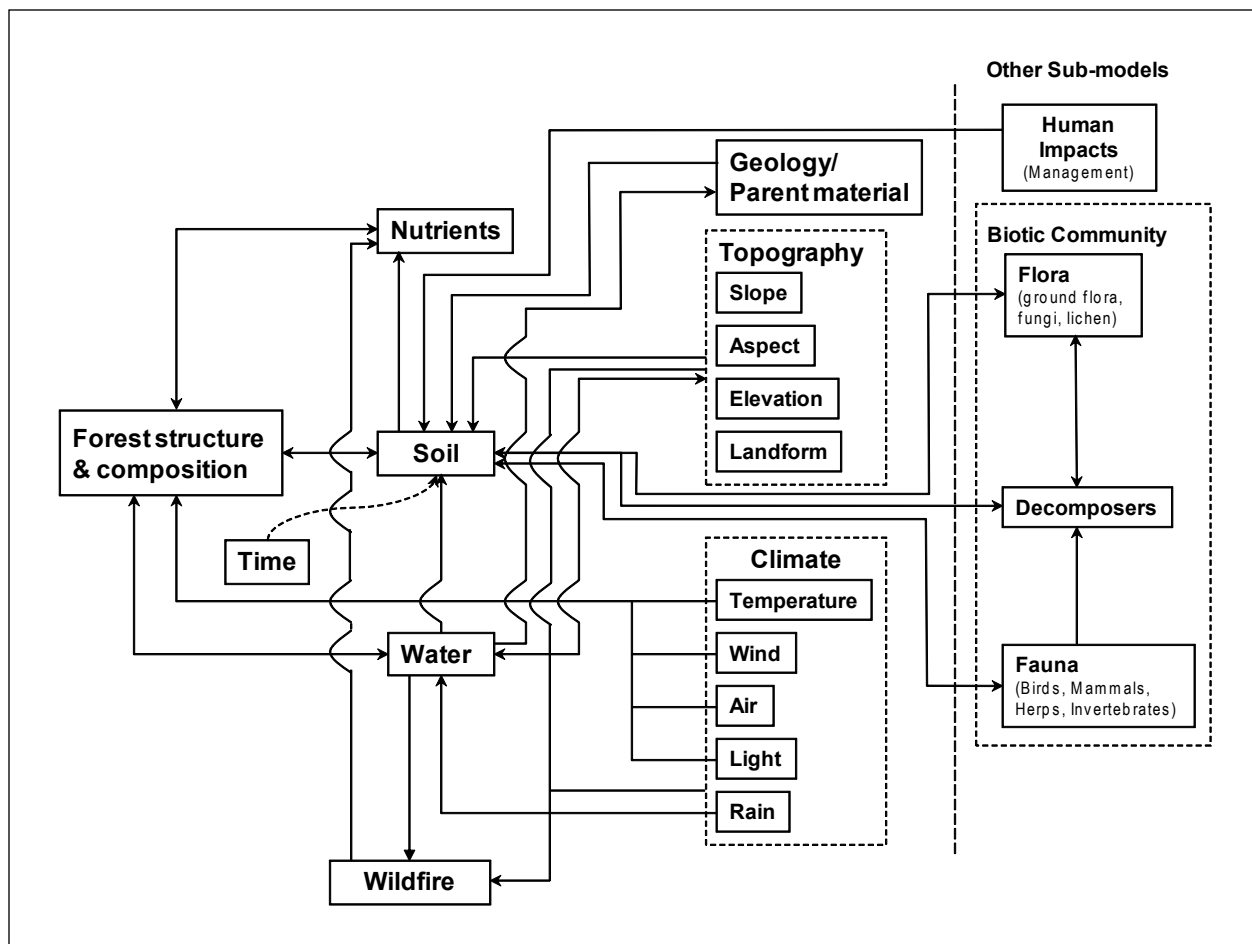


Figure 1.—An example of one of the five MOFEP conceptual models. This conceptual submodel for the MOFEP physical environment shows key linkages among physical environment components, forest structure and composition, and other submodels. Integration is central to the effort to learn more about the connections of these important ecosystem components.

INTEGRATION CHALLENGES

Recent attempts at integration have provided substantial insight into this challenging task. Furthermore, various meetings of the MOFEP researchers and the MOFEP Steering Committee members have highlighted some of the challenges. For integration to grow, many issues must be resolved. These issues can be categorized into five areas: definition, planning, data management, scale, and personnel.

DEFINITION OF INTEGRATION

As pointed out earlier in this paper, to advance integration we must be sure that the term is well defined, and hence, well understood. We have attempted to define the term in this paper. It is important that all stakeholders discuss this definition and that current and future researchers and managers agree upon and communicate common definition. We do not necessarily believe that the definition reached through this consensus will be static throughout the life of MOFEP, but it needs to be explicitly stated and referenced by all involved.

PLANNING

Planning for integration is one of the most important challenges in MOFEP. Ideally, any integration or data collection should not be done until the planning stage has been well articulated. Individual study designs rely upon knowing how the data will be analyzed and used in integration analyses. Armed with this knowledge, researchers can design the studies of individual ecosystem components to facilitate integration analyses. In reality, integration has been somewhat ad hoc with individual researchers interested in integration taking the lead. Because many of MOFEP's studies were not designed knowing what integration analyses were to be conducted, integrating data among studies has been analytically difficult. Integration done in the absence of planning also does not guarantee that priority integration questions will be addressed.

Planning for integration starts with clear and precise questions/hypotheses to justify and drive integration. These questions are essential to give context and focus to integration. According to Pickett and others (2007), integration requires that we know what we want to integrate and how to achieve it. Integration is a means to an end, requiring agreement about the goal before deciding on how to get there. Attempting to implement integration without clear questions is likely to produce poor decisions and questionable ecosystem management results. Formulating the integration questions will help select a smaller number of ecosystem components or elements whose measurements will effectively inform management. Developing these integration questions ensures that limits on scientists' time, expertise, and funding are accommodated. It also ensures that actions (methods, expertise, funding) to address these questions can be developed or identified. Limited resources and prevalence of many interactions at the ecosystem level mean that not everything can be integrated. Thus, planning will ensure that limited resources are spent on addressing the critical management questions.

Because integration questions should be driven by stakeholder interests, time spent initially on stakeholder analysis, and identifying stakeholder concerns, will avoid subsequent development of partial, inadequate integration. It is recommended that the MOFEP Steering Committee should continue to take a lead role in facilitating discussions to develop the integration questions.

DATA MANAGEMENT

Data archiving and metadata are integral components of integration. Before integration and synthesis of MOFEP studies can begin in earnest, the data need to be well defined, error-free, and readily available. MOFEP has implemented strong data management systems and policies that define clear roles of researchers in terms of data and metadata submission, and there are clear policies in terms of data access and data sharing. MOFEP data are stored in a computerized information system, making it valuable not only to the research staff but to the broader scientific community. The MOFEP data archive is a Web-based system, which allows primary investigators to enter information about their projects and publications, to upload datasets to the archive, and to view metadata and data from other projects. The MOFEP Steering Committee developed guidelines for data access. The guidelines specify who should provide the data, when the data will be available to the public, and conditions and mechanisms for access. One of the defining characteristics of the policy is MDC's commitment to open access to MOFEP data. Open access is viewed as essential for maintaining and improving knowledge of ecosystem response to management activities in Missouri and beyond. It is hoped that these guidelines will facilitate the appropriate use of MOFEP data by the bona fide scientific community, including outside scientists interested in cross-site analyses and integration.

The challenge in MOFEP is timely submission of data and quality control of the data. A future challenge will be to evaluate data collection plans with a view to revise them so that they include appropriate data to answer integration questions as they are developed.

SCALE

MOFEP data are collected at different spatial scales: leaf (e.g., oak herbivores), tree (e.g., stump sprouting), small plot (e.g., ground flora), large plot (e.g., overstory vegetation), and compartment (e.g., birds); and at different temporal scales (e.g., four times per year, annually, every 5 years). Integrating such data presents challenges. In addition, the various studies have different sampling methods (different plot locations, different plot sizes, different layouts, etc), making integration even more challenging. To make integration work, effective analytical methods must be developed to accommodate the different range of scales for the various ecosystem components.

RESOURCES

One of the challenges limiting integration is probably the physical separation of scientists – some scientists are in Oklahoma and Ohio and those in Missouri represent different agencies and are scattered throughout the state. This is not to say that scientists widely separated cannot work together, but it can be a constraint to building good working relationships needed for integration to work well.

With MOFEP, we found the greatest progress toward achieving integration occurred when a scientist was hired to work full time on integration. A major challenge was finding a qualified and experienced person in integrated research and ecosystem modeling. Few scientists have training or experience using the appropriate statistical tools needed for the analysis of complex datasets or the expertise to synthesize disparate datasets spanning multiple disciplines without assistance. Integrative research requires good administrative support by the research institution, a clear mandate guided by integration questions, and good working relationships with the principal investigators who developed data to be integrated.

We believe that developing incentives may accelerate the integration efforts. An example of such incentives is funding (e.g., multidisciplinary study proposals that specifically include integration will receive funding priority).

CONCLUSION

MOFEP integration has been encouraged since the project started. In fact, one of the bases for MOFEP is integration; therefore, MOFEP and integration are synonymous. Integration has been occurring in MOFEP, but more could be done. Integration has been constrained by a lack of common understanding of what integration is, and lack of clear plans, including formulation of the integration questions. The breadth of studies, wealth of existing data, integration experience gained from past efforts, existing integration tools, development of more sophisticated integration tools, continued discussions on how to improve integration, and commitment to continue this long-term study are sufficient to advance MOFEP for the benefit of the ecological community.

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STAND AND INDIVIDUAL TREE GROWTH OF MATURE RED OAK AFTER CROP TREE MANAGEMENT IN SOUTHERN NEW ENGLAND: 5-YEAR RESULTS

Jeffrey S. Ward¹

Abstract.—In winter 2003-04, four oak management study areas were established in Connecticut. Each study area had three 0.62-acre treatment plots: B-level thinning, crop tree, and unmanaged. Each plot was located within a 3- to 5-acre area with similar treatment. The mature red oak sawtimber stands had no prior management and were 80 to 112 years old; upper canopy oaks averaged 17.2 inches in diameter at breast height. All trees larger than 4 inches were permanently numbered and measured annually for 5 years. Basal area was reduced from 138 ft²/acre prior to harvest to 77 and 72 ft²/acre on the B-level and crop tree management plots, respectively. For all size classes combined, basal area growth over the next 5 years was greatest in the crop tree plots, followed by B-level thinning, and finally unmanaged controls. Concurrently, sawtimber basal area growth did not differ among treatments and averaged 1.3 ft²/acre/yr, suggesting management can maintain stand volume growth rates. Relative to 5-yr diameter growth of upper canopy oaks on unmanaged plots (0.9 inches), diameter growth increased by 29 percent on B-level thinning and 54 percent on crop tree management plots to 1.1 and 1.3 inches, respectively. Diameter growth increase was related to degree of crown release. Completely released trees grew more than partially released trees, which in turn grew more than trees that were not released.

INTRODUCTION

The northeastern and north-central United States has more than 13.3 million acres of 80-yr or older oak-hickory forests (Smith and others 2009). Many of these holdings, especially those less than 50 acres, are managed by family forest owners for non-commodity amenities, such as aesthetics, privacy, and wildlife (Butler and Leatherberry 2004). While owners of these mature oak forests may consider management practices, such as crop tree management, that retain non-commodity amenities to offset ownership costs (e.g., taxes and insurance), there are few guidelines beyond initiating regeneration prescriptions.

Few recent studies have focused on releasing mature oaks (> 80 yr old), in part because of earlier reports that mature trees exhibit little response to thinning (Sander 1977, Hibbs and Bentley 1983, Dale and Hilt 1989). However, most early thinning studies in sawtimber stands were areawide, low thinnings. Low thinning, with its emphasis on removing smaller trees, would not release residual trees sufficiently (on two or more sides) to initiate a positive growth response (Lamson and others 1990). More recent work has shown that high thinning, removing upper canopy competitors on at least two sides, increases diameter growth of 70- to 94-yr-old upland oaks (Smith and others 1989, Smith and Miller 1991, Miller and Stringer 2004, Ward 2008).

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The objective of this study was to compare the response of 80- to 110-yr-old oak sawtimber to B-level thinning and crop-tree management at both the stand and individual tree levels over a 5-yr period.

STUDY AREAS

In winter 2003-04, four oak management study areas were established in Connecticut in cooperation with the Division of Forestry, Connecticut Department of Environmental Protection and the Metropolitan District Commission. These mature red oak sawtimber stands had no prior management or management limited to removing dead and dying trees for local firewood harvests.

Stand ages ranged from 79 to 112 yr old (Table 1), with some scattered oaks 130 to 195 yr old as a legacy of charcoal cutting and pasturing in the 1800s. Site indices ranged from 60 to 67 ft. Stands averaged 51 sawtimber oaks per acre with a mean diameter of 17.2 inches. Red maple, black birch, and hickory accounted for 63 percent of the remaining 18/acre upper-canopy trees. Stocking ranged from 106-124 percent (Gingrich 1967) with an average of 106 ft²/ac of sawtimber (≥ 11.0 in. diameter at breast height [d.b.h.]) and 33 ft²/ac of poletimber (4.0-10.9 in. d.b.h.) (Table 2). Oak accounted for 82 percent of upper canopy basal area.

Table 1.—Mean (standard error) initial tree characteristics in an oak management study for four study areas in Connecticut. Crop trees include trees inside and outside fixed area plots, oak and non-oak sawtimber only include trees inside fixed area plots.

	Study areas			
	Hamden	MDC	TunxisD	Winsted
Crop-trees				
D.B.H. (in)	20.2 (0.3)	21.2 (0.3)	21.5 (0.4)	20.8 (0.3)
Buttlog grade	1.2 (0.1)	1.0 (0.1)	1.0 (0.1)	1.0 (0.1)
Tree height (ft)	82 (0.6)	83 (0.8)	84 (0.6)	91 (0.7)
Sawlog length (ft)	40 (0.7)	45 (0.8)	41 (0.8)	50 (0.8)
Live crown ratio (%)	30 (0.6)	31 (0.7)	32 (0.8)	28 (0.6)
N	87	87	84	77
Oak sawtimber				
D.B.H. (in)	16.3 (0.3)	18.1 (0.5)	17.0 (0.4)	17.4 (0.4)
Buttlog grade	2.1 (0.1)	1.5 (0.1)	1.8 (0.1)	1.4 (0.1)
Tree height (ft)	78 (0.7)	83 (0.9)	83 (0.8)	90 (0.7)
Sawlog length (ft)	33 (1.1)	42 (1.4)	37 (1.2)	48 (1.0)
Live crown ratio (%)	30 (0.7)	30 (1.0)	29 (0.7)	26 (0.5)
Tree age (yr)	95 (0.4)	112 (3.8)	90 (3.7)	79 (0.5)
N	98	70	114	96
Non-oak sawtimber				
D.B.H. (in)	14.3 (0.7)	13.8 (0.4)	14.0 (0.5)	14.8 (0.8)
Buttlog grade	3.1 (0.2)	2.5 (0.2)	3.1 (0.2)	2.8 (0.2)
Tree height (ft)	74 (1.7)	77 (2.0)	70 (1.5)	80 (1.6)
Sawlog length (ft)	27 (2.4)	32 (1.7)	21 (1.8)	33 (3.0)
Live crown ratio (%)	39 (2.4)	41 (2.8)	46 (3.9)	40 (3.8)
N	31	44	30	23

Table 2.—Mean (standard error) basal area (ft²/ac) of residual and harvested trees by management type and initial tree grade.

	Management prescription		
	Uncut	B-level	Crop tree
Initial levels	145 (28)	137 (38)	132 (23)
Harvested trees			
Grade 1 sawtimber	--	19 (8)	29 (5)
Grade 2 sawtimber	--	10 (3)	7 (20)
Other sawtimber	--	23 (6)	12 (3)
Poletimber	--	9 (2)	13 (2)
Total harvested	--	61 (18)	61 (11)
Residual trees			
Grade 1 sawtimber	67 (11)	45 (7)	36 (3)
Grade 2 sawtimber	16 (4)	3 (3)	5 (2)
Other sawtimber	27 (7)	12 (5)	8 (2)
Poletimber	36 (6)	17 (5)	23 (5)
Total residual	145 (28)	76 (20)	72 (12)

METHODS

PLOT ESTABLISHMENT AND DATA COLLECTION

Each of the four study areas was divided into three 3- to 5-acre blocks. Blocks were randomly assigned one of three treatments (management prescriptions): B-level thinning, crop tree release, and unmanaged controls. All harvests were conducted as commercial thinning operations. The goal for the crop tree release plots was to reduce stocking to levels similar to an appropriate thinning treatment. Harvesting on three of the plots was completed prior to the 2004 growing season. Harvest on the fourth plot was completed in early August 2004. Stand ages were determined by sanding stumps of at least 15 harvested upper canopy oaks and counting rings.

A minimum of 25 crop trees (~10/acre) were selected and measured in each treatment block at each study area before treatment assignment. Therefore, there were crop trees on all treatment blocks, but only those on the crop tree release blocks were completely released. Crop tree density was low because stands averaged 51 sawtimber oaks per acre prior to treatment. Selection criteria for crop trees were: red oak group (*Quercus rubra* or *Q. velutina*), codominant or dominant crown class, at least 17 ft to first fork, diameter ≥ 18 inches, and grade 1 butt log. Composition of crop trees was 91 percent northern red oak and 9 percent black oak. Crop trees were banded at 4.5 feet and systematically numbered with orange paint. Diameters were recorded to the nearest 0.04 inches. The following data were also recorded for all crop trees: species, crown class, tree-grade (Alerich 2000), live crown ratio (rounded to the nearest 10 percent), total height (ft), and saw log height (ft). Heights were estimated using a laser hypsometer at approximately a 45° angle of total height. The degree of release was assessed within 1 year of cutting in 10-percent increments. Trees that were not released were assigned a release factor of 0.

A 164-ft x 164-ft (0.62-ac) plot was permanently established within each block to monitor changes in stand basal area and volume growth. All trees ≤ 4-inch diameter within each plot were permanently banded at

4.5 feet and systematically numbered with red paint. Data collected for crop trees were also recorded for these trees. A total of 1,742 stems (all species) were measured prior to treatment. Diameter and crown class of all live trees were measured annually during the dormant season.

STATISTICS

Multiple regression analysis was used to examine the effects of study area and treatment on 5-yr stand basal area (ft^2/ac) growth. Tukey's HSD test was used to test differences in basal area growth among factors. Differences were judged significant at $P \leq 0.05$. Two-factor (study area, treatment) analysis of variance with initial diameter as a covariate was used to examine the overall effect of treatment on diameter growth. Tukey's HSD test was used to test differences of tree growth among diameter classes. All differences were judged significant at $P \leq 0.05$. Because of small sample sizes across treatments and plots, the effect of treatment on diameter growth was not examined for non-oak sawtimber ($n=85$) and oak poletimber ($n=39$). Oak sawtimber was split into six diameter classes for analysis: 11.0-13.9 in, 14.0-16.9 in, 17.0-19.9 in, 20.0-22.9 in, 23.0-25.9 in, and ≥ 26.0 in. For each size class, the effect of treatment on 5-yr diameter growth was determined using a two-factor (study area, treatment) analysis of variance. Similar procedures were followed to assess the influence of degree of release on diameter growth after trees were split into six release classes: 0, 10-40, 50-60, 70-80, and ≥ 90 percent.

RESULTS AND DISCUSSION

Initial stocking averaged 113 percent, ranging from 106-124 percent. Residual stand stocking after harvesting averaged 62 percent on B-level thinning plots and 60 percent on crop-tree plots. Harvesting reduced the number of sawtimber oaks from an initial 51 per acre to only 27 and 22 per acre on B-level and crop tree plots, respectively.

Harvesting removed an average of 61 ft^2/ac (44 percent) of stand basal area on both the B-level thinning and crop-tree plots (Table 2). Both treatments reduced poletimber basal area by 36 percent. However, the distribution of harvested sawtimber basal area differed among the treatments. Harvesting reduced tree-grade 1 basal area by 44 percent on crop-tree plots and 30 percent on B-level thinning plots. On the crop-tree plots, 60 percent of harvested sawtimber basal area was from tree-grade 1 stems, compared with only 36 percent on the B-level thinning plots. In contrast, nearly half of the sawtimber basal area harvested on B-level thinning plots was from grade 3 or poorer stems.

STAND GROWTH

Stand basal area growth averaged 8.3 ft^2/ac for the 5-yr period after harvesting (Table 3) and did not differ among treatments ($F=1.000$, $df=2$, $P=0.431$). Basal area growth of oak sawtimber ($F=0.259$, $df=2$, $P=0.780$), and perhaps more important, grade 1 oak sawtimber ($F=0.052$, $df=2$, $P=0.950$) did not differ among treatments. For the 5-yr period after harvesting, both managed and unmanaged stands average 6.1 ft^2/ac for oak sawtimber, of which 4.8 ft^2/ac were grade 1 oaks. This result was unexpected because while initial oak stocking averaged 68 percent, residual oak stocking was only 41 and 34 percent on B-level and crop-tree plots, respectively. Earlier studies have reported that stand volume growth following complete release can be maintained for 14 yr or longer (Beck 1987, Dwyer and Lowell 1988, Ward and others 2005).

Table 3.—Mean (standard error) 5-yr basal area growth (ft²/ac) by size class, species group, and management prescription.

Size Class	Species Group	Management prescription		
		Uncut	B-level	Crop tree
Sawtimber (> 11 in.)	Non-oak	0.3 (1.1) a [†]	-0.4 (1.1)* a	0.6 (0.2) a
	Oak	6.7 (2.0) a	6.0 (0.6) a	5.6 (0.4) a
	All species	7.0 (2.3) a	5.6 (0.7) a	6.3 (0.5) a
Poletimber (4-11 in.)	Non-oak	0.2 (0.3) a	2.5 (0.6) b	3.9 (0.3) b
	Oak	-0.3 (0.1) a	0.1 (0.1) a	-0.2 (0.3) a
	All species	-0.1 (0.2) a	2.6 (0.6) b	3.6 (0.6) b
Total (> 4 in.)	Non-oak	0.6 (1.3) a	2.1 (1.3) a	4.5 (0.4) a
	Oak	6.3 (2.0) a	6.1 (0.6) a	5.4 (0.5) a
	All species	6.9 (2.2) a	8.2 (0.7) a	9.9 (0.9) a

[†]Row values followed by the same letter are not significantly different at P < 0.05.

*Negative values indicate mortality was greater than growth.

Previous reports noted that general thinning increased stand basal area growth (Gingrich 1971, Dwyer and Lowell 1988, Ward 1991), but recommended against thinning in stands more than 70 yr old (Sander 1977, Hibbs and Bentley 1983). The current study found that B-level thinning did not decrease stand basal area growth in stands up to 115 yr old (Table 3). That managed stands had half the sawtimber oaks of unmanaged stands and had similar basal area growth rates (Table 3) suggests oak volume growth was shifted onto the larger, higher-quality residual stems. This conclusion assumes that higher quality trees were selected as residuals and saw log length did not increase more in unmanaged than managed stands.

Less information is available for stand growth following crop tree management. Stand basal area growth was increased by crop tree management in poletimber oak stands in Missouri (Dwyer and Lowell 1988) and sawtimber oak stands in Kentucky (Miller and Stringer 2004). Multi-age crop tree management in oak sawtimber stands resulted in high volume growth in Connecticut (Ward and others 2005). The initial cutting for a shelterwood prescription can result in a residual stand structure that is similar to crop tree management, especially in the larger diameter classes. Several studies have shown that stand volume growth can be maintained following a shelterwood or deferment cut (Beck 1987, Smith and others 1989, Ward and others 2005).

An alternative to initiation of regeneration prescriptions in mature oak sawtimber stands would be useful for large landowners with a management goal of developing a more balanced age structure (e.g., state forestry agencies) and small family forest owners wanting to simultaneously maintain large trees and income from harvesting. Based on this study, both crop-tree and B-level thinning can be used in 100-yr or older oak sawtimber stands to generate income without sacrificing volume growth, assuming that merchantable volume is correlated with sawtimber basal area. Other studies have reported that crop tree management improved (Dwyer and Lowell 1988) or had no adverse effect on quality of residual trees (Miller and Stringer 2004).

INDIVIDUAL TREE GROWTH

Treatment Effects

In the following section, “oak crop trees” refers to the 25 crop trees that were selected and measured in each treatment block at each study area before treatment assignment and “oak sawtimber” refers to all other oak sawtimber trees. The maintenance of stand basal area growth in managed stands (Table 3) after a significant reduction of basal area (Table 2) indicates there was increased diameter growth of residual trees (Fig. 1). Relative to unmanaged controls, both B-level thinning and crop-tree management increased diameter growth of non-oak poletimber ($F=122.33$, $df=2$, $P<0.001$), oak sawtimber ($F=11.90$, $df=2$, $P<0.001$), and oak crop trees ($F=57.62$, $df=2$, $P<0.001$). In addition, 5-yr diameter growth of crop trees on the B-level thinning plots (1.2 ± 0.04 in.) was less than that of crop-tree management plots (1.4 ± 0.04 in.). The possibility that the degree of release, rather than the plot-wide treatment, is the causal mechanism of increased growth is examined below.

During the 5-yr period, diameter growth of completely released crop trees averaged 0.28 in/yr compared with 0.19 in/yr for unreleased trees (Fig. 1). Similar growth responses were reported for 54-yr-old red oaks in West Virginia (Lamson and others 1990), 61-yr-old red oaks in Arkansas (Graney 1998), 70- to 75-yr-old white oaks in Kentucky (Miller and Stringer 2004), 75- to 80-yr-old red oaks in West Virginia (Smith and others 1989, Smith and Miller 1991), and 74- to 94-yr-old red oaks in Connecticut (Ward 2008).

In other words, completely released trees will gain an extra inch in diameter relative to unreleased trees 11 years after treatment if these growth rates continue. Extended periods (>10 yr) of increased growth for sawtimber-sized oaks after release have been reported (Sonderman 1984, Beck 1987, Graney 1998, Perkey and Onken 2000, Ward 2008). Thus, the upper limit at which northern red oaks respond to release can be extended to at least 115 years.

Unexpectedly, only 1 year elapsed before diameter growth increased for completely released crop trees (Fig. 1a). While some studies have reported a more delayed growth response for red oak sawtimber (Graney 1998, Meadows 1998, Ward 2008), diameter growth of 85-yr-old oaks increased immediately after release in North Carolina (Beck 1987).

Initial Diameter

Both B-level thinning and crop-tree management increased diameter growth of oak sawtimber, as noted above; however, the response varied among diameter classes (Table 4). Treatment had no effect on diameter growth of the smallest (11.0-13.9 in., $F=2.75$, $df=2$, $P=0.076$) and largest (>26.0 in., $F=1.84$, $df=2$, $P=0.191$) diameter classes in the study. This outcome may have been due to small sample sizes rather than an actual lack of treatment effect. Oaks with diameters between 14.0 and 22.9 inches grew more on the crop-tree management and B-level thinning plots than on the uncut control plots. For two diameter classes, 17.0-19.9 and 23.0-25.9 in., oaks on crop-tree management plots grew more than those on B-level thinning plots.

Degree of Release

Earlier studies have reported that the amount of diameter growth response of oaks is related to the proportion of the crown that is released for saplings, poles, and small sawtimber (Lamson and Smith 1978, Graney 1998, Miller and Stringer 2004, Schuler 2006, Ward 2009). This study extends the response to large oak sawtimber

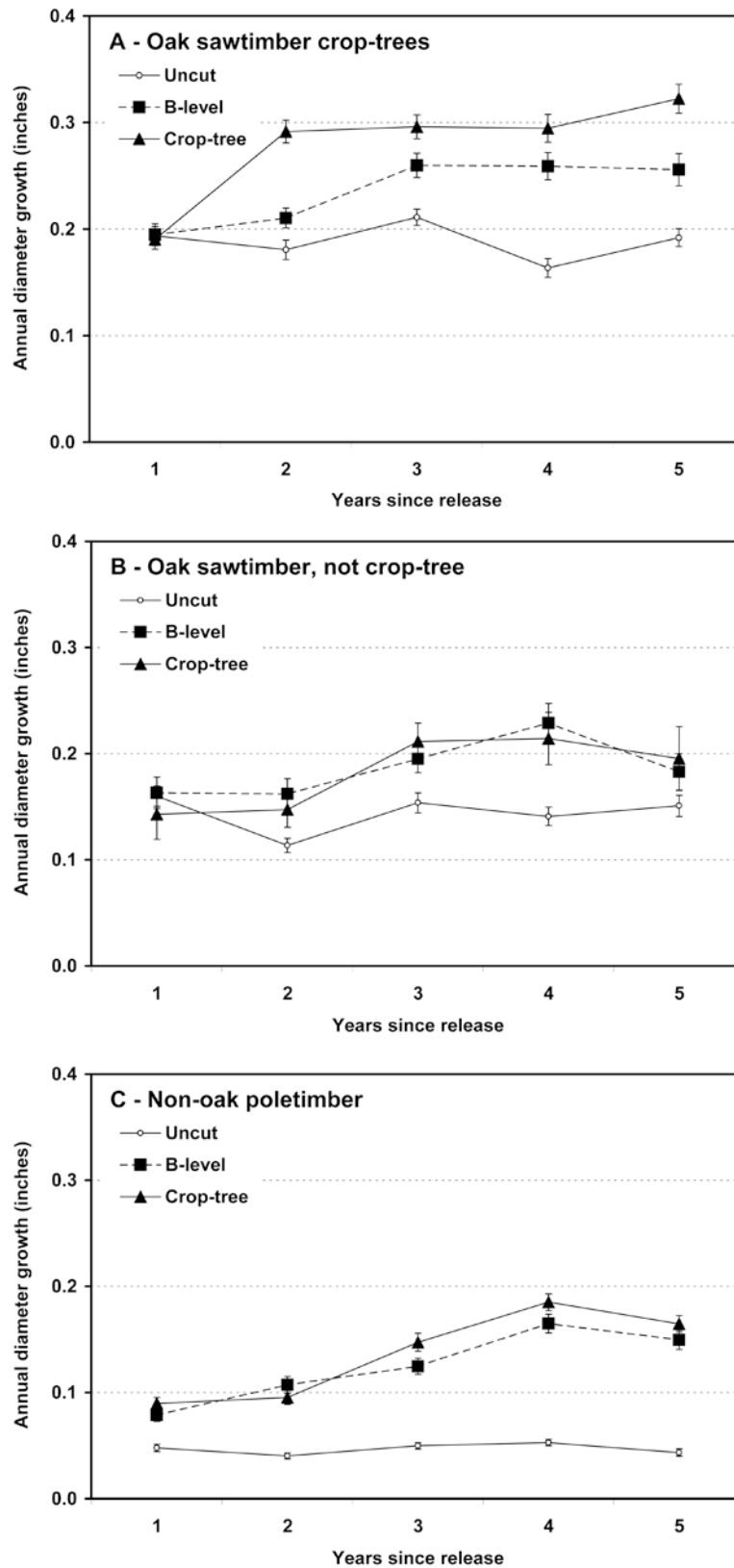


Figure 1.—Mean (standard error) annual diameter growth (inches) by management prescription for (a) oak sawtimber crop-trees, (b) oak sawtimber trees except for crop-trees, and (c) non-oak poletimber.

Table 4.—Mean (standard error) 5-yr diameter growth (in.) of sawtimber oaks by initial diameter class and management prescription.

Diameter (inches)	Management prescription			Sample size		
	Uncut	B-level	Crop tree	U	B	C
11.0-13.9	0.58 (0.05) a*	0.80 (0.08) a	0.66 (0.11) a	28	10	6
14.0-16.9	0.71 (0.04) a	0.99 (0.08) b	1.20 (0.09) b	57	16	14
17.0-19.9	0.89 (0.04) a	1.07 (0.04) b	1.28 (0.04) c	62	53	44
20.0-22.9	0.95 (0.05) a	1.25 (0.05) b	1.36 (0.05) b	39	47	46
23.0-25.9	1.06 (0.08) a	1.26 (0.10) a	1.57 (0.08) b	17	14	19
≥26.0	0.96 (0.18) a	1.22 (0.11) a	1.36 (0.10) a	3	9	10
Total	0.87 (0.02) a	1.09 (0.03) b	1.27 (0.03) c	206	149	139

*Row values followed by the same letter are not significantly different at $P < 0.05$.

as the diameter growth increase was directly related to the amount of release (Fig. 2a). The observation that a 25-percent (one-sided) release did not increase diameter growth is consistent with studies in West Virginia (Lamson and others 1990) and Connecticut (Ward 2008), indicating that trees should have a canopy release of 50 percent or more (i.e., two or more sides). Releasing trees on two sides increased diameter growth by 26 percent relative to unreleased trees (Fig. 2a).

A greater diameter-growth response was observed when more than half the crown was released. Releasing oak sawtimber by 70-80 percent (three sides) increased diameter growth by 35 percent, and ≥ 90-percent release increased growth by 53 percent, nearly 0.5 in over 5 years (Fig. 2a). Because the increased growth response should continue for 10 years or longer as noted above, diameters of completely released trees may grow nearly 2 in more over 20 years than diameters of unreleased trees, 5.3 and 3.5 in, respectively.

In contrast to oak sawtimber, partial release nearly doubled diameter growth of poletimber (Fig. 2b). Poletimber trees released less than 50 percent grew 0.73 in/5yrs compared with only 0.37 in/5yrs for unreleased trees. While releasing trees 50-60 percent further increased diameter growth to 0.96 in/5yr, poletimber diameter growth was not higher than for trees with ≥ 90-percent release.

CONCLUSIONS

This study found complete release of large, sawtimber red oak crop trees resulted in significant and sustained increases in diameter growth (Table 4, Figs. 1 and 2) without a loss of stand basal area growth (Table 3). Other studies across a wide geographical range have also reported that releasing oak sawtimber on more than two sides results in increased diameter growth (Lamson and others 1990, Graney 1998, Miller and Stringer 2004, Ward 2008). The upper age limit at which red oaks respond to thinning is at least 115 years on average-quality sites (red oak SI = 65 ft), which allows initiation of regeneration treatments to be delayed in some regions. Therefore, management prescriptions that include releasing trees on at least two sides should be considered when development of stands with a balanced age structure would require that the regeneration cuts in some stands be delayed past economic maturity. Such treatments would be appropriate for private forest owners for whom high forest cover and non-commodity attributes are important considerations, and as an entry to forest management for owners who are hesitant to initiate harvesting.

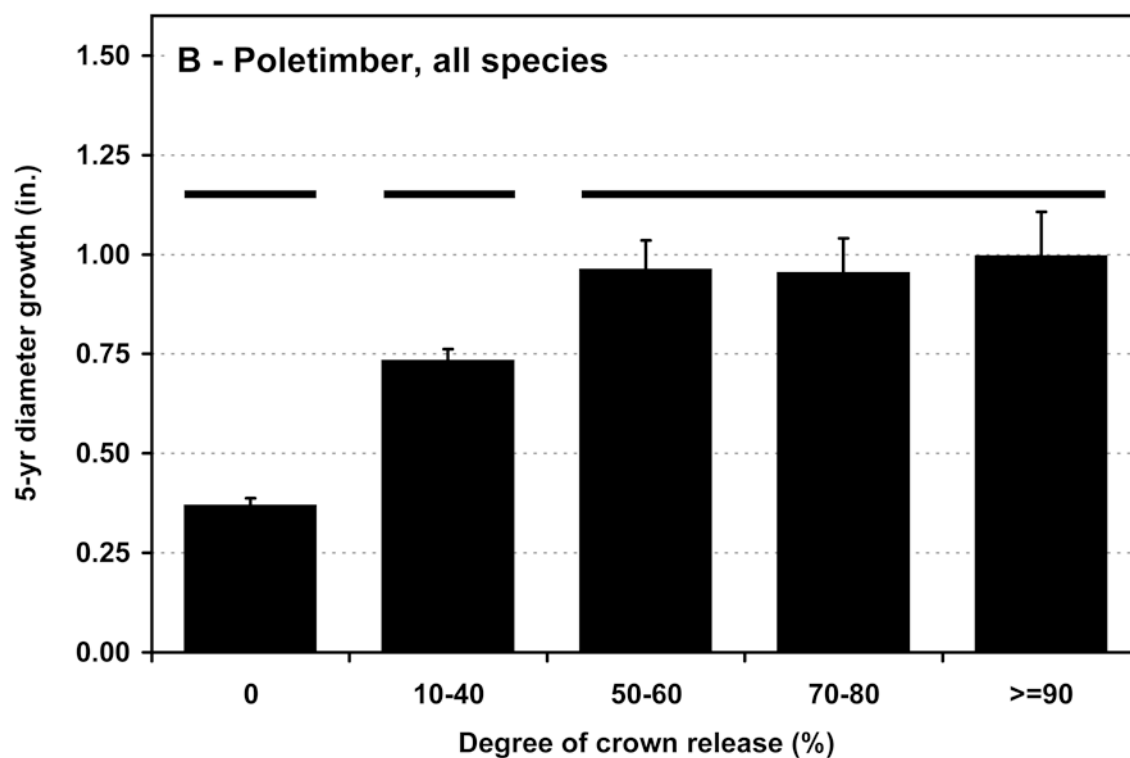
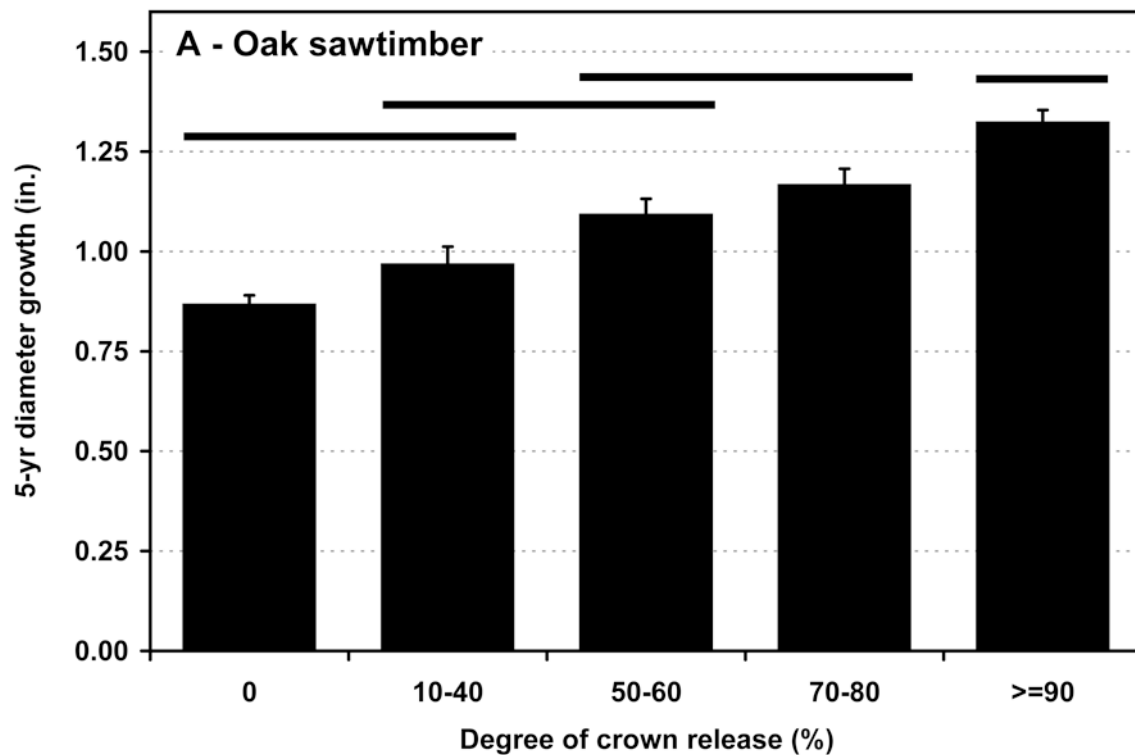


Figure 2.—Mean (standard error) 5-year diameter growth (inches) by degree of release for (a) sawtimber oaks and (b) poletimber trees. Release levels linked by horizontal lines above bars were not found different using Tukey's HSD test at $P < 0.05$.

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

THIRTY-YEAR ASSESSMENT OF RELEASED, OVERTOPPED WHITE OAKS

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Abstract.—White oak (*Quercus alba* L.) is a slow-growing species that is often overtopped in the mixed upland hardwood forests of the Cumberland Plateau. Forest managers have concerns about the potential economic value of releasing overtopped white oaks to become valuable crop trees. This study follows the growth and development of 104 overtopped white oaks after a diameter-limit cut in a forest stand on the Cumberland Plateau in Tennessee. A 30-year assessment is presented following initial and 15-year reports on diameter, height, volume growth, and stem quality of released and unreleased individuals. Released trees experienced more diameter growth when compared to unreleased trees. However, released trees developed a multitude of epicormic branches along the central bole. Over time, many of these persistent epicormics became larger, degrading log quality. Therefore, long-term results indicate that overtopped white oaks are not suitable as future crop trees. Data after 30 years are consistent with previous studies suggesting the release of overtopped white oaks perpetuates a low-quality stand that does not achieve the potential productivity of the site.

INTRODUCTION

The upland hardwood forests of the Cumberland Plateau contain a considerable number of overtopped white oaks (*Quercus alba* L.). Few studies have been developed to investigate the potential for overtopped white oaks to become high-quality timber following release. Land managers and researchers have expressed varied beliefs concerning the benefit of releasing suppressed white oaks. Most of this uncertainty is based on the diameter of trees at the time of release. Forest managers have concerns about the potential economic value of releasing overtopped white oaks to become valuable crop trees. Many practitioners assume small-diameter trees are a younger age class than overstory trees and will respond to release. Further, results of previous studies concerning this issue have varied. Some indicate that release treatments of suppressed white oaks produced lower quality and less desirable timber (Roach and Gingrich 1968, O'Hara 1986, Clatterbuck 1992). Other studies, however, demonstrate the value of releasing pole-sized white oaks where increased diameter growth is gained post-treatment (Minckler 1967, Schlesinger 1978, Dale and Sonderman 1984).

This report is an extension of two previous reports (McGee and Bivens 1984, Clatterbuck 1992) of a study that addressed the potential for released overtopped white oak trees to become merchantable after harvesting. Trees within this study were initially harvested in 1978 using a 14-inch diameter-limit cut. The effects of the release were remeasured and reported at 4 and 15 years post-treatment. The 30-year results are reported in this paper. Our objective was to quantify long-term growth effects following release of overtopped white oaks on the Cumberland Plateau.

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STUDY AREA

The study was conducted in Franklin County, TN, on the forest domain owned by Sewanee: The University of the South. The site is typical of upland hardwood forests of the weakly dissected Cumberland Plateau surface with elevations of 1,900 ft (Smalley 1982). Repeated harvesting, burning, and grazing occurred on this site for more than a century preceding the study. However, no cutting occurred during the 20 years before the study began in 1978. The study site is approximately 100 acres and contains predominately oak species, including white, chestnut (*Q. prinus*), northern red (*Q. rubra*), and black (*Q. velutina*) oaks. Other species associates include sourwood (*Oxydendrum arboreum*), yellow-poplar (*Liriodendron tulipifera*), blackgum (*Nyssa sylvatica*), and maples (*Acer* sp.). The site index at base age 50 for upland oaks ranged from 60 to 70 feet. Agronomical, climatological, and geological characteristics of the area are described by Smalley (1982). A diameter-limit harvest was conducted on 60 acres in 1978. Prior to the harvest, the site was fully stocked. Prism tally conducted following harvesting activities indicated an average residual basal area of 30 square feet per acre following the harvest. Current site stocking is around 60 percent, near the “B” line of the Gingrich Stocking Diagram (Roach and Gingrich 1968).

METHODS

Previous studies by McGee and Bivens (1984) and Clatterbuck (1992) described the use of tree age, diameter, height, and tree quality to predict response to release. This study closely follows the methodology outlined by both.

In 1978, suppressed/overtopped white oaks were released via a commercial harvesting operation. The harvesting method utilized was a 14-inch diameter-limit harvest and could be described as a “logger’s choice harvest,” where all merchantable trees were harvested. Following harvest, remaining overstory trees adjacent to sample trees were also felled. The study examined 104 overtopped white oaks after the harvest. Sample trees were selected and subjectively classified as low-, medium-, and high-quality trees based on crown form, crown size, and straightness of the stem (refer to McGee and Bivens [1984] for definitions), tagged, and recorded with an azimuth and distance from the previous tree. Of the released trees, 42 were high quality, 37 were medium quality, and 25 were low quality. In the uncut area (40 acres), 40 unreleased trees were selected and classified as high quality using the same methodology as the aforementioned released trees. The original study design was to compare the performance of high-quality released and unreleased trees only. Since medium- and low-quality unreleased trees were not a component of the original study design, comparisons of medium- and low-quality released and unreleased trees could not be evaluated. Immediately post-harvest, volume was calculated and age, diameter at breast height (d.b.h.), and total height were measured for each tree. Trees were again re-measured and calculated at 4 (McGee and Bivens 1984), 15 (Clatterbuck 1992), and 30 years post-release. Epicormic branches greater than 6 inches long on the bottom 16-foot log, hereafter referred to as the “butt log,” were tallied as an indication of bole quality. Age was initially determined using a tree corer. Height and volume were measured initially and at 4 years with a Barr and Stroud dendrometer from fixed, relocatable ground locations. At 15 years, a Suunto clinometer was used to measure height from the same locations. In autumn 2008, 30-year measurements were conducted. Total height was measured with a Haglof Vertex III hypsometer. Diameter at breast height was measured with diameter tape. Volume of total wood above stump was estimated using regression equations (Formulas 1 and 2) from Clark and others (1986).

Formulas 1 and 2.—Regression equations for estimating white oak cubic-foot volume of above-stump total tree wood.

1: Trees <11.0 in. d.b.h.:

$$Y = 0.00274(D^2Th)^{0.98507}$$

2: Trees ≥11.0 in. d.b.h.:

$$Y = 0.00171(D^2)^{1.08362}(Th)^{0.98507}$$

Where: Y = Volume of above-stump total tree wood in cubic feet.

D = Tree d.b.h. in inches.

Th = Tree total height in feet.

DATA ANALYSIS

The model for this case study was a completely randomized design with pseudo-replication. Dependent variables were height, d.b.h., volume, and epicormic count. Independent variables were quality rating and existence of release. Analysis of variance was conducted using mixed models (SAS Institute, Inc., Cary, NC) and least square means were separated using Tukey's significant difference test. Data were analyzed with a confidence interval of 0.05.

RESULTS

Diameter distributions of white oak sample trees in 1978 ranged from 2 to 12 inches (McGee and Bivens 1984). However, only 7 percent of the sample trees exceeded 10 inches in diameter. Initial heights of sample trees ranged from 20 to 70 feet. Age class of sample trees ranged from 20 to 120 years before the release treatment. Only two trees were greater than 100 years old.

The release of high-quality overtopped white oaks resulted in significantly more volume growth after 30 years when compared to unreleased trees ($p < 0.05$) (Table 1). Growth in diameter, height, and volume from initial measurements to 30 years were similar for the high- and medium-quality trees ($p < 0.05$). Diameter growth of high-quality released oaks after 30 years was weakly correlated ($r^2 = 0.59$) with volume growth. Diameter growth of high-quality unreleased oak after 30 years was moderately correlated with volume growth ($r^2 = 0.69$). High-quality unreleased oaks responded with greater volume growth at a lower diameter growth rate compared to high-quality released oaks (Fig. 1). However, released high-quality trees accumulated more volume when d.b.h. growth rates were greater. Volume growth rate for high- and medium-quality trees was 8.6 and 7.7 ft³ per decade, respectively. Volume growth rate for low-quality and high-quality unreleased trees were 5.6 and 5.5 ft³, respectively (Table 1). Of the 104 released trees, eight trees were dead and missing, standing dead, or dead and down after 30 years. After 4 years, 20 of the 104 trees experienced negative growth (McGee and Bivens 1984). Negative growth is the result of dead or drooping tops. However, these 20 trees recovered their growth rates after 15 and 30 years. In year 30, only two trees experienced negative growth. Negative growth of trees influenced the mean height growth values in Table 1.

Following harvest, all overtopped oaks showed a sharp increase in the number of epicormic branches. At year 15, the number of new epicormics decreased (Table 2). At year 30, however, the epicormics remained persistent, degrading log quality. Unexpectedly, the epicormic count for low-quality trees decreased after 30 years, while the number increased for higher-quality trees.

Table 1.—Initial characteristics of released and unreleased overtopped white oak sample trees and their average growth responses after 15 and 30 years for the overtopped white oak study, Franklin County, TN. Significance at an alpha level of 0.05 is indicated by letter for 30-year growth.

Treatment	Sample trees (n)*	Avg. age in 1978 (yrs.)	Initial mean values (1978)			Avg. 15-year growth			Avg. 30-year growth		
			Ht. (ft.)	d.b.h. (in.)	Vol. (ft ³)	Ht. (ft.)	d.b.h. (in.)	Vol. (ft ³)	Ht. (ft.)	d.b.h. (in.)	Vol. (ft ³)**
High-quality	38	53	42	5.7	5.5	5.8	3.5	7	28.7a	6.4a	25.7a
Medium-quality	35	60	42	5.9	4.8	4.8	3.2	7.5	27.3a	5.7ab	23.19ab
Low-quality	23	62	35	5.2	4.3	7	3.1	4.5	25.7ab	4.9b	16.9b
No release, High-quality	40	49	45	6.2	6.2	7.2	2	4.4	22.2b	3.5c	16.5b

* Value at the time of the 30-year assessment

** Volume estimates from Clark and others 1986.

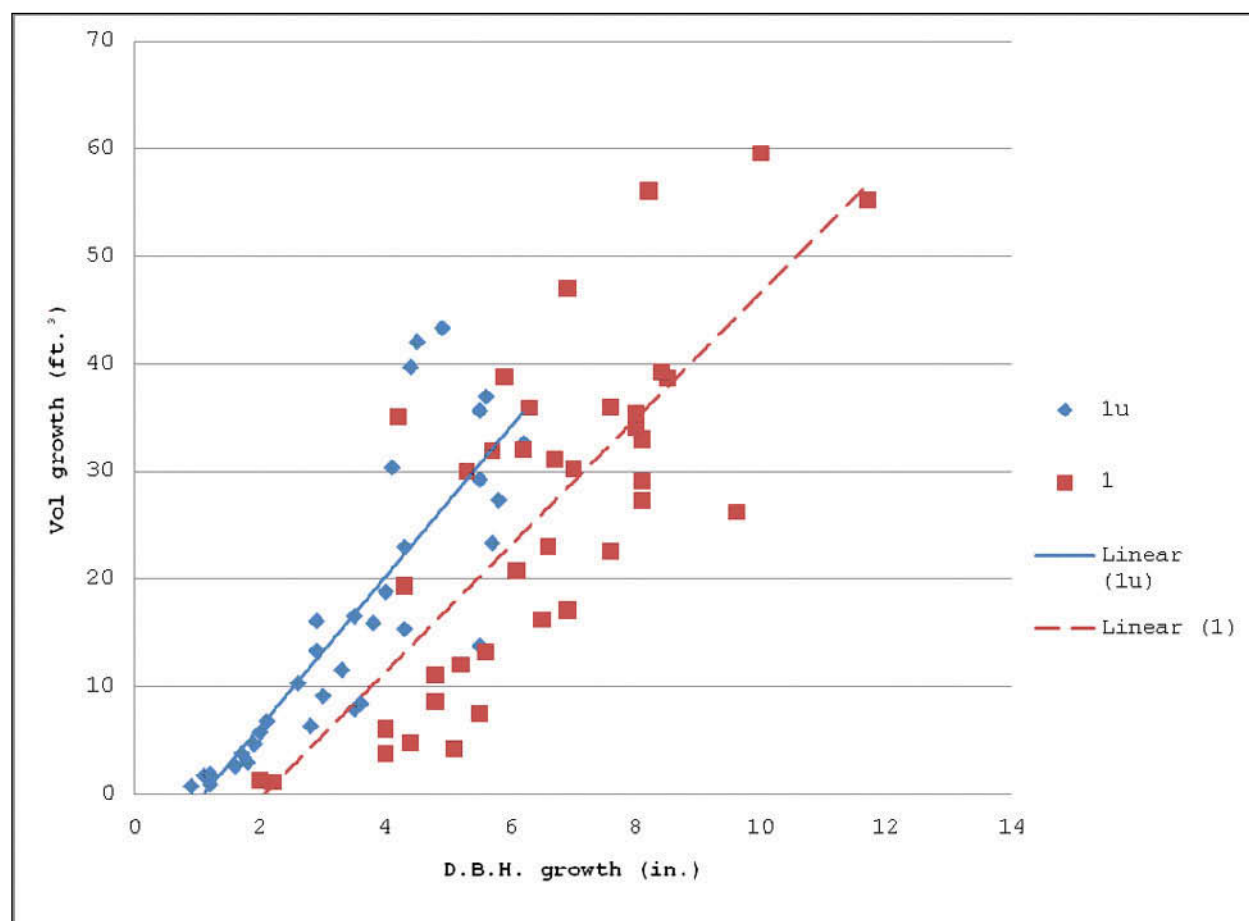


Figure 1.—Thirty-year volume growth for high-quality, released (1) and unreleased (1u) white oak in relation to growth in stem diameter for the overtopped white oak study, Franklin County, TN. Correlation for high-quality released oaks was $r^2 = 0.59$. Correlation for high-quality unreleased oaks was $r^2 = 0.69$.

Table 2.—Average number of epicormic branches (greater than 6 inches in length) on released and unreleased white oak butt logs relative to tree quality for the overtopped white oak study, Franklin County, TN. Significance at an alpha level of 0.05 is indicated by letter at 30 years.

Treatment	Average number of epicormic branches			
	Initial	4 years	15 years	30 years
High-quality	3.8	15.4	6.4	4.1ab
Medium-quality	6.4	14.9	7.2	5.1ab
Low-quality	5.9	13.2	6.9	2.7b
No release, high-quality	4.8	7.2	7.0	7.4a

DISCUSSION

Overtopped white oaks on the Cumberland Plateau will respond to release with increased growth rates. High-quality trees are associated with larger, healthier crowns as described by McGee and Bivens (1984). As a result, higher-quality trees performed better than released lower-quality trees and unreleased high-quality trees. Low-quality overtopped oaks experienced crown dieback and drooping more frequently. Upon release, an increase in epicormic branching of overtopped white oaks can be expected. Many of these initial epicormics have become persistent branches, degrading log quality. However, the epicormic count decreased after 15 years, and remained constant to 30 years. Although this study did not specifically address potential changes in tree grade or log grade by following the placement or growth of epicormic branches of released white oaks, the persistent epicormics that do remain, the larger diameters of the epicormics, the advanced age of the released trees (nearing 100 years), and the small diameters (less than 12 inches) of released, overtopped trees indicate that these trees did not increase in value after 30 years.

The overtopped white oaks in this study were released in a diameter-limit harvest that is similar to high-grading or a logger's choice harvest. Most of these overtopped oaks were greater than 75 years of age, averaged 6 inches in diameter and 40 feet tall at release, and had small, sparse, flat-topped crowns. These are not vigorous trees and did not respond well to release from overstory removal. These trees may be the result of a series of past high-grading events, as has been common practice in the hardwood forests of the Cumberland Plateau (McGee 1982) and may represent inferior genetic stock. After 30 years, these same trees averaged 10 to 12 inches in diameter and 60 to 70 feet tall. Considering that site index for upland oaks on these sites is 60 to 70 feet in 50 years, the overtopped trees took more than twice as long to reach these heights, suggesting that these trees did not realize the potential productivity of the site.

Although volumes and total heights of released and unreleased trees did increase over 30 years and more so in the last 15 years than the previous 15 years, the rate of diameter growth decreased in the last 15 years compared to the first 15 years (Table 1). The increase in volume may be skewed because the volume equation is more sensitive to the 50-percent increase in total height of small, overtopped trees rather than diameter growth. In addition, some bias in collective height measurements could have been introduced with the different instruments (dendrometer, clinometer, hypsometer) used during each of the measurement periods.

While overall volume growth was greater for high-quality released trees compared to high-quality unreleased trees (Table 1), the unreleased trees had more volume per unit d.b.h. (Fig. 1). Two causes related to study implementation are hypothesized for this relationship: (1) minor site differences within the study area; and

(2) skewed designation of high-quality trees between the cut and uncut areas during the initial tree selection process, i.e., the released trees were selected after the harvest cut and unreleased trees were selected in a fully stocked stand that was not cut.

The major decision to be made in these stands before the harvest is whether the remaining stems are capable of increasing in value (McGee 1982, Ezell 1992, Clatterbuck 2006). Is there enough acceptable growing stock remaining to continue to manage these stands, or should the stand be completely regenerated? The performance of advanced-age, overtopped white oak in this study on the Cumberland Plateau suggests that these stems will not increase in value appreciably, although volumes of small-diameter trees will increase somewhat with additional light and space resources. Thus, 30 years of growth that could be occurring on young vigorous regeneration is being sacrificed for 30 years of growth on residual, overtopped trees that will not increase in value.

In contrast to this study, a typical crop tree release is an intermediate silvicultural treatment intended to provide increased growing space to selected trees through removal of crown competition from adjacent trees (Miller and others. 2007). Generally, crop tree release is performed during the stem exclusion stage at much younger ages (10 to 30 years of age), when codominant trees are released from surrounding vegetation. The increased growth rates of the more vigorous codominant trees occur with little degradation of log quality.

CONCLUSION

Although overtopped oaks exhibited greater growth rates when released, conditions varied. Persistent epicormics, drooping tops, low-quality stems, and tree mortality were all present after release (McGee and Bivens 1984, Clatterbuck 1992). Thirty-year results were similar to 4- and 15-year results and showed that overtopped white oaks were not good candidates for release. Selection of crop trees based on form and quality offers the greatest potential benefits associated with release (Minckler 1957). However, slow growth rates of overtopped trees exhibited in our study indicate little benefit from the release (McGee and Bivens 1984, Clatterbuck, 1992). The results of this long-term study suggest that diameter-limit harvests or high-grading on the Cumberland Plateau will not create a future stand of high-quality and valuable white oaks. Although released overtopped white oaks did increase in volume, with greater volumes on higher-quality trees, these trees will remain degraded due to the persistence of epicormic branches, advanced age, and relatively slow growth.

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

MIDSTORY REMOVAL METHODS TESTED FOR TIMELY RELEASE OF NEWLY ESTABLISHED OAK SEEDLINGS UNDER OAK SHELTERWOODS

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Abstract.—The oak shelterwood method calls for the removal of shade-tolerant, undesirable midstory species to create adequate diffuse light conditions at the forest floor for oak seedling establishment, preferably timed to coincide with a large acorn crop. Injection (INJ) with an herbicide, chainsaw felling and girdling with herbicide application (SAW), and a low-volume basal bark herbicide application (BAS) were tested for efficacy and timely removal of the midstory to quickly increase light reaching newly established oak seedlings in Indiana. All three midstory treatments killed the majority of treated trees by the beginning of the second growing season. The BAS treatment produced the most rapid decline in treated tree survival across diameter classes and across the range of species, with the exception of hickory. Although all BAS-treated hickory ultimately died, INJ and SAW treatments produced more rapid control. Mean BAS treatment cost was \$0.20 per diameter-inch treated (DIT), approximately two times more costly than the SAW treatment at \$0.11 per DIT and the INJ treatment at \$0.09 per DIT.

INTRODUCTION

Timely removal of shade-tolerant, undesirable midstory species before or after acorn dispersal is important to the establishment and survival of a cohort of oak seedlings. In undisturbed mature oak forests with well developed shade-tolerant midstory canopy layers, more than 70 percent of planted oak seedlings died within 5 years of planting (Lorimer and others 1994). Loftis (1983) reported that survival of a cohort of northern red oak (*Quercus rubra* L.) seedlings in undisturbed oak forests in the southern Appalachians after 12 years was less than 10 percent. For new oak seedlings to develop into competitive oak advance reproduction, adequate understory light conditions must be obtained. Removing too much of the overstory may allow sufficient light to the forest floor to stimulate germination and establishment of fast-growing, sun-loving competitors to oak, such as yellow-poplar (*Liriodendron tulipifera* L.). Thinning from below to 60 to 70 percent of initial stand stocking, starting with the shade-tolerant midstory may provide the optimum diffuse light levels required to establish competitive oak seedlings prior to overstory removal (Loftis 1990, Larsen and Johnson 1998). This silvicultural treatment has come to be known as the “oak shelterwood method” (Stringer 2006).

Techniques and herbicides for controlling undesirable noncommercial trees in intermediate stand management operations are well established. When appropriately applied, these methods provide good control. What is less well known, however, is the rate at which these various techniques and herbicides inflict tree mortality across a range of species and diameter distributions. A new cohort of oak seedlings requires release from a well developed shade-tolerant midstory fairly quickly to ensure adequate light levels to maintain

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an acceptable proportion of that cohort into the future canopy (Phares 1971, Crow 1988). Forest managers may not recognize the need for midstory removal until after oak seedlings are observed. Thus, treatments to remove the midstory must be relatively quick as well as cost-effective.

The purpose of this study was to test three different methods for removing midstory canopy trees for overall efficacy over a range of species and midstory tree sizes, rate of mortality over time, and cost-effectiveness.

METHODS

The study was initially established in October 2005 to test the use of soil scarification for incorporating acorns into the soil to improve oak reproduction establishment. The scarification methods are described in Rathfon and others (2008). The study was established in three approximately 4-acre blocks located in three different forest tracts (stands I, N, and P) at the Southern Indiana Purdue Agricultural Center located in south-central Indiana. All three stands were oak-hickory forest cover types; overstories were dominated by mature oak, primarily white (*Q. alba* L.), northern red, and black oak (*Q. velutina* Lam.). Stand I was on a mesic to dry-mesic site on Gilpin-Berks complex soils with a southwest aspect, 20- to 30-percent slopes, and a northern red oak site index, base age 50 (SI_{50}) = ~80 feet. Stand N was on a mesic to dry-mesic site on a Zanesville silt loam soil with a northwest aspect, 15-percent slope, and a northern red oak SI_{50} = ~70 to 80 feet. Stand P was on a dry-mesic site on Zanesville and Gilpin silt loam soils with a west aspect, 12-percent slope, and a northern red oak SI_{50} = ~70 feet.

Each 4-acre block was divided into four approximately 1-acre plots and randomly assigned one of three different methods for removing midstory trees, plus a no-treatment control. A 100-percent inventory of midstory trees was conducted prior to treatment in each treatment plot. Trees were tallied by species and 1-in. diameter classes (e.g., 1-in. diameter class is 0.5 in. to 1.4 in.) as measured at breast height (d.b.h.). Species were grouped by abundance and genera. Individual species occurring in low numbers were grouped into two miscellaneous categories according to potential stature at maturity. Species groups were as follows: sugar maple (*Acer saccharum* Marsh.) (SUM), American beech (*Fagus grandifolia* Ehrh.) (AMB), elm (*Ulmus* spp.; American, slippery, and winged elm were included) (ELM), hickory (*Carya* spp.; bitternut, pignut, and shagbark hickory were included) (HIC), miscellaneous large species (including blackgum, yellow-poplar, black cherry, sassafras, white ash, hackberry, and red maple) (MISCL), and miscellaneous small species (including American hornbeam, flowering dogwood, eastern hophornbeam, redbud, and pawpaw) (MISCS).

Treatments were applied from June 29 to July 7, 2006, following emergence of a new cohort of oak seedlings. To provide consistency, one applicator applied all treatments. For the injection treatment (INJ), the applicator used an ax with a 1.75-in. bit to make evenly spaced incisions around the trunk circumference at approximately 3 to 4 feet above the ground, one for each inch in tree diameter. Approximately 3 ml of undiluted Pathway^{®2} herbicide (5.4 percent picloram [4-amino-3,5,6-trichloropicolinic acid], 3 percent acid equivalent [ae] + 20.9 percent 2,4-D [2,4-dichlorophenoxyacetic acid, triisopropanolamine salt], 11.2 percent ae) was delivered to each incision using a piston pump backpack sprayer and gunjet nozzle with adjustable cone tip at a pump pressure of less than 20 lbs per in.² In a second treatment a Stihl MS 200T professional arbor chainsaw (2.15-cu. in. displacement) was used to make a single, continuous girdle cut around the circumference of trees ≥4 in. d.b.h. Girdle cuts were no more than 1 in. deep. Smaller stems were felled. The

²Pathway[®] and Garlon 4[®] are registered trade names of Dow AgroSciences.

same Pathway® herbicide as was used with the INJ treatment was applied to the girdle cut or cut stump using an ordinary garden pump spray bottle. Herbicide was sprayed to wet the cut surfaces, but not to the point of runoff. A low-volume basal bark (BAS) herbicide spray was applied using a diaphragm pump backpack sprayer with adjustable cone nozzle. A 20-percent triclopyr (61.6 percent 3,5,6-trichloro-2-pyridinyloxyacetic acid, butoxyethyl ester, 44.3 percent ae) solution diluted in AX-IT³ oil-surfactant low-volume basal oil was sprayed on the lower 12 to 15 in., and around the full circumference of treated stems. Finally, a no-removal control treatment was maintained. The objective of the mid- and understory removal was to remove the low shade component while maintaining intact the high shade of the extant overstory. All midstory trees ≥ 1 in. d.b.h. were treated, with the exception of a small number of oaks or occasional shade-tolerant trees growing in overstory canopy gaps. All treated dead trees were tallied by species and 1-in. diameter classes in each treatment plot at 1, 2, 4, 11, 12, 16, and 24 months after treatment (MAT), between July 2006 and June 2008.

A generalized linear model was initially fit to the number of live midstory stems surviving in each treatment plot i to provide parameters for fitting generalized estimating equations in a repeated-measures analysis. Parameter estimate significance was determined at $\alpha = 0.05$. This model is essentially a repeated measures analysis of variance, except that errors are assumed to be Poisson, rather than normally, distributed. The Poisson model is more appropriate for count data. The predictors were midstory control treatment (INJ, SAW, BAS), tree species groups (AMB, ELM, HIC, SUM, MISCL, and MISCS), and diameter classes (1 to 9 in. with the 9-in. class including all trees 9 in. and larger). To account for problems associated with potential pseudoreplication, stand was initially included in the model as a predictor variable to test for random effects. Random effects due to stand were insignificant; therefore stand was dropped from the final model.

In this hierarchical design, the experimental unit is the finest level of resolution for which the data were collected—in this case, the individual stem counts for each treatment, species group, diameter class, and stand combination. A full-factorial model was fit initially, including all possible interaction terms. Interaction terms were subsequently dropped if the associated regression coefficients were not significant ($\alpha = 0.05$). Data were reported as proportion of stems surviving. Labor time, herbicide volume dispensed, and saw fuel and bar oil volumes used were recorded for each treatment plot. Cost data were calculated by diameter inches treated (DIT). DIT was calculated as the sum of the d.b.h. measurements of all treated trees in each treatment plot. Cost analyses were performed using a one-way ANOVA, using the Ryan-Einot-Gabriel-Welsch Multiple Range Test Means Separation procedure at $\alpha = 0.05$. Analyses were performed using SAS, version 9.1.3 (SAS Institute, Cary, NC).

RESULTS

STAND COMPOSITION

Across the three stands, oak basal area averaged 89 ft²/acre. Oak composed 67 percent of the overstory basal area and 57 percent of the total basal area. Midstory trees accounted for 24 to 29 ft² of basal area per acre (Table 1) or 15 to 19 percent of the total basal area across all three stands. A total of 3,953 midstory stems ranging from 1 to 15 in. d.b.h. were tallied across all three stands. Only 118, less than 3 percent, of midstory stems were not treated, the majority of these being oak species. Stands I and N had similar midstory tree diameter distributions (Fig. 1). Stand P deviated a little from the other two in that it had smaller numbers of the 1-in. diameter class, had no stems greater than 11 in. d.b.h., and carried larger numbers of stems in the 3- to 6-in. diameter classes.

³AX-IT® is a registered trade name of Townsend Chemical.

Table 1.—Midstory stem and basal area stocking for three mature upland oak stands (I, N, and P) in southern Indiana receiving midstory removal treatments.

Midstory Stocking	Tract			Mean
	I	N	P	
Midstory Stems	(no. stems/acre)			
Sugar maple	176	268	304	248
American beech	48	3	6	20
Elm spp.	30	4	3	12
Hickory spp.	3	10	19	11
Miscellaneous large	21	11	21	18
Miscellaneous small	45	29	12	29
Total	324	324	364	338
Midstory basal area	(ft ² /acre)			
Sugar maple	12.9	19.5	23.9	18.7
American beech	3.7	0.2	0.1	1.4
Elm spp.	2.0	0.1	0.1	0.8
Hickory spp.	0.7	2.0	2.0	1.5
Miscellaneous large	4.1	2.1	3.0	3.9
Miscellaneous small	1.0	0.3	0.2	0.5
Total	24.4	24.2	29.2	26.8

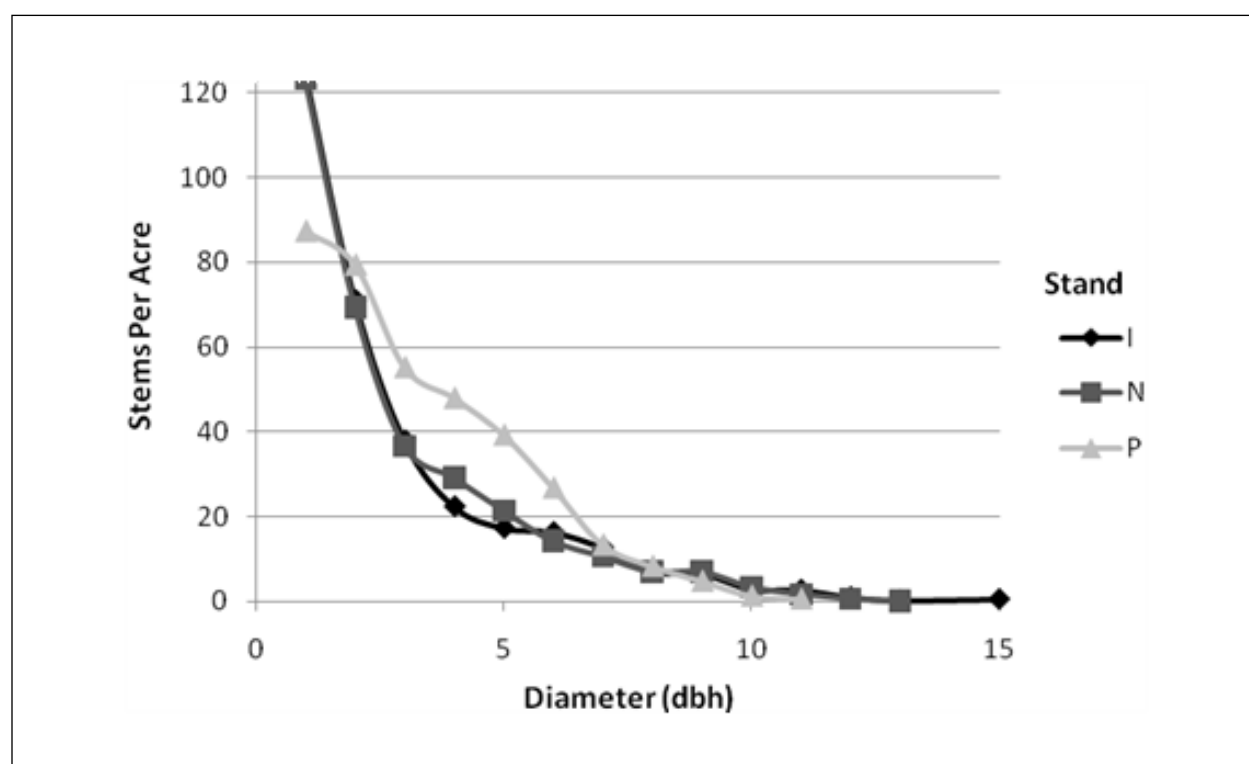


Figure 1.—Pretreatment midstory diameter distributions for three oak-dominated stands in southern Indiana receiving midstory removal treatments.

Sugar maple was overwhelmingly the most abundant midstory species in stands N and P. Across all three stands, sugar maple accounted for 73 percent and 70 percent of total number of midstory stems and midstory basal area, respectively (Table 1). American beech was the next most abundant species, making up 6 percent of midstory stems.

OVERALL EFFECTS

All three midstory control treatments resulted in fewer live stems compared to the untreated control (CON) beginning 1 MAT (Table 2). All MAT had fewer live stems than the pretreatment (0 MAT). There were overall slight differences in survival rates between some species groups. However, there were significant treatment (TRT) x species group (SPP) interactions. Overall survival rates were higher with increasing d.b.h. Treatment differences between d.b.h. classes decreased over time. However, there were significant TRT x D.B.H. interactions. These will be explored further below. By 16 MAT, survival for all three treatments was at or near zero (Fig. 2).

Table 2.—Generalized estimating equation parameter estimates for the effects of midstory removal treatment (TRT), species (SPP), diameter class (d.b.h.), and months after treatment (MAT) on treated stem survival (n = 3,834)

Parameter			Estimate	SE	Z	Pr > Z
Intercept			0.7026	0.0738	9.52	<.0001
MAT	1		-0.2597	0.0434	-5.98	<.0001
	2		-0.6024	0.0808	-7.45	<.0001
	4		-0.8559	0.1138	-7.52	<.0001
	11		-1.0372	0.1331	-7.79	<.0001
	12		-1.0963	0.1420	-7.72	<.0001
	16		-1.2414	0.1668	-7.44	<.0001
	24		-1.2468	0.1677	-7.44	<.0001
TRT	INJ ^a		-0.4359	0.1096	-3.98	<.0001
	SAW		-0.4673	0.0721	-6.48	<.0001
	BAS		-0.5958	0.0888	-6.71	<.0001
SPP	AMB ^b		-0.0153	0.0118	-1.29	0.1966
	SUM		-0.0164	0.0065	-2.53	0.0115
	ELM		-0.0276	0.0136	-2.03	0.0423
	HIC		0.0080	0.0087	0.92	0.3599
	MISCL		-0.0492	0.0254	-1.94	0.0524
d.b.h.	1		-0.0201	0.0097	-2.06	0.0390
	2		-0.0373	0.0128	-2.92	0.0035
	3		-0.0401	0.0183	-2.19	0.0286
	4		-0.0545	0.0207	-2.64	0.0084
	5		-0.0209	0.0097	-2.16	0.0308
	6		-0.0147	0.0084	-1.76	0.0783
	7		-0.0041	0.0071	-0.58	0.5599
	8		-0.0323	0.0277	-1.17	0.2439
Interactions	INJ	AMB	0.5163	0.1062	4.86	<.0001
	INJ	SUM	0.0938	0.0827	1.13	0.2565
	INJ	ELM	-0.3810	0.1259	-3.03	0.0025
	INJ	HIC	-0.5737	0.1217	-4.72	<.0001
	INJ	MISCL	-0.5407	0.1328	-4.07	<.0001
	SAW	AMB	-0.0383	0.0566	-0.68	0.4985
	SAW	SUM	0.0520	0.0210	2.48	0.0132
	SAW	ELM	-0.3434	0.2414	-1.42	0.1549
	SAW	HIC	-0.6068	0.1146	-5.30	<.0001
	SAW	MISCL	-0.4845	0.1383	-3.50	0.0005
	BAS	AMB	-0.0554	0.0731	-0.76	0.4480

(Table 2 continued on next page.)

Table 2 (continued).—Generalized estimating equation parameter estimates for the effects of midstory removal treatment (TRT), species (SPP), diameter class (d.b.h.), and months after treatment (MAT) on treated stem survival (n = 3,834)

Parameter			Estimate	SE	Z	Pr > Z
Interactions	BAS	SUM	-0.0829	0.0660	-1.26	0.2092
	BAS	ELM	-0.2547	0.0988	-2.58	0.0100
	BAS	HIC	0.3292	0.0877	3.75	0.0002
	BAS	MISCL	-0.2597	0.0882	-2.94	0.0033
	INJ	1	-0.9593	0.0876	-10.95	<.0001
	INJ	2	-0.8443	0.0902	-9.36	<.0001
	INJ	3	-0.7405	0.0781	-9.48	<.0001
	INJ	4	-0.4082	0.0782	-5.22	<.0001
	INJ	5	-0.2193	0.0851	-2.58	0.0100
	INJ	6	-0.0561	0.0893	-0.63	0.5300
	INJ	7	-0.1456	0.1088	-1.34	0.1810
	INJ	8	-0.0609	0.1166	-0.52	0.6018
	SAW	1	-1.3393	0.0726	-18.46	<.0001
	SAW	2	-1.3155	0.0733	-17.95	<.0001
	SAW	3	-1.2391	0.0741	-16.72	<.0001
	SAW	4	-0.6278	0.1045	-6.01	<.0001
	SAW	5	-0.4418	0.1131	-3.91	<.0001
	SAW	6	-0.2432	0.0874	-2.78	0.0054
	SAW	7	-0.2010	0.1296	-1.55	0.1207
	SAW	8	-0.1198	0.0969	-1.24	0.2164
	BAS	1	-0.7943	0.0806	-9.86	<.0001
	BAS	2	-0.6127	0.0638	-9.60	<.0001
	BAS	3	-0.4548	0.0707	-6.43	<.0001
	BAS	4	-0.3110	0.0673	-4.62	<.0001
	BAS	5	-0.2161	0.0727	-2.97	0.0029
	BAS	6	-0.1852	0.0727	-2.55	0.0108
	BAS	7	-0.1909	0.0986	-1.94	0.0529
	BAS	8	-0.0074	0.1158	-0.06	0.9492

^aMidstory treatments (TRT): Injection (INJ), chainsaw girdle (SAW), and low- volume basal bark spray (BAS).

^bSpecies groups (SPP): American beech (AMB), sugar maple (SUM), elm spp. (ELM), hickory spp. (HIC), and miscellaneous large trees (MISCL).

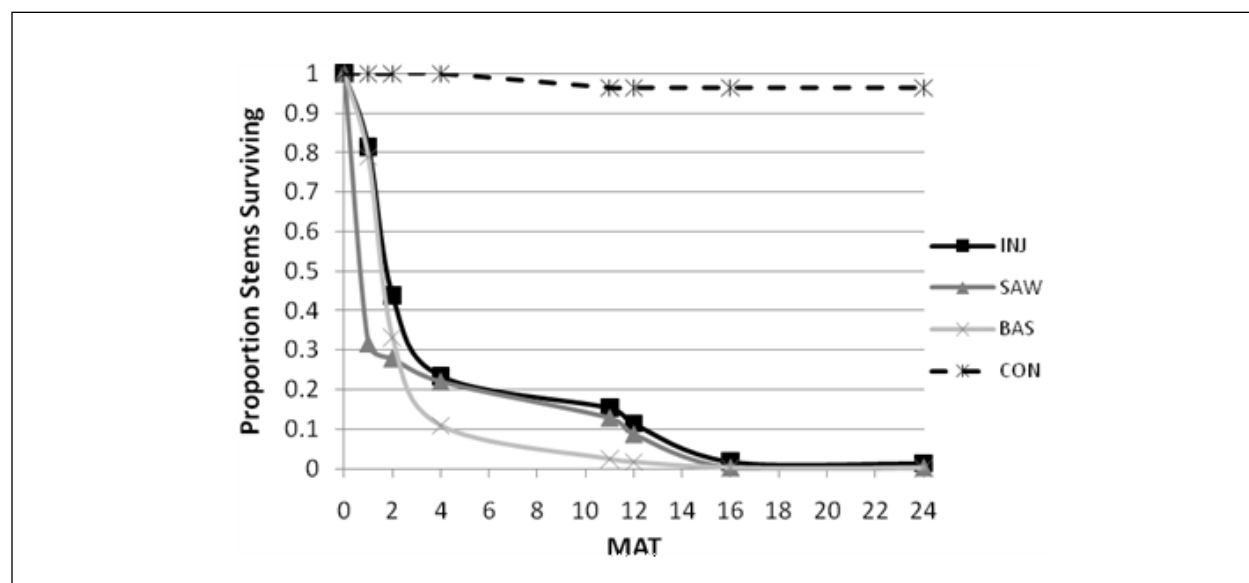


Figure 2.—Mean proportion of stems surviving over the 24-month period following treatment (MAT) with injection (INJ), chainsaw cutting/girdling (SAW), and low-volume basal bark (BAS) midstory removal methods, and a no-treatment control (CON).

MIDSTORY TREATMENT (TRT) EFFECTS

Midstory stems were reduced to 32-percent survival 1 MAT in the SAW treatment, primarily because all stems less than 4 to 5 in. d.b.h. were cut down (Fig. 2). By the end of the first growing season, 4 MAT, the BAS treatment reduced the number of live stems to 11-percent survival, surpassing both INJ and SAW treatments in early efficacy.

Early in the second growing season, 12 MAT, 98 percent of BAS-treated stems were dead. Only 11 and 9 percent of INJ- and SAW-treated stems, respectively, remained alive at 12 MAT. By the end of the second growing season, 16 MAT, only 2 percent of INJ-treated stems remained alive, while all of SAW-treated stems were dead. By 24 MAT, a remnant 14, or 1 percent, of INJ treated trees ranging in size from 6 to 9+ in. d.b.h. were still alive.

SPECIES DIFFERENCES

Significant TRT x SPP interactions indicated further differences in the rate of stem mortality among species in response to the treatments (Table 2). INJ produced the most rapid declines in tree survival among the elm, hickory, and MISCL species groups (Fig. 3). Elm survival at 1 and 2 MAT was 42 and 10 percent, respectively. By 4 MAT, all INJ elm stems were dead. Although hickory did not respond immediately to INJ 1 MAT, only 19 percent survived at 2 MAT. By 4 MAT, only 5 percent of hickories survived; complete mortality occurred by 12 MAT. Beech and sugar maple survival rates under the INJ treatment were higher than for other species groups. Their temporal pattern of mortality was similar to each other, with beech having slightly higher rates of survival over time.

Differences in SAW treatment survival 1 MAT between species groups reflect differences in relative proportions of stems in the smaller d.b.h. classes. For instance, MISCS had 0-percent survival 1 MAT because all stems in this species group were less than 5 in. d.b.h. Following 1 MAT, beech and sugar maple mortality rates slowed, compared to other species groups, to 29 and 25 percent survival, respectively, 4 MAT.

Two MAT the BAS treatment had reduced survival to between 9 and 33 percent for all species but hickory. For elm, MISCL, and MISCS species groups, the BAS treatment reduced survival to the same degree as the INJ and SAW treatments 4 MAT. For beech and sugar maple, however, the BAS treatment surpassed the INJ and SAW treatments by reducing stem numbers to 4 and 11 percent survival, respectively, 4 MAT. Hickory responded more slowly to the BAS treatment than did all other species. By the end of the first growing season, 4 MAT, hickory still had 67-percent survival. By 12 MAT, 25 percent of hickory stems remained alive. However, only 4 percent of hickory stems survived to 16 MAT; 100-percent control occurred by 24 MAT with the BAS treatment.

D.B.H. CLASS DIFFERENCES

Significant TRT x D.B.H. interactions indicated differences in the rate of stem mortality among d.b.h. classes in response to the treatments (Table 2). In general, the smaller d.b.h. classes had lower survival rates beginning 1 MAT (Fig. 4). The SAW treatment reduced small d.b.h. stem numbers by nearly 100 percent 1 MAT because, as stated earlier, virtually all stems less than 4 in. d.b.h. were cut down. Some 4- and 5-in. d.b.h. stems were felled and some girdled in the SAW treatment, accounting for much of the stem reduction seen 1 MAT in these d.b.h. classes. In larger d.b.h. classes (> 6 in.), stem survival was similar between treatments 1 MAT, ranging between 85 and 100 percent.

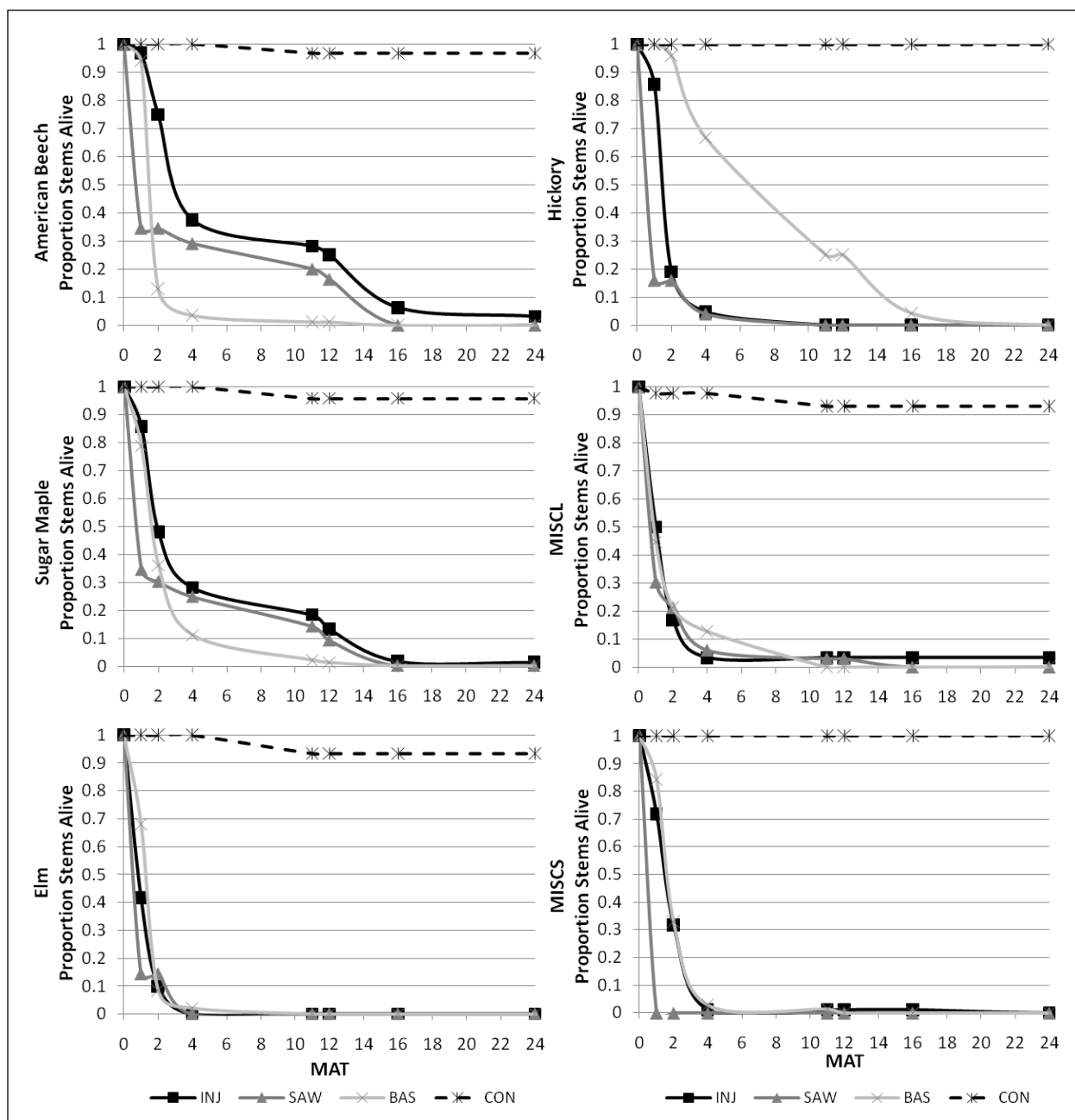


Figure 3.—Mean proportion of stems surviving for six species or species groups over the 24-month period following treatment (MAT) with injection (INJ), chainsaw cutting/girdling (SAW), and low-volume basal bark (BAS) midstory removal methods, and a no-treatment control (CON). MISCL are miscellaneous tree species capable of attaining forest tree stature and include blackgum, yellow-poplar, black cherry, sassafras, white ash, hackberry, red maple, and others. MISCS are miscellaneous tree species that typically are of low stature and include American hornbeam, flowering dogwood, eastern hophornbeam, redbud, pawpaw, and others.

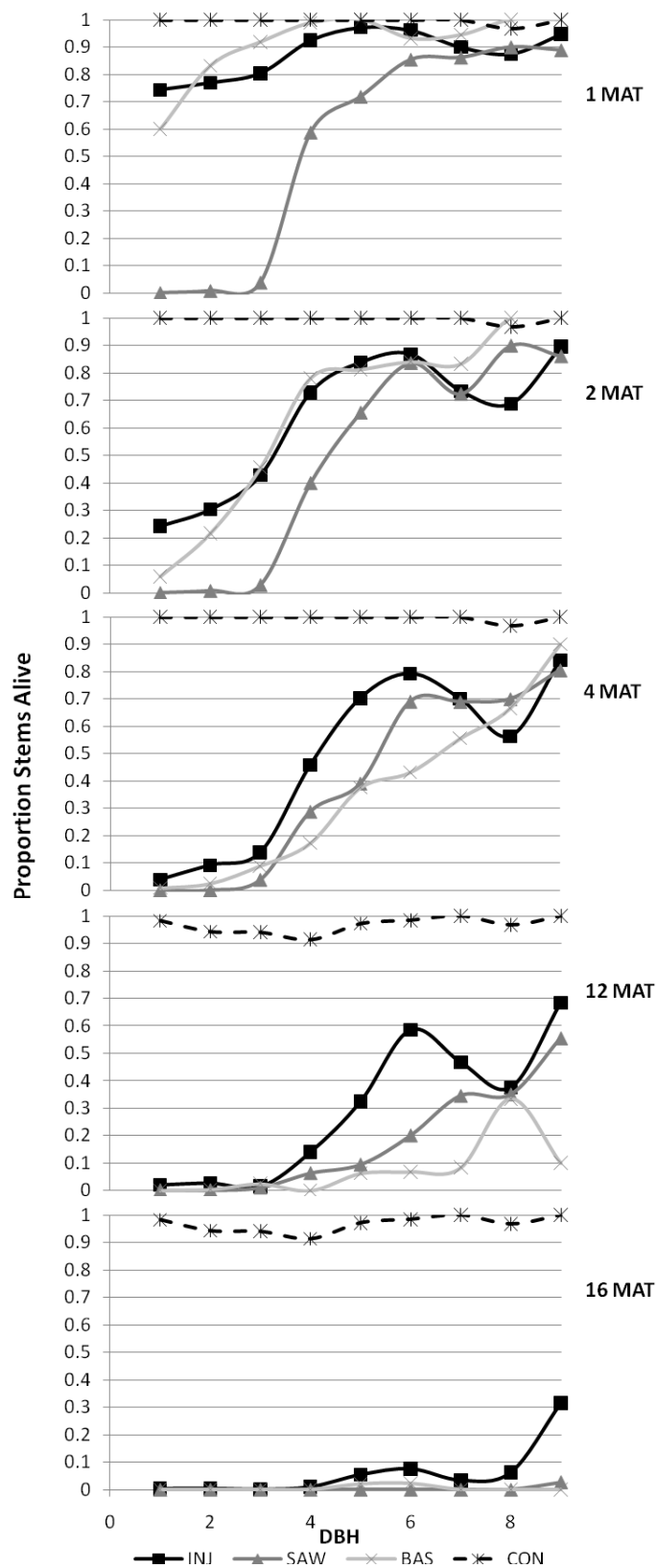


Figure 4.—Mean proportion of stems surviving across a range of stem diameters (d.b.h.), 1, 2, 4, 12, and 16 months following treatment (MAT) with injection (INJ), chainsaw cutting/girdling (SAW), and low-volume basal bark (BAS) midstory removal methods, and a no-treatment control (CON).

By the end of the first growing season, 4 MAT, most stems less than 4 in. d.b.h. for all three treatments were dead. The BAS treatment produced a more rapid reduction of stems larger than 4 in. d.b.h. between 4 MAT and 12 MAT compared to INJ and SAW treatments. By 12 MAT, however, most trees in all three treatments showed symptoms of herbicide damage and had thinning or partially damaged crowns.

By the end of the second growing season, 16 MAT, SAW and BAS treatments produced nearly 100-percent mortality in treated stems across all d.b.h. classes. The INJ treatment still had from 1- to 8-percent survival in 4- to 8-in. d.b.h. class stems and 32-percent survival in 9 in. and larger d.b.h. stems. At 24 MAT, three out of the originally treated 19 stems, or 16 percent, of 9 in. d.b.h. and larger stems in the INJ treatment were still alive. By 36 MAT only two of those stems remained alive.

TREATMENT COSTS

The BAS treatment proved most efficient in labor time (Table 3). Mean DIT per hour was 495, which was 1.7 times more productive than the SAW treatment. The INJ treatment was not significantly less productive than the BAS treatment. Foresters typically think of production rates on a per-acre basis. This approach is not the most accurate way to reflect treatment differences in labor and cost rate since it may not account for differences in stocking and diameter distributions between the treated areas. For this study, nevertheless, SAW, INJ, and BAS treatments had labor productivity rates of 2.9, 2.2, and 1.7 hrs per acre, respectively.

The BAS and INJ treatments required similar volumes of herbicide concentrate. However, the cost per unit volume between the two different herbicides plus the cost of the oil carrier used in the BAS treatment made herbicide costs for the BAS treatment more than 4 times that of the INJ treatment and 11 times that of the SAW treatment. Mean BAS treatment cost was \$0.20 per DIT, nearly two times more costly than the SAW treatment at \$0.11 per DIT and more than twice as costly as the INJ treatment at \$0.09 per DIT. SAW and INJ treatment costs were not significantly different from each other. For these three stands, treatment costs per acre at a labor rate of \$25 per hour averaged \$169.64, \$86.46, and \$82.75 across the three stands for BAS, SAW, and INJ treatments, respectively.

Table 3.—Cost analysis for three midstory removal methods used in the implementation of the oak shelterwood method in three mature oak stands in southern Indiana. Differences in means indicated by letters within columns ($\alpha = 0.05$).

Trt	Per acre DIT ^a (in./ac)	Per acre labor time (hr/ac)	Per hour DIT (in./hr)	Per acre BAT ^b (ft ² /ac)	Per hour BAT (ft ² /hr)	Labor ^c cost per DIT (\$/in.)	Labor cost per BAT (\$/ft ²)	Labor cost per acre (\$/ac)
INJ	977 a	2.2 b	453 a	23.1 a	10.7 a	0.06 a	2.35 a	54.03 b
SAW	842 a	2.9 a	299 b	23.9 a	8.4 a	0.09 b	3.05 a	72.41 a
BAS	852 a	1.7 c	495 a	18.4 b	10.7 a	0.05 a	2.37 a	43.38 c

Trt	Herb ^d cost Per DIT (\$/in)	Herb cost per BAT (\$/ft ²)	Herb cost per acre (\$/ac)	Saw cost per DIT (\$/in)	Saw cost per BAT (\$/ft ²)	Total cost Per DIT (\$/in)	Total cost per BAT (\$/ft ²)	Total cost per acre (\$/ac)
INJ	0.03 b	1.26 b	28.71 b			0.09 b	3.61 b	82.75 b
SAW	0.01 b	0.47 c	11.24 c	0.003	0.12	0.11 b	3.64 b	86.46 b
BAS	0.10 a	4.72 a	86.01 a			0.15 a	7.09 a	129.39 a

^aDIT = diameter inches treated or the sum of diameters of all treated trees.

^bBAT = basal area (ft²) treated or the sum of basal areas of all treated trees.

^cLabor costs based on labor rate of \$25 per hour.

^dHerbicide costs based on \$35.25 per gallon of Pathway[®] concentrate (INJ and SAW), \$104.85 per gallon of Garlon4[®] plus \$14.70 per gallon of AX-IT[®] basal oil (BAS).

DISCUSSION

Given that the overall midstory species composition of the three stands included in this study was predominantly sugar maple, the BAS treatment was the most effective of the three treatments in removing the midstory canopy in the timeliest manner across the range of stem diameters encountered. By the end of the first growing season following treatment (4 MAT), the BAS treatment reduced total midstory stem numbers by 89 percent. By 12 MAT, or the beginning of the second growing season, the BAS treatment had killed 98 percent of all treated midstory stems. However, both the INJ and the SAW treatments removed a significant amount of the midstory canopy, 77 and 78 percent respectively, by the end of the first growing season. Early in the second growing season, they had removed 89 and 91 percent, respectively, of the midstory stems, greatly increasing forest floor light levels over pretreatment conditions. Tree d.b.h. within the range treated did not influence the final outcome 2 years following treatment, regardless of treatment used. In general, however, the larger the d.b.h., the longer it took for tree mortality to occur, with important differences occurring between the three treatments. Up to the end of the first growing season, mortality was strongly related to stem d.b.h. in the BAS treatment. After the first growing season, BAS treatment survival collapsed dramatically across all d.b.h. classes. Mortality rates were highly variable in the INJ and SAW treatments when stems exceeded 5 in. d.b.h. up to the beginning of the second growing season. By the end of the second growing season, however, most remaining INJ- and SAW-treated stems died irrespective of d.b.h., except for 9 in. and larger d.b.h. stems in the INJ treatment. The remaining stems in the INJ treatment continued to slowly die over the next 20 months, with a small remnant surviving to the present.

Although species did not influence the final outcome, there were important differences in how species groups responded to the three treatments. Beech and sugar maple responded more slowly to INJ and SAW treatments than did elm, hickory, MISCL, and MISCS species groups. While beech and/or sugar maple stands may respond most quickly to the BAS treatment, stands with midstory canopies composed primarily of hickory species would likely respond much more slowly to the BAS treatment. In such stands INJ and SAW treatments provide the most rapid mortality.

Kochenderfer and others (2004) demonstrated near 100-percent crown control of 6- to 11-in. and larger d.b.h. American beech in West Virginia 12 months after treatment in August using a similar injection method with a 50-percent solution of glyphosate. In the same study, a basal bark treatment using a 10-percent solution of triclopyr ester in an oil carrier provided 99-percent crown control of 1- to 6-in. d.b.h. beech. A 25-percent solution of triclopyr ester in oil applied in late winter top killed 92 percent of sugar maple stems 2 years following treatment in a study in Missouri (Thomas and Garrett 1993). In an Arkansas study, Williams and others (1996) showed decreasing efficacy of various triclopyr + imazapyr basal bark combinations applied in January with increasing d.b.h. for sweetgum, southern red oak, and hickory, approximately 9 months after treatment. For the predominantly sugar maple midstories of this study, stem diameter did not appear to be a factor in BAS treatment efficacy after the first growing season (4 MAT). Stem diameter continued to influence INJ and SAW treatment efficacy up to the end of the second growing season (16 MAT). However, most of these trees had reduced leaf area and thus were allowing more light to pass through their crowns. Hickory mortality in Williams and others' (1996) study was substantially lower (37.9 percent) for the 4-in. d.b.h. class than for the same size stems of the other two species treated. This result is comparable to the low 43-percent hickory mortality following the BAS treatment in this study at the end of the first growing season.

There is interest in reducing or removing the midstory as quickly as is financially feasible to provide to a new cohort of oak seedlings the light needed to sustain positive growth and maintain survival (Phares 1971). A new large cohort of oak seedlings is often not detected in a stand until after seedling emergence. New northern red oak seedlings establishing in undisturbed forest understories rely on the reserves stored in the seed cotyledons to sustain them through the first growing season (Crow 1988). Low light level stress-induced mortality in northern red oak therefore may be limited in the first growing season (Crow 1992). First-year oak seedling mortality may be more related to other factors, such as predation (Rathfon and others 2008). However, low light levels also reduce carbon allocation to the roots, resulting in oak seedlings with lower root:shoot ratios than needed for competitive advance oak reproduction (Kolb and others 1990). After cotyledon reserves are exhausted, seedling mortality in undisturbed stands with well developed midstory canopies commences. Crow (1992) found that after 6 years, 36 percent of a cohort of northern red oak seedlings survived under an intact overstory canopy (undisturbed), compared to 54 and 92 percent survival for partial and no overstory or open conditions where understory vegetation was initially controlled by a broadcast herbicide application. The 2006 cohort of oak seedlings in the three subject stands of this study had 48- and 80-percent greater survival at the end of the first growing season in the INJ and SAW treatments, respectively, compared with seedlings growing in the CON treatment (Rathfon and others 2008).

Photosynthetically active radiation (PAR) data were measured on the forest floor prior to implementation of the midstory control treatments and for 2 years after. These data are not presented. However, understory light levels increased substantially in all three treatments, but not until the second growing season. During the first growing season, dead foliage did not immediately drop off, thereby still providing shade. The felling of trees in the SAW treatment provided an immediate small increase in ambient understory light levels. When performing intermediate stand treatments, many foresters choose to fell, rather than girdle, larger stems than were felled in this study. Thus, a more substantial immediate increase in light could be attained. However, complete midstory control as prescribed in the oak shelterwood method would require the felling of many more stems than would normally be done in most standard timber stand improvement operations.

Yet another factor for forest managers to consider when choosing midstory control options is potential herbicide damage to oak reproduction. First-year oak seedling survival was substantially reduced in the BAS treatment due to volatilization of triclopyr ester that was absorbed as a vapor by the foliage of the young seedlings (Rathfon 2008). At least 59 percent of seedlings tallied as dead were top killed by triclopyr ester exposure. Nontarget hardwood seedling damage in stands treated with the basal bark treatment is not well documented in the literature. Ester formulations of triclopyr, however, volatilize when air temperatures exceed 85 to 90 °F (Tu and others 2001). Midsummer applications of the BAS treatment may be ill-advised in midstory control applications where a new cohort of oak seedlings has recently emerged or established.

Cain (1995) reported labor productivity rates for cleaning hardwoods in shortleaf pine stands in Arkansas. It required approximately 2 to 3 hrs per acre for chainsaw felling and 3 to 4 hrs per acre for injection using a tube type injector to clear 300 to 400 hardwood stems per acre. In hardwood stands in West Virginia where primarily beech stems 1 to 6 in. d.b.h. were treated, a basal bark treatment required approximately 2 hrs per acre (Kochenderfer and others 2004). Midstory stems 6 in. d.b.h. and larger treated in the same study using an injection treatment similar to that used in this study required 2.7 hrs per acre.

Given the substantially greater costs of applying the BAS treatment for an already costly silvicultural treatment, land managers faced with limited budgets will likely choose lower cost alternatives. The SAW treatment remains financially attractive at labor rates below \$25 per hr. At higher labor costs, the INJ treatment becomes the lower-cost alternative. If lower herbicide concentrations were used in a BAS treatment that would still prove effective over a broad range of species and stem diameters, BAS treatment could become more cost-effective. However, reducing the triclopyr concentration by half to 10 percent would still make the BAS treatment \$0.07 per DIT or \$51.90 per acre, more costly than the INJ treatment, where labor rates are \$25 per hr. Some employers are looking for safer ways to implement silvicultural treatments to reduce insurance costs. This objective often means looking for alternatives to the use of chainsaws. Although the INJ treatment left a small number of residual surviving stems, both the lower cost and the potential to reduce chainsaw use, along with its overall effectiveness, make this an attractive choice.

CONCLUSIONS

INJ, SAW, and BAS treatments all were effective in removing nearly all or the majority of the midstory canopy by the end of the first or the beginning of the second growing season of a cohort of oak seedlings. Although a substantial number of stems greater than 5 in. d.b.h. continued to survive during the second growing season in INJ and SAW treatments, most of these had damaged crowns that admitted much more light than prior to treatment. BAS treatment costs are considerably higher than INJ and SAW treatments costs when applied to midstory control work where 300 or more stems per acre will be treated.

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

A COST COMPARISON OF FIVE MIDSTORY REMOVAL METHODS

Brian G. Bailey, Michael R. Saunders, and Zachary E. Lowe¹

Abstract.—Within mature hardwood forests, midstory removal treatments have been shown to provide the adequate light and growing space needed for early establishment of intermediate-shade-tolerant species. As the method gains popularity, it is worthwhile to determine what manner of removal is most cost-efficient. This study compared five midstory removal treatments across 10 acres on two sites in Indiana. The treatments were: manual (chainsaws and brush saws), mechanical (tree mower), herbicide (15-percent triclopyr and 3-percent imazapyr), mechanical plus herbicide, and manual plus herbicide. Financial inputs, consisting of labor hours, equipment maintenance, and material costs, were closely documented. The removal increased light to a mean of 15 percent of an open canopy, and basal area was reduced by 20 ft² ac⁻¹. Large equipment was shown to be very costly relative to its efficacy, and could not be recommended unless an individual either has access to less expensive equipment or is working with unusually large and easily accessible stands. In terms of basal area and stems removed, chainsaws and brush saws were more economical and more efficient than the other four treatments.

INTRODUCTION

Tree species intermediately tolerant of shade (i.e., intermediates) and unable to persist under intact and mature hardwood forests are often at a competitive disadvantage when canopy gaps are created. However, preemptive silviculture management can help establish advance regeneration of intermediate species, positioning them to recruit into the next stand after overstory removal (Loftis and McGee 1993, Gould and others 2005). Because advanced regeneration is so critical for oak (*Quercus* spp.) recruitment (Lorimer 1993), much work has been done to develop techniques to encourage that regeneration (Teclaw and Isebrands 1993, Paquette and others 2006). The two-step shelterwood, beginning with a preparatory removal of the shade-tolerant midstories common to many hardwood-dominated stands, has been shown to produce the appropriate light and space conditions for establishment of intermediates (Lhotka and Loewenstein 2009). While intermediates cannot persevere under a full canopy, too much light allows aggressive and competing shade-intolerant species to flourish as well. The slight increase in light intensity from a midstory removal can strike a balance between the extremes, a condition necessary for success (Lockhart and others 2000). Only after the intermediate trees gain a competitive advantage maintainable over both shade-tolerant and aggressive shade-intolerant species, which can take up to 10 years (Loftis 1990), can the final shelterwood overstory removal be conducted.

The primary objective of this study was to determine the most effective method of implementing midstory removal, in terms of financial costs and treatment efficiency. Five treatments were included for comparison: manual (chainsaws and brush saws), mechanical (tree mower), basal bark herbicide (15-percent triclopyr and

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3-percent imazapyr), mechanical plus cut stump herbicide, and manual plus cut stump herbicide. Much of the available literature on thinning does not refer specifically to midstory removals, but to different canopy, basal area, and diameter removals (McCauley and Marquis 1972). Accurate prediction costs are essential to planning, however, so determining the expense of these strategies is important (Olson and others 1978). The tested methods encompass the common range of options accessible to practitioners. By comparing several treatments and separating costs by input, practitioners may interpret these findings with regard to their own objectives, sites, and resources. Trends of cost per basal area or numbers of stems are examined to help extrapolate these results to a broader range of stands, and help illustrate the resiliency of methods between sites.

STUDY AREAS

This study was replicated across two relatively small, albeit quite different, sites. The Cox-Haggerty property is located approximately 2 miles west of West Lafayette, IN, and Purdue University's campus (latitude: 40° 25.7'N; longitude: 86° 58.2'W), and is approximately 6 ac in size. Cox-Haggerty had a pretreatment stem density of 942 ac⁻¹, basal area of 156 ft² ac⁻¹, and a quadratic mean diameter of 8.3 inches. Soil is Miami silt loam, a moderately well-drained soil. By stem count the overstory (diameter at breast height [d.b.h.] > 6.0 in.) composition was dominated by oak (46 percent), of which 33 percent was white oak (*Q. alba* L.), and 8 percent was northern red oak (*Q. rubra* L.). Hickory (*Carya* spp.) accounts for 20 percent of the overstory, and sugar maple (*Acer saccharum* Marsh.) 5 percent. The midstory (d.b.h. >0.5 in. or ≤6.0 in.) was dominated by sugar maple (36 percent), and Amur honeysuckle (*Lonicera maackii* [Rupr.] Herder) (27 percent); Ohio buckeye (*Aesculus glabra* Willd.) accounted for 8 percent.

The second study area is located at Purdue's Samuel G. Meigs Farm, hereinafter referred to as "Meigs Farm," approximately 8 miles south of West Lafayette, IN (latitude: 40° 17.3'N; longitude: 86° 52.5'W). Meigs Farm is approximately 10 ac and had a pretreatment stem density of 494 ac⁻¹, a basal area of 113 ft² ac⁻¹, and a quadratic mean diameter of 10 inches. Soil class is predominately Crosby-Miami and Richardville silt loam, with pockets of the poorly drained Mahalasville-Treaty complex. By stem count the overstory composition was dominated by hickory (54 percent), elm (*Ulmus* spp.) (24 percent), and black cherry (*Prunus serotina* Ehrh.) (10 percent). The midstory comprises elm (65 percent), and hackberry (*Celtis occidentalis* L.) (12 percent).

METHODS

In summer 2008, six approximately 197-ft x 197-ft treatment plots were laid out at Meigs Farm, and six approximately 131-ft x 197-ft treatment plots at Cox-Haggerty. Within each plot, buffered by at least 33 ft from any side to alleviate edge effects, a single measurement plot was centered. Within the measurement plot, a complete inventory of overstory trees was taken. At each corner of the measurement plot, 33-ft x 33-ft subplots were installed to sample the midstory.

In March and April 2009, we implemented five different midstory removal techniques, randomly applied to one of the six treatment plots at each site, leaving the remainder as an untreated control. Unfortunately, we deemed within-site replication unfeasible, largely due to the small acreage of each study area and the importance of keeping fairly large treatment plot sizes for long-term monitoring of tree regeneration response. Furthermore, we felt that plots smaller than those used may not easily accept the larger mechanical equipment. Therefore, since there were only two replications of each treatment, no mean separation statistics are presented.

For manual treatments we used a Stihl MS 260 chainsaw, a Husqvarna 575XP chainsaw, and two Stihl X-series 345F brush saws. Brush saws were used on smaller-diameter stems (generally < 4.0 in. d.b.h.), while chainsaws were used on the rest. The mechanical treatments were conducted using a New Holland model 7610 4wd, 90 hp tractor equipped with a Brown 2620 Folding Deck Tree Cutter attachment (Brown Mfg. Corp., Ozark, AL), followed by cleanup with a chainsaw to clear trees that were missed by or inaccessible to the tree mower. For the herbicide-only treatment, a backpack sprayer was used to spray a basal bark application using a mixture of 15-percent triclopyr (Garlon 4 Ultra®; DowAgroSciences), 3-percent imazapyr (Stalker®; Cyanamid), and 82-percent Ax-It® (Townsend Chemical) bark oil. For both the manual and mechanical plus herbicide treatments, the same herbicide mixture was applied to cut stumps using smaller 6.8 fl. oz. hand sprayers.

Labor was split between either one- or two-person crews. Manual treatments had either one individual running the brush saw while a second person operated a chainsaw, or one person first using the brush saw before switching to a chainsaw. For the manual plus herbicide treatments, the saw operators were able to carry the hand sprayers, and spray each stump immediately after cutting. Mechanical treatments were installed with one tractor operator, plus another individual coming in after the tractor to clear missed or inaccessible stems with a chainsaw. The individual would also spray the stumps cut by the Tree Cutter when prescribed. Herbicide treatments were implemented with one or two individuals carrying backpack sprayers, working independently throughout the plot.

Energy inputs, including labor, petroleum, herbicide, and type and time of equipment used, were recorded for each treatment. To evaluate costs we assumed set rates for labor (\$20 hr⁻¹), herbicide (\$38.04 gal⁻¹), and fuel (\$2.80 gal⁻¹). Equipment costs were based upon rental prices. While some equipment is commonly owned by many practitioners, the larger tree mower is not. Equipment costs were estimated using rental rates to avoid the complexities of life span and maintenance cost assumptions. Equipment rental rates were: brush saw, \$30 day⁻¹; chainsaw, \$8 day⁻¹; herbicide sprayer, \$15 day⁻¹; and tree mower, \$675 day⁻¹.

We used hemispheric photography to measure the effectiveness of midstory removal treatments to increase light availability. In August 2009, eight photos were taken within each treatment plot using a Canon EOS (SLR) 20D digital camera with a Sigma F3.5 EX DG 8mm circular fisheye lens, mounted on a tripod 3.3 ft above the ground. Images were analyzed with Gap Light Analyzer software version 2.0 (Millbrook, NY) to estimate canopy openness by treatment. Canopy openness is defined as percentage of unobscured sky (Raaimakers and others 1995). A Tukey's HSD test was used to compare canopy openness between removal treatments and the control ($\alpha = 0.05$).

RESULTS

LIGHT AND STAND STRUCTURE

All removal treatments showed significantly higher canopy openness levels than the control ($p < 0.05$). Mean canopy openness increased 111 percent between the midstory removal treatments and the control plots (Fig. 1). Understory mean light transmission within the controls was 7 percent of open sky on both sites, and between 14 and 17 percent within the midstory removal treatments. Basal area was reduced by a mean of 20 ft² ac⁻¹ or 12 percent on both sites, and stem densities were reduced by a mean of 62 percent at Meigs Farm and by a mean of 70 percent at Cox-Haggerty.

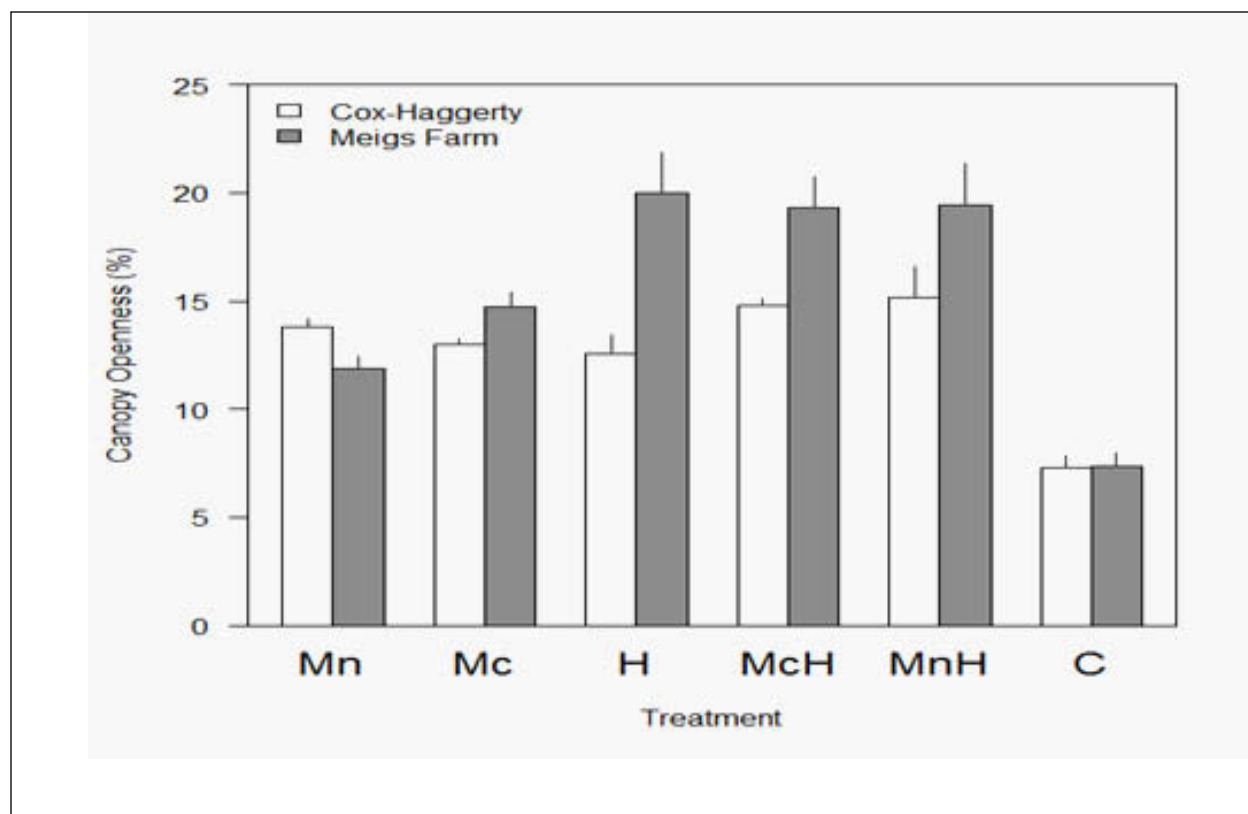


Figure 1.—Percent open canopy by site and treatment with standard error bars shown. Treatments: manual (Mn), mechanical (Mc), herbicide (H), mechanical plus herbicide (McH), manual plus herbicide (MnH), and control (C).

COSTS OF MIDSTORY REMOVALS

Treatment costs ranged from \$36 ac^{-1} to \$406 ac^{-1} (Fig. 2). Between the two extremes, equipment cost of the mechanical treatment alone accounted for 80 percent of the difference (Table 1). Variation in costs for a treatment was driven by stand structure and topographical differences between sites. The largest difference within a single treatment between sites was \$104 ac^{-1} , between the two manual plus herbicide treatments. The Meigs stand had approximately 544 stems ac^{-1} and a slope of zero, while the Cox-Haggerty stand had approximately 958 stems ac^{-1} and steep topographic relief, both factors that regularly impact intermediate stand treatments costs (Wickstrom and Alley 1967). Where herbicide was applied, it cost between \$120 ac^{-1} and \$139 ac^{-1} (Table 1). The exception was in the mechanical treatments, where the equipment shattered most stems, reducing the probability of sprouting enough for us to skip spraying those stems (Ducrey and Turrel 1992).

Cost per square foot of basal area removed ranged from \$2.39 ac^{-1} to \$19.92 ac^{-1} , with the exception of the herbicide treatment at Cox-Haggerty, which cost \$55.50 ac^{-1} (Fig. 3a). In this one case, however, there was an unusually low amount of basal area within that plot to remove, so the estimate appears inflated by the number of stems, and not the work done within. On both sites the manual treatment was clearly the least expensive method per basal area removed. While total cost per acre suggested manual plus herbicide was one of the most expensive treatments (Fig. 2a), on the basis of per unit basal area removed, it was one of the most efficient (Fig. 3a).

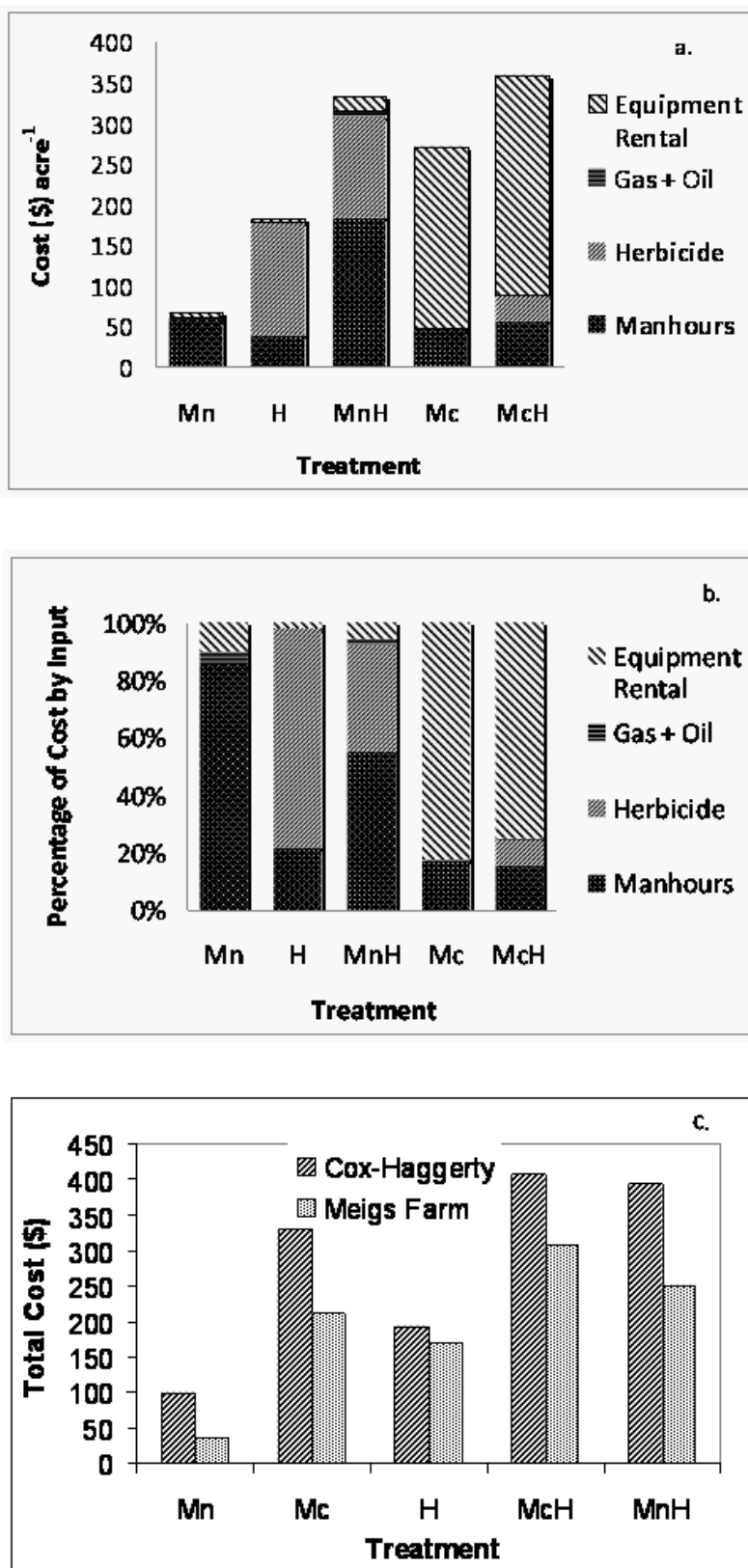


Figure 2.—Mean cost of treatment over both sites, per input (a), percent of mean cost of treatment by input (b,) and total cost of treatment by site (c). Treatments: manual (Mn), mechanical (Mc), herbicide (H), mechanical plus herbicide (McH), manual plus herbicide (MnH).

Table 1.—Total cost (\$) and costs (\$) per input by treatment. Range shows minimum and maximum observations at Cox-Haggerty and Meigs Farm.

Treatment	Labor	Herbicide	Fuel/Oil	Equipment	Total
Manual	30 - 85	-	2 - 4	4 - 10	36 - 99
Mechanical	36 - 56	-	1 - 2	173 - 272	210 - 330
Herbicide	28 - 50	138 - 139	-	4	171 - 193
Manual and Herbicide	120 - 246	120 - 126	4 - 10	6 - 12	251 - 395
Mechanical and Herbicide	50 - 62	17 - 45	1 - 2	240 - 298	309 - 406

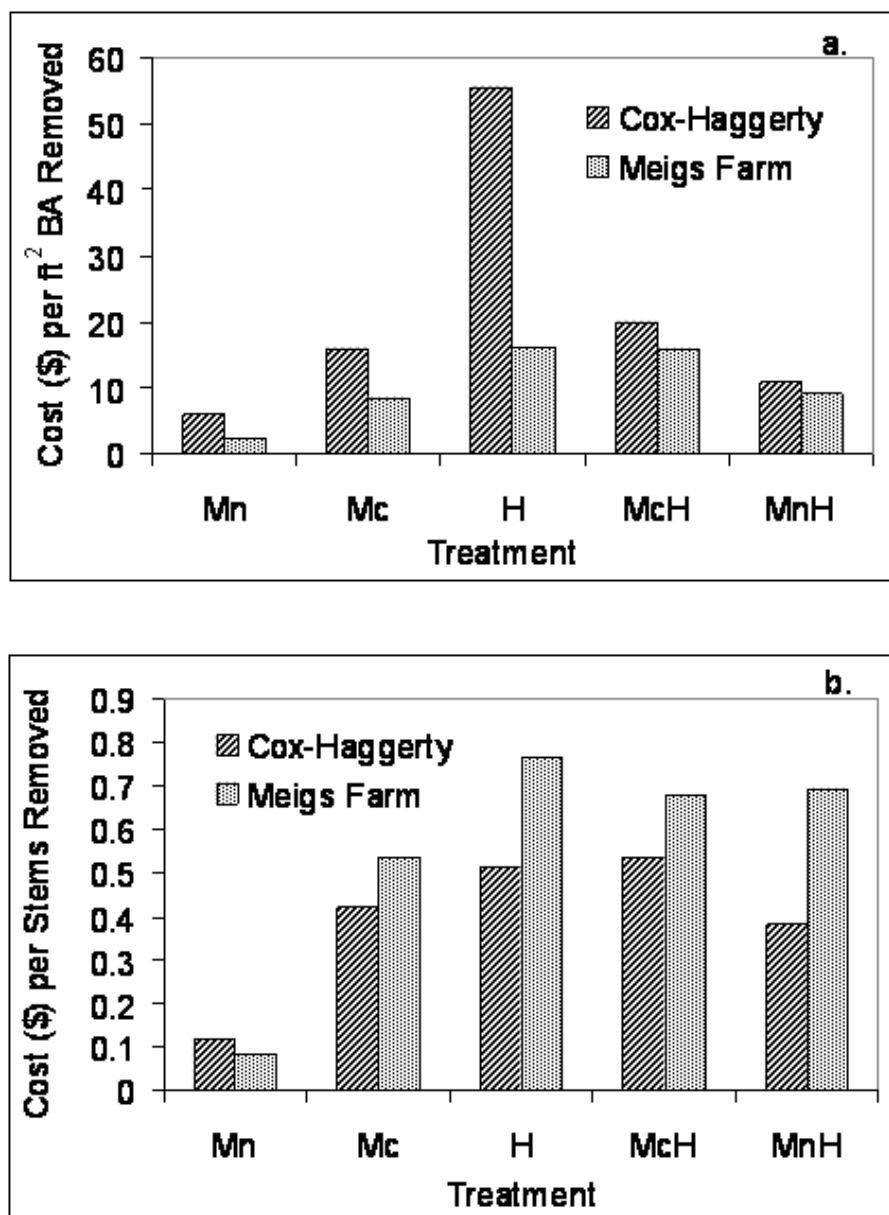


Figure 3.—Cost per square feet of basal area removed (a), and cost per stem removed (b). Treatments: manual (Mn), mechanical (Mc), herbicide (H), mechanical plus herbicide (McH), manual plus herbicide (MnH).

Cost per stem removed also presents the manual treatment as the most cost-efficient (Fig. 3b). The mechanical treatments are very costly on a per-acre basis (between \$210 and \$406 ac^{-1}), but they were not consistently more expensive than the others on the basis of per square foot of basal area removed or per stem removed.

DISCUSSION

The canopy openness within the uncut controls was similar to that found in other mature hardwood stands (<8 percent) (Jenkins and Chambers 1989). Likewise, the midstory removal increased light by approximately the same proportion as other midstory removal treatments previously documented (Lhotka and Loewenstein 2009). Our post-thinning light levels are noteworthy because they correspond well with results from shade cloth experiments, which have repeatedly shown that similar light levels, those nearing 20 percent, cause increases in growth of intermediate-shade-tolerant seedlings (Gottschalk 1994). While midstory removal does not optimize light for optimal seedling growth (Walters and others 1993), the main strategy of midstory removal is to provide a relative growth increase of intermediates compared with the competitive shade-tolerant and shade-intolerant species (Beaudet and others 2004).

Though the mechanical treatments were some of the most expensive, they also showed a degree of resilience to increased stem densities. Cost per stem removed increased 21 percent when mechanical treatments were implemented on a site with high stem densities, but there was a 45-percent increase in cost per stem removed when the manual plus herbicide treatment was implemented on the similar sites. Two reasons help explain this difference, and both are contributing factors worthy of consideration when adopting a prescription for an individual site. The herbicide and herbicide plus manual treatments were most sensitive to stem density, simply because more stems needed to be sprayed. With exceptionally high stem densities to treat (up to 1,000+ ac^{-1}), more herbicide and application time are needed. Second, high stem densities make it difficult to maneuver through a thick understory with a backpack sprayer, which will inevitably slow the technician's progress. Though both reasons add time to workers on foot, it makes little difference in the speed of a mechanical treatment. Even with extremely high stem densities, the cost of the manual treatments would not approach that of the mechanical, but the mechanical could be more cost-effective than the herbicide and manual plus herbicide treatments, particularly on large tracts with flat to moderately sloping terrain.

Each methods had its limitations. The accessibility of the mechanical equipment was limited to slopes less than 15-20 percent, and downed logs impeded the tractor's movement. Chainsaws also had to be used after the mechanical treatment to clean up missed or inaccessible stems. For the herbicide treatments, it was difficult to achieve a complete kill in one pass, particularly during leaf-off conditions. Follow-up treatments are likely necessary. Further, while the manual treatments were less expensive, the efficacy of the treatment has not yet been fully measured. Choosing not to use herbicide will likely result in an increase in the amount of sprouting, particularly for invasive species, in turn potentially jeopardizing the success of the treatment. Sprouts may grow much faster than naturally regenerated or planted desirables, once again becoming competition in a few years (Ducrey and Turrel 1992). When a method is being chosen, assessment is therefore critical in matching the appropriate tools and resources to the site and stand.

As the first step of a shelterwood, midstory removal is often an essential part of the future success of the system. It is therefore important to find a suitable method that fits within the constraints of a manager's resources, yet is still compatible with the site and objectives. By using these data to help inform that decision, managers can achieve better planning and results.

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SOCIOECONOMICS

ROUNDWOOD MARKETS AND UTILIZATION IN WEST VIRGINIA AND OHIO

Shawn T. Grushecky, Jan Wiedenbeck, and Ben Spong¹

Abstract.—West Virginia and Ohio have similar forest resources and extensive forest-based economies. Roundwood is harvested throughout this central Appalachian region and supports a diverse primary and secondary forest products sector. The objective of this research was to investigate the utilization of the forest resource harvested in West Virginia and Ohio. Utilization and market data were collected on 30 active timber harvests in 2008 in West Virginia and 9 active harvests in 2009 in Ohio. Results indicated that the average number of roundwood markets to which loggers transported materials from these 39 harvest sites was 2.4. The main markets supplied were saw logs and softwood pulp in West Virginia and saw logs in Ohio. The overall average utilization achieved at these 39 sites was 82 percent. The utilization rate averaged 80 percent for logging operations with only one or two markets and 85 percent for operations hauling products to three or more markets. The highest utilization rates were measured for pine and yellow-poplar while the lowest rates were for cherry and lower-density hardwood species. Differences exist in the characteristics of roundwood markets utilized by logging operators in Ohio and West Virginia. These results demonstrate the impact that the number and nature of markets have on log utilization at harvest sites.

INTRODUCTION

Two utilization factors are used in translating Timber Products Output (TPO) data collected in surveys of primary wood processors—a logging utilization factor, which is the subject of this study, and a milling utilization factor. The logging utilization factor provides information on the volume of residual material left in the woods compared to the volume of roundwood produced for different primary processing markets. The milling utilization factor provides information on the volume of bark, coarse residues, and fine residues produced in primary processing operations for different primary processing sectors. Throughout the region encompassed by the U.S. Forest Service Northern Research Station, these utilization factors are outdated.

For the 13 northeastern states from Maine to West Virginia and Ohio, a single set of factors has been in use, despite wide variation in forests, species, and markets. The hardwood and softwood utilization factors currently used for the region are 29.5 percent and 26.3 percent, respectively (Smith and others 2001). In many southern states, the U.S. Forest Service Southern Research Station has been updating utilization factors on a routine basis. Utilization rates have been recently completed in Virginia, Florida, and North Carolina (among others) in the last decade (Bentley and Johnson 2008, 2009a, 2009b).

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There is good reason to believe that the factors developed in past decades are no longer applicable. Logging utilization has been impacted in many regions by the development and expansion (and subsequent contraction) of the oriented strandboard (OSB) industry. Recent closings of paper mills also have impacted markets for roundwood. Rapidly increasing energy costs create new markets for top and stemwood in some regions. The economic feasibility of hauling logs to various markets is strongly influenced by prices paid for roundwood in these markets. Roundwood prices have risen in most of these markets over the last two decades but have fallen off since 2007.

The logging utilization factor is applied to the roundwood consumption volumes provided by TPO survey respondents to derive estimates of forestland removals (Blyth and Smith 1979). This factor finds widespread use by forest planners and analysts assessing regional timber supply and demand scenarios. Location analyses for new milling operations rely on these factors. The current emphasis on the developing wood-based energy sector has led to heavy use of U.S. Forest Service Forest Inventory and Analysis (FIA) data and maps as they provide insights on raw material availability for this sector. In particular, the map of *“Unused wood and bark residues from timber harvesting and primary processing mills in the conterminous United States”* (Perry and others 2009) is useful in bio-energy project planning.

Logging utilization information, when combined with logging residue information obtained through physical surveys of sites where harvests have recently been conducted, gives a complete picture of a region's woody residue volumes and characteristics. Grushecky and others conducted physical surveys of harvest site residues in West Virginia in 1995 and 2002 (Grushecky and others 2004). The 2002 survey found that the average weight of wood residues left at harvest sites in southern West Virginia was 10.4 tons per acre (Grushecky and others 2004). The quantity of residue found at the 70 sites sampled in this study was highly variable and no association was found between residue volumes left in the woods and the average slope of the harvested sites. This result indicates that economic, market, and operational factors must be understood when evaluating logging efficiencies and utilization rates. Alderman and Luppold (2005) evaluated logging company characteristics and market factors for 30 West Virginia harvesting operations. Their analysis, however, was not tied to logging site-based material utilization measurements. A combined analysis that includes an assessment of the physical resource and market influences is needed to enhance our understanding of the potential for further utilization of the roundwood resource in the central Appalachian region.

OBJECTIVES

The objectives of this research were to:

1. Derive a current logging utilization factor for the central Appalachian region.
2. Identify which environmental, operational, and market factors have the strongest influence on the utilization rate achieved by logging operations.

METHODOLOGY

SITE SELECTION AND DATA COLLECTION

It was estimated that available resources would allow for sampling at 40 harvest sites during 2008 and 2009. Thirty sites were selected in West Virginia by identifying active harvesting. Harvest sites were located using logging notification forms filed with the West Virginia Division of Forestry (WVDOP) (WVDOP 2010) as

mandated under the 1992 Logging Sediment Control Act (WVDOF 2010). Under this act, all loggers are required to submit a harvest plan before timber operations commence. An equal number of harvest sites were selected randomly from 2008 WVDOF notifications from each of the three WVDOF administrative regions. Logging companies were then contacted to seek their participation. The distribution of sample sites in West Virginia reflects the distribution of active logging jobs during 2008.

In Ohio, the process of identifying active harvesting operations was not guided by a registry, since registration is not required. Nonetheless, state personnel within Ohio's Department of Natural Resources—Division of Forestry maintain awareness of logging activity and were able to provide contact information for logging contractors. Various primary processors in Ohio also were contacted and asked to provide a list of loggers operating in their region. Sampling sites in Ohio were concentrated in the eastern and southeastern regions as these regions share similar geography and markets with West Virginia. These common features allow the data from these sites to be combined with the West Virginia data to produce regional logging utilization insights and factors. Market and weather constraints in 2009 caused our site sampling effort to come up one site short of our goal of 10 sites. Figure 1 shows the locations of the 39 logging utilization study sites.

Sampling methodology was patterned after the system developed by members of the FIA group of the U.S. Forest Service's Southern Research Station. Project personnel underwent a day of field training with the Southern Research Station to review data collection procedures.

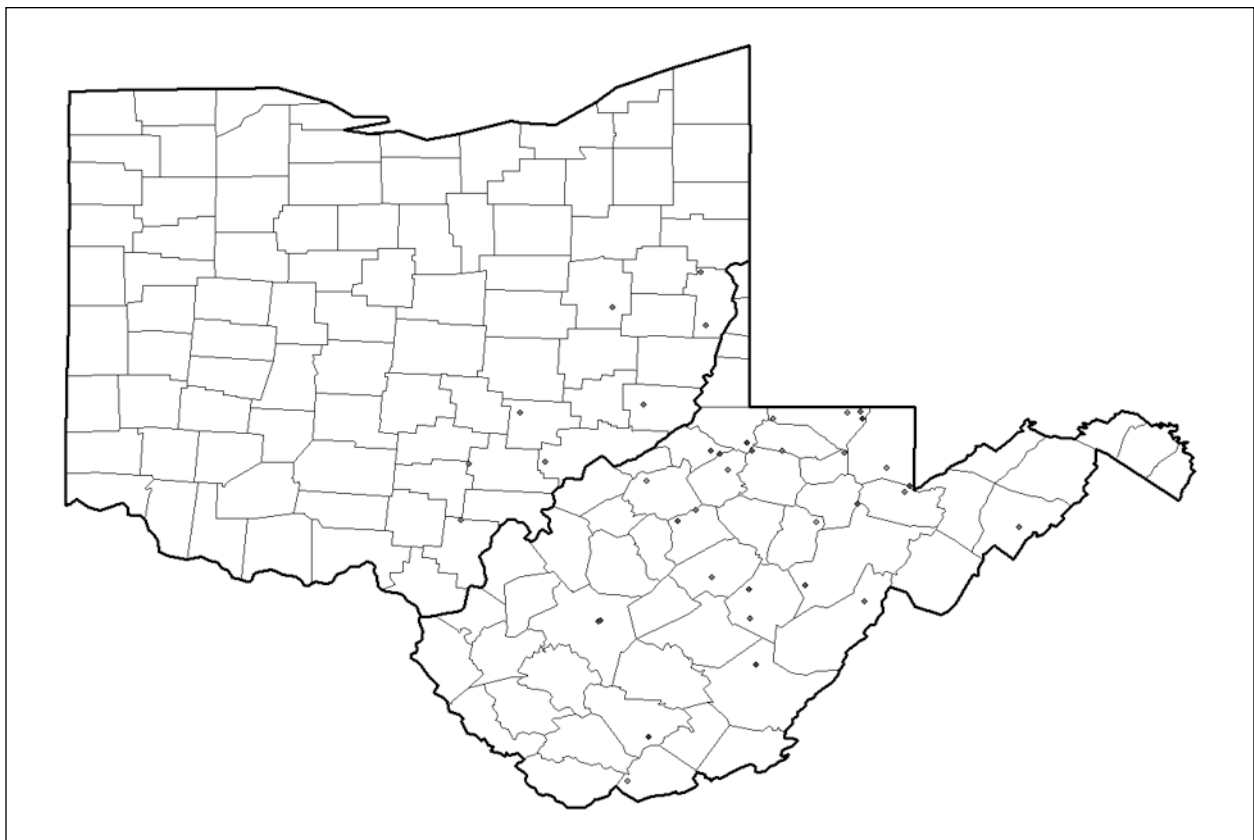


Figure 1.—Locations of 39 sites in West Virginia and Ohio where logging utilization studies were conducted in 2008 and 2009.

According to the standard sampling methodology:

Each [felled] tree was measured from the top of the cut stump to the end of utilization. Measurements were made along the main stem in sections no longer than 16 feet until the end of utilization. The end of utilization usually is determined by the sawyer, according to particular specifications set by the receiving mill(s). FIA merchantability standards for growing-stock volume are defined as the volume in the main stem of the tree from a 1-foot stump to a 4-inch top. However, most trees are not cut exactly at a 1-foot stump, nor are they cut off at exactly 4 inches. For example, trees cut off above a 1-foot stump and below 4 inches would be considered underutilized, and that volume not utilized would be considered growing-stock residue [The other required measurement, besides d.b.h. [diameter at breast height] and end of utilization, is the top of the sawtimber portion (9.0 inches in hardwoods). This measurement allows calculation of the sawtimber and poletimber portion of the growing-stock section (Bentley and Johnson 2009).

The goal at each site was to sample 30 felled trees for utilization—both in the woods and at the landing. Sample trees were randomly selected from within the active harvest zone. Details about the logging operation were obtained from the logging contractor including: number of employees; types of equipment; names, locations, and material specifications associated with the markets to which the harvested material was being sent; loads shipped per day; harvest tract size; type of harvest being executed; and other relevant information provided by the logger. The study crew consisted of three people—two taking utilization measurements in the woods on the felled trees and one measuring utilization at the landing where log bucking and sorting to product specifications was executed.

DATA ANALYSIS

A large amount of data was collected in this study so significant attention was given to checking data and resolving data discrepancies before analysis could proceed. The utilization rates for each logging job (also referred to as “sites” throughout this paper) were based on the overall volume of utilized wood divided by the total wood volume. The total volume included both the growing stock portions and the nongrowing stock portions of sample trees. While nongrowing stock portions in the tree tops above the 4-inch diameter cutoff point were not measured, nongrowing stock portions such as cull sections along the tree’s main stem and limbs with diameters greater than 4 inches were measured.

Summary statistics for several variables were calculated: in-woods utilization, at-landing utilization, overall utilization, average stump height and diameter, number of markets, and haul distances to markets. These measurement variables were summarized for the entire sample and for the West Virginia and Ohio subsamples. Summaries based on species and species-groups also were conducted. The overall utilization rate is obtained by multiplying the utilization rate for in-woods operations (felling, topping, bucking to the merchantable length) by the utilization rate for operations at the log landing (bucking to specified product lengths, bucking out unacceptable defects/sections).

Analysis of variance (ANOVA) was used to investigate whether utilization rates varied based on: 1) whether the operation was in West Virginia or Ohio; and 2) the number of markets supplied by each logging operation ($\alpha=0.10$). Two-sample t-tests ($\alpha = 0.10$) were used to determine whether the utilization rates and stump heights measured in West Virginia were statistically similar to those measured in Ohio. Correlation analysis was conducted to test for association between average stump height and average stump diameter ($\alpha = 0.10$) among species groups.

RESULTS AND DISCUSSION

SPECIES GROUPS

At the 39 harvest operations visited, we measured 1,087 felled trees. This sample consisted of 815 felled trees measured at 30 sites in West Virginia (average of 27 trees per site) and 272 trees measured at 9 sites in Ohio (about 30 trees per site). In West Virginia, the three most prominent species being harvested during our site visits were yellow-poplar (*Liriodendron tulipifera* L.), white oak (*Quercus alba* L.), and soft maple (*Acer rubrum* L.). In Ohio, red oak (*Quercus rubra* L.), white oak, and black cherry (*Prunus serotina* Ehrh.) were the most frequently encountered tree species (Fig. 2). Species groupings used in analyses were “mixed hardwoods,” “soft hardwoods,” “pine,” and “hickory”. Two of these categories contain multiple species. “Soft hardwoods” includes several lower-density hardwood species such as aspen (*Populus* spp.), basswood (*Tilia americana* L.), blackgum (*Nyssa sylvatica* Marsh.), and sourwood (*Oxydendrum arboreum* DC). “Mixed hardwoods” includes the ashes (*Fraxinus* spp.), elms (*Ulmus* spp.), and walnuts (*Juglans* spp.).

UTILIZATION RATES

The key result of this research is the overall average utilization rate for harvesting operations in the central Appalachian region (addressing Objective 1). The average in-woods utilization rate was 83.8 percent while the average utilization rate at the log landing was 98.3 percent (Table 1). When multiplied, these two utilization rates indicate an overall utilization rate of 82.3 percent. This value means that 82 percent of the growing stock volume of harvested trees was utilized for products and 18 percent was left as residue. When the pine data are separated, the overall utilization rates for softwoods (pine) and hardwoods were 95 and 76 percent, respectively (Table 1). These rates align well with the rates determined by Bentley and Johnson (2009b) for Virginia, which were 95 and 79 percent, respectively.

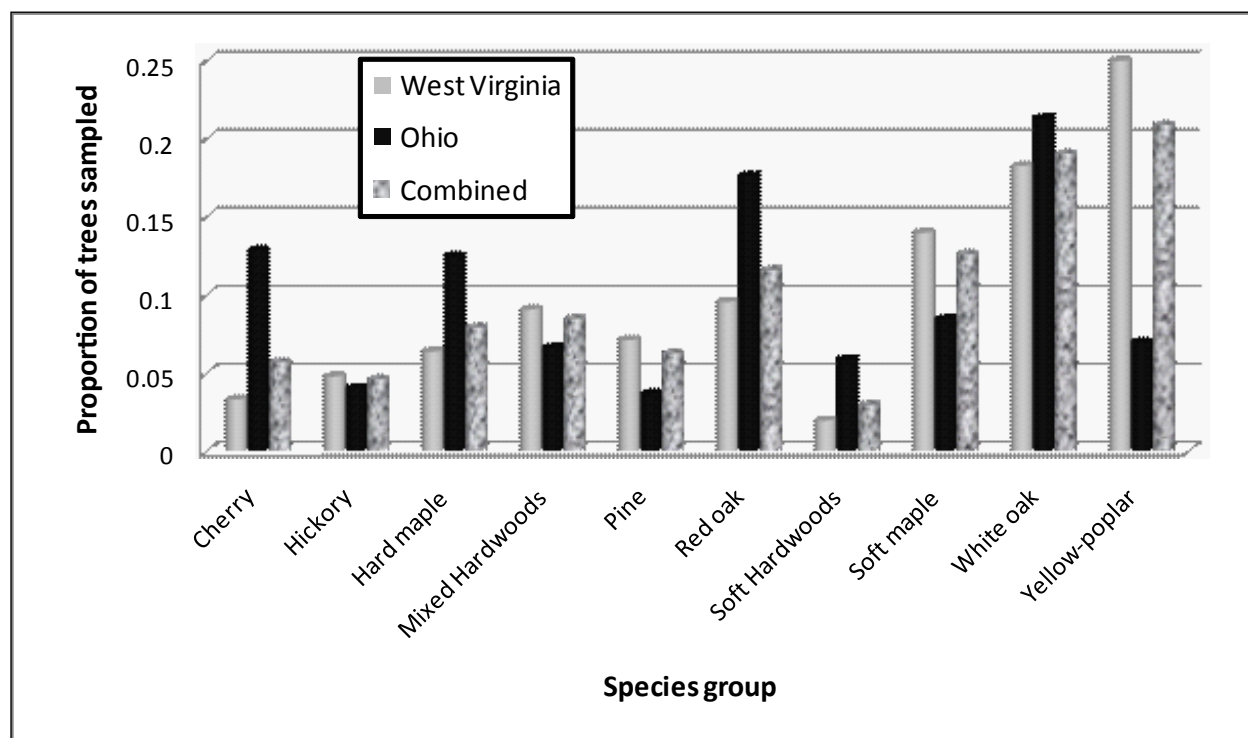


Figure 2.—Distribution of tree species examined in logging utilization studies conducted in West Virginia and Ohio in 2008 and 2009.

Table 1.—Logging utilization rates for the central Appalachian region and for state-based subgroups.

Utilization Measurement	West Virginia (n=30)		Ohio (n=9)		All sites (n=39)	
	Utiliz. value	Standard dev.	Utiliz. value (percent)	Standard dev.	Utiliz. value	Standard dev.
In-woods utilization	88.6	6.6	67.7	21.4	83.8	14.4
At-landing utilization	97.8	1.9	99.9	0.2	98.3	1.9
Overall utilization	86.7		67.6		82.3	
Hardwood utilization	84.9		66.6		75.8	
Softwood utilization	94.7		96.1		95.4	

To understand how various factors influence utilization requires the further analysis of the rates that were measured at the 39 harvest sites. Of particular note is the fact that both the average overall utilization rate and the average in-woods utilization rate were lower for the nine Ohio logging jobs than for the 30 West Virginia jobs. The overall rates averaged 67.6 percent in Ohio and 86.7 percent in West Virginia while the in-woods rates averaged 67.7 percent in Ohio and 88.6 percent in West Virginia. Analysis of variance for these utilization rates indicated both were significantly different between the two state-based groups. The p-values for the tests were .0278 and .0189 for the overall and the in-woods utilization, respectively. The Welch's ANOVA model was used after the Bartlett's test indicated the variance among rates within each group displayed heteroscedacity.

The at-landing rates for the two groups were 99.9 percent in Ohio and 97.8 percent in West Virginia. These rates also were significantly different (Welch's $p = .0001$), but the influence of this difference on total utilization is minor compared to the in-woods rate difference. The absolute difference is small (2 percent) compared to the absolute difference for the in-woods utilization rates (21 percent).

DIFFERENCES IN UTILIZATION RATES BASED ON NUMBER OF ROUNDWOOD MARKETS

Loggers in West Virginia identified 10 types of markets to which they shipped their harvested volume: veneer, saw, peeler, and scragg mill logs; hardwood and softwood pulp; fence posts and rails; specialty products; and firewood. The nine loggers visited in Ohio identified only five markets: saw and scragg logs, hardwood pulp, wood chips, and firewood. All 39 logging operations had sawmill markets. Additionally, in West Virginia, 60 percent had softwood pulp markets, 57 percent had peeler markets, and 53 percent had hardwood pulp markets. In Ohio, 44 percent had a market for firewood and 33 percent had a market for hardwood pulp.

It is generally thought that differences in the number of roundwood markets available to a logging contractor at any given logging site will influence the logger's ability to optimally utilize the harvested trees. In this study, approximately one-half of the logging operations had a limited number of markets (20 loggers had only one or two markets) and the rest (19 loggers) had more merchandising options (three or more). The average overall utilization for operations with only one or two markets was 79.7 percent. For operations with three or more markets, utilization was higher (Table 2).

Table 2.—Number of roundwood markets and average overall utilization for all logging operations sampled in the West Virginia and Ohio region.

Markets	No. of logging sites (n)	Ave. utilization (%)
Few (1 or 2)	20	79.7
Several (3, 4, or 5)	19	85.2
Total	39	82.4

A two-factor ANOVA was performed that included the classification variables *State* (*two levels*) and *Markets* (*two levels: few and several*). The test yielded a p-value for the overall model of 0.0003. The variable *State* was highly significant ($p = 0.0003$). The *Markets* variable, with a p-value of 0.0774 was significant, as was the interaction term *State* $\times$ *Markets*, with a p-value of 0.0486.

This result suggests that if a logging operation has more rather than fewer markets available for sales/shipment of roundwood and chips, the logging utilization rate is likely to be improved. The test results also indicate that the effect of markets on the level of utilization is different for sites from Ohio versus West Virginia. State-based evaluation of utilization for the two-level market variable shows that the difference in utilization between the groups is much greater in Ohio than in West Virginia.

HAUL DISTANCE TO MARKETS

The distance from the logging site to the product market affects the profitability associated with different roundwood products. The effect typically is not as large as the effect of changes in the price of roundwood products, but it is real and meaningful. Frequently, studies will allude to the “maximum feasible haul distance,” which is the greatest distance that a logging contractor can transport logs to specific markets. If roundwood products are high-value, roads are in good condition such that travel time is minimized, and trucks can be fully loaded, the feasible haul distance is longer than in cases where these conditions are sub-optimal.

Just as all these transportation issues factor into the equation for estimating the maximum feasible haul distance, they impact the total manufacturing cost for roundwood. If the logging site is closer to the market, the trucking and related costs are lower, and the profit margin for a load of roundwood products is higher. Therefore, a logging operation with a market in close proximity can spend more time and effort in the woods and at the landing to recover greater amounts of roundwood product than would be the case if the same market were more distant. Whether loggers consciously behave in this way is not clear, but the possibility that they do provides the rationale for evaluating utilization relative to haul distances.

The average haul distances for the region are charted in Figure 3. The bars in the figure show what we expect—the average haul distance for lower-valued products (scragg logs and fencing logs) is shorter than the average haul distance for the higher-value logs. A question that arises from this figure is why the haul distance to pulp markets (pulpwood and chips, also a low-value product) appears to be one of the longest average distances. Most sites have a lot of material of pulpwood size and quality that needs to be harvested, and there is a wide geographic distribution of plants that buy pulp logs and chips. For harvest sites where these lower-value markets are closer in, the utilization rate of limbs, tops, minor species, and defective sections should

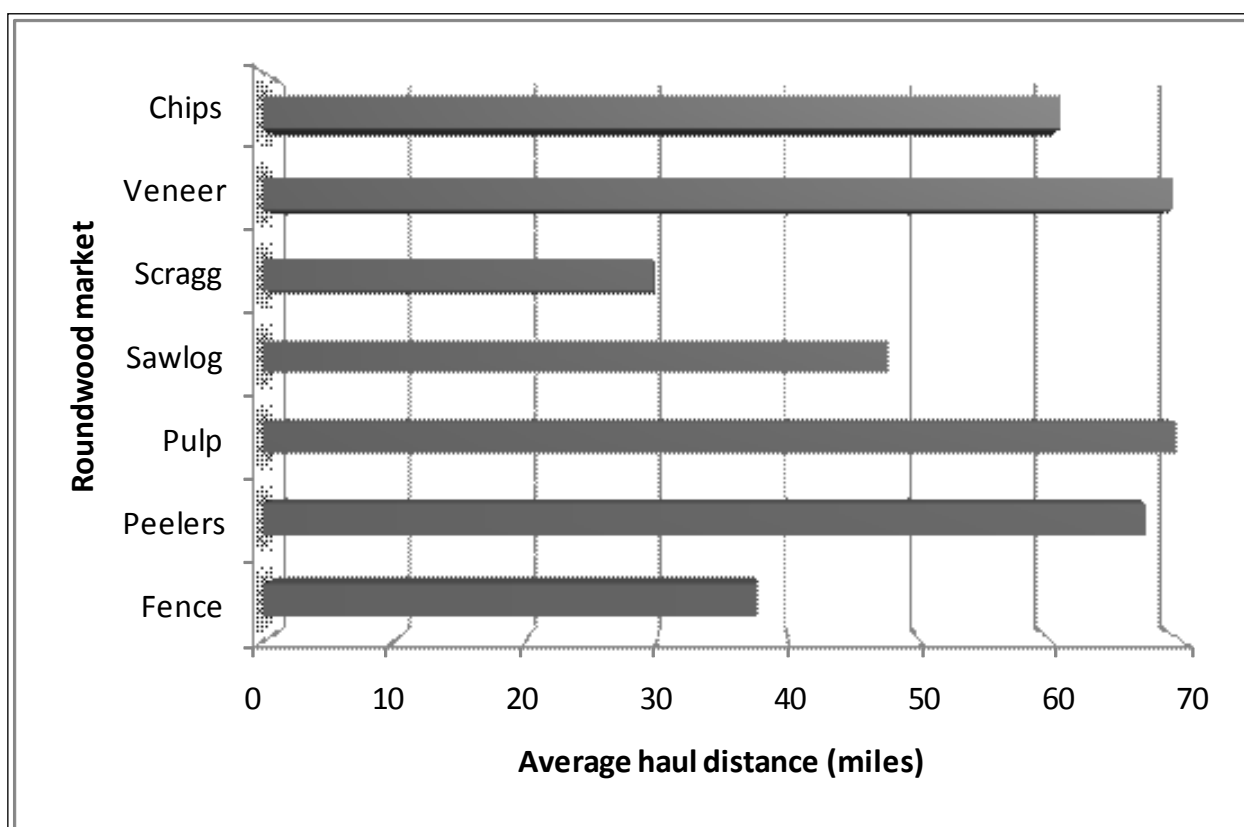


Figure 3.—Average distance from log landing to primary processing facility for 39 logging operations in West Virginia and Ohio.

be higher than for cases where these markets are closer to the maximum haul distance. When pulpwood markets are at hand, loggers report their pulpwood sales often provide the profit on their logging jobs—saw log markets alone are, at best, a breakeven proposition in today's market. For pulpwood/chip products, the average haul distance to market was 68 to 69 miles in both West Virginia and Ohio.

Since all 39 logging operations produced saw logs, haul distance from the logging site to the sawmill also is an important factor to consider. The average haul distance from the West Virginia harvest sites to sawmill markets was shorter than the distance from the Ohio sites. West Virginia loggers visited in this study hauled their saw logs an average of 45 miles while Ohio loggers hauled theirs 56 miles on average.

SPECIES-BASED DIFFERENCES IN UTILIZATION RATES

The distribution of tree species harvested at the nine study sites in Ohio varies from the distribution encountered at the 30 West Virginia sites in some important ways that may help explain differences in utilization. In Ohio, the percentages of harvested trees sampled that were black cherry and red oak were more than 5 percent higher than in West Virginia (Fig. 2). Conversely, in West Virginia, the percentages of harvested trees that were yellow-poplar and soft maple were notably higher than in Ohio (Fig. 2).

Table 3 shows the summary breakdown of utilization by species alongside the overall species distribution information for all 39 sites. Pine stems were utilized at a rate of 94.9 percent—the highest rate among all species—but they represented only 6.3 percent of the sample. Yellow-poplar stems ranked second in

Table 3.—Logging utilization, by species group, and sample tree species distribution, for 39 study sites in West Virginia and Ohio.

Species group	Regional utilization %	Utilization rank (1=highest)	Regional no. of trees	Percentage of sampled trees
Pine	94.9	1	68	6.3
Yellow-poplar	88.0	2	227	20.9
Mixed hardwoods	84.9	3	92	8.5
Soft maple	84.6	4	137	12.6
White oak	79.8	5	207	19.0
Hickory	79.5	6	50	4.6
Red oak	78.3	7	126	11.6
Hard maple	73.3	8	86	7.9
Soft hardwoods	71.4	9	32	2.9
Cherry	68.0	10	62	5.7

utilization and were the most frequently encountered (20.9 percent of the sample). Of all species groups accounting for 10 percent or more of the sample, red oak's overall utilization rate was the lowest at 78.3 percent. High utilization rates for pine and yellow-poplar can be attributed to two factors: 1) markets for these two species (groups) are dominated by pulpwood markets, which allow utilization of greater amounts of the tree; and 2) the growth form of these two species is a dominant, straight, long bole. The excurrent (single bole) form tends to have fewer defective sections than do trees with a deliquescent (forking) growth form. Branching defects and stem crook often have to be removed from trees with deliquescent form.

The degree of difference in utilization rates, by species, noted in Table 3, and the difference in the species' distributions of the West Virginia and Ohio samples, depicted in Figure 2, suggests that comparing the utilization rates of the different species groups for each state may inform our analysis of factors impacting logging utilization. Figure 4 provides side-by-side comparisons of this information. Analysis must take into account the relative size of the felled tree sample in the two states—75 percent of the 1,087 stems measured were in West Virginia while only 25 percent were in Ohio.

In both West Virginia and Ohio, the pines were the most fully utilized but they made up only 7 and 4 percent of the sample, respectively (Fig. 4). If Ohio's pine percentage were the same as West Virginia's, the overall logging utilization rate for the nine Ohio sites, which was 67.6 percent, would have been about 1 percent higher.

Yellow-poplar, having the second highest overall utilization rate, is substantially different both in terms of utilization and proportional representation in the sample, between the two states. While yellow-poplar was utilized at a rate of 89 percent at logging sites in West Virginia and represented 26 percent of the sample, its utilization rate in Ohio was only 76 percent and it composed only 7 percent of the felled tree sample (Fig. 4). These results would appear to be a clear case of how market availability impacts logging utilization. In both West Virginia and the two FIA regions of eastern Ohio where sampling was conducted, the highest-volume species, both in terms of growing stock and sawtimber, on forest land is yellow-poplar (Widmann and others 2007, Widmann and others 2009). In West Virginia, where OSB and peeler markets create strong demand for yellow-poplar, the change in yellow-poplar growing stock volume on forest land increased only 4 percent

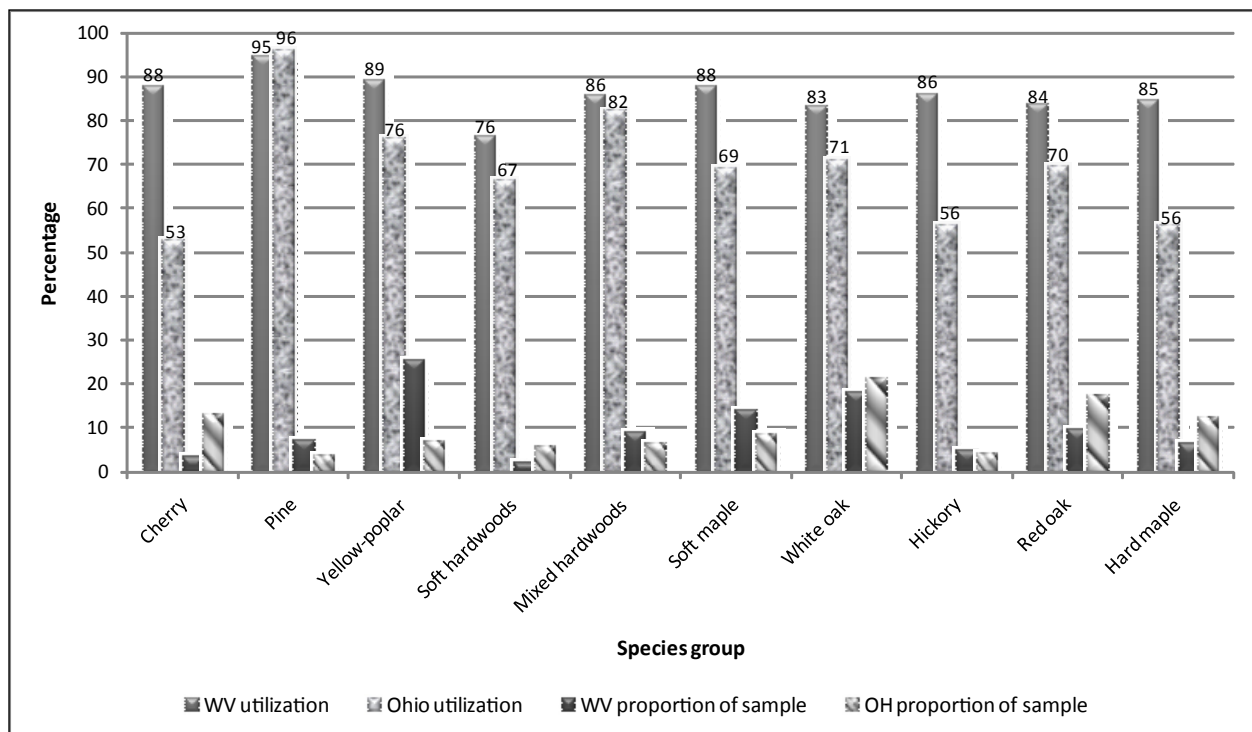


Figure 4.—Utilization rates and sample proportions, by species, for West Virginia and Ohio.

between 1989 and 2000 (Widmann and others 2007). In Ohio, where OSB and peeler markets do not exist, the yellow-poplar growing stock volume increased by 45 percent between 1991 and 2006 (Widmann and others 2009); yellow-poplar is not being harvested to the same extent in Ohio as in West Virginia despite its availability.

The utilization discussion for soft (red) maple is very similar to that for yellow-poplar. The same West Virginia OSB markets that procure yellow-poplar also utilize soft maple. Even the closest regions of Ohio (i.e., surrounding Cambridge and Marietta) are more than 125 miles from the two West Virginia OSB facilities. As a result, we see soft maple utilization rates in West Virginia of 88 percent, representing 14 percent of the harvested stems, as compared to 69 percent utilization for 9 percent of the harvested stems in Ohio (Fig. 4). The yellow-poplar and soft maple utilization situation explains approximately half of the difference in average overall utilization between the two states.

A closer analysis of the data will be required to estimate possible causes of the remaining variation in utilization. Falling saw log prices between 2008, when the West Virginia data were collected, and 2009, when the Ohio data were collected, likely enters into the explanation. Differences in product specifications (log size and quality limitations) in conjunction with available volumes and market proximity should also be considered.

DIFFERENCES IN STUMP HEIGHTS AND DIAMETERS

Data collected on stump heights and diameters provide the final piece of information on species-based differences impacting logging utilization. The average stump height for all 815 felled trees sampled in West Virginia was 9.8 inches. In Ohio, the average stump was shorter, averaging 6.5 inches for 272 felled trees.

This observation indicates that utilization of the stump section of the tree was better at the 9 Ohio logging operations than at the 30 West Virginia sites. Factors that can affect stump utilization include felling equipment, feller skills, tree form, terrain, obstructive understory vegetation, tree diameter, and season of harvest.

The average stump diameter of the West Virginia tree sample was greater than that of the Ohio sample—20.6 inches compared to 19.8. While the value and volume associated with achieving higher utilization at the base of the tree are higher than for other parts of the tree, due to the higher quality and larger diameter of this part of the tree, larger-diameter trees are more challenging to fell. The reduction in recovered log volume associated with the 3.3-inch difference in stump height measured in West Virginia and Ohio averages 0.45 cubic feet per tree. This volume translates into a missed utilization opportunity of about 1 percent for West Virginia logging operations. Conversely, larger stump diameters typically mean larger diameters at other heights up the bole compared to trees with smaller stump diameters. Larger diameter trees tend to yield more marketable roundwood, thus increasing utilization rate.

Average stump heights and diameters for the 10 species groups are shown in Figure 5. The result of the paired sample t-test that was conducted to test the average stump heights for the species groups for West Virginia and Ohio was highly significant ($p \leq .0001$). Species with smaller average diameters have shorter average stump heights (Fig. 5). Correlation tests indicated the positive association between stump diameter and height is stronger for the Ohio data ($r = 0.62$) than for the West Virginia data ($r = 0.25$).

SUMMARY AND CONCLUSIONS

A series of 39 logging utilization studies was conducted during 2008 and 2009 in which the utilization, to a 4-inch top, of felled trees was measured. In total, 1,087 trees were measured—815 in West Virginia and 272 in Ohio. Sample trees were randomly selected from among the trees felled by the loggers at each site. The overall logging utilization factor for the region, determined in this study, is 82.3 percent.

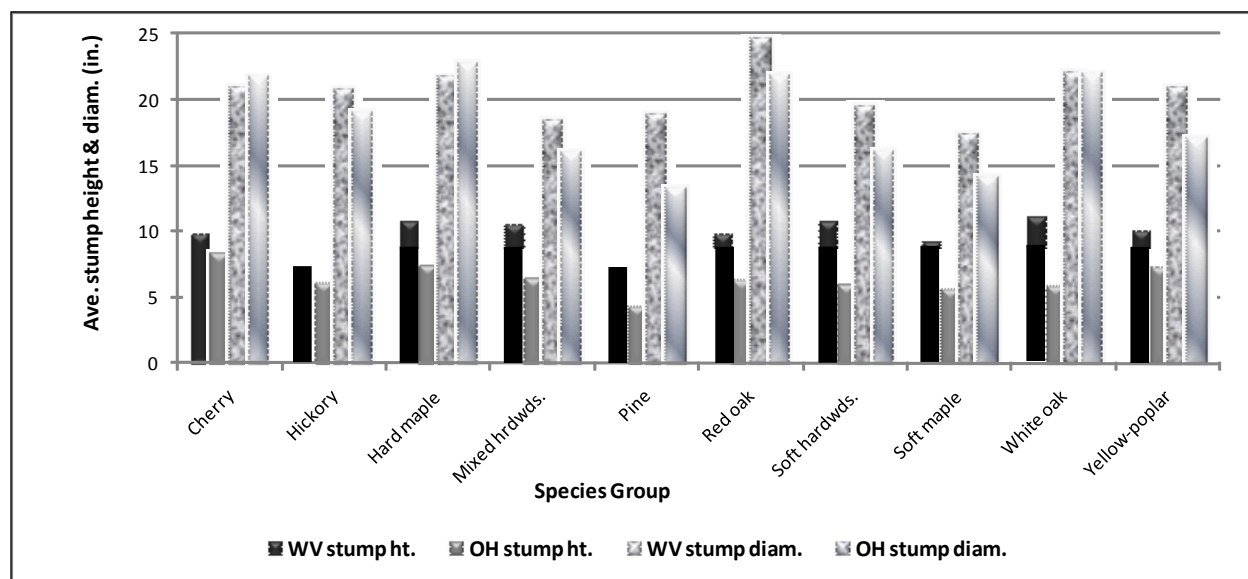


Figure 5.—Average stump heights and diameters for sample trees in West Virginia and Ohio, by species.

Average utilization rates for all species were significantly different for harvests in Ohio and West Virginia, with average rates of 67.6 percent in Ohio and 86.8 percent in West Virginia. In this study, one-half of the logging operations had a limited number of markets (one or two markets) and the rest had more merchandising options (three or more). Analysis of variance indicated that logging operations with three or more markets achieve higher utilization than operations with only one or two markets.

The distribution of tree species harvested at the 9 study sites in Ohio varies from the distribution encountered at the 30 West Virginia sites in some important ways that may help explain differences in utilization. Pine stems were utilized at a rate of 94.9 percent—the highest rate among all species—but they represented only 6.3 percent of the sample. Yellow-poplar stems ranked second in utilization and were the most frequently encountered (20.9 percent of the sample). Of all species groups that composed 10 percent or more of the sample, red oak's overall utilization rate was the lowest at 78.3 percent.

Yellow-poplar, having the second highest overall utilization rate, is substantially different both in terms of utilization and proportional representation in the sample between the two states. While yellow-poplar was utilized at a rate of 89 percent at logging sites in West Virginia and accounted for 26 percent of the sample, its utilization rate in Ohio was only 76 percent and it made up only 7 percent of the felled tree sample. This difference can be largely attributed to the availability of OSB and peeler markets in West Virginia.

The number and types of markets available to a logging contractor operating at a specific site is an important determinant of potential utilization on the logging job. The species of poletimber and sawtimber to be harvested at a site interact with markets to affect utilization potential. Species that tend to have straight, dominant boles can produce especially high roundwood product yields if demand exists for that species. Stump heights have an effect on logging utilization, but the effect is much smaller than the market and species effect.

The average stump height for all 815 felled trees sampled in West Virginia was 9.8 inches. In Ohio, the average stump was shorter, averaging 6.5 inches for 272 felled trees. The average stump diameter of the West Virginia tree sample was greater than that of the Ohio sample—20.6 inches compared to 19.8. While the value and volume associated with achieving higher utilization at the base of the tree are higher than for other parts of the tree, larger-diameter trees are more difficult to fell. The larger-diameter tree, however, can result in higher volume utilization, all other factors being equal.

A closer analysis of the data will be required to estimate possible causes of the remaining variation in utilization. Falling saw log prices between 2008, when the West Virginia data were collected, and 2009, when the Ohio data were collected, likely enter into the explanation. So too might differences in the product specifications (log size and quality limitations). Differences in the equipment used by the loggers also should be explored.

Logging utilization rates vary both temporally and spatially. Several factors—including size and location of markets, logger training, resource availability and quality, and landowner objectives—influence utilization rates. Developing timely estimates of logging utilization helps the forest industry understand resource availability and facilitates the development of new industries based on the characteristics of underutilized wood resources.

ACKNOWLEDGMENT

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

A SYNOPSIS OF THE WOOD-BASED ENERGY AND HEATING INDUSTRIES IN THE NORTHEASTERN UNITED STATES WITH CONSIDERATION OF POTENTIAL IMPACTS ON FUTURE DEMAND FOR ROUNDWOOD

Jan Wiedenbeck, Chuck D. Ray, and Li Ma¹

Abstract.—The project team identified 323 facilities in the northeastern United States that input pulpwood or “energy wood.” Of these, 88 are located in Pennsylvania, Ohio, and West Virginia, part of the central hardwood forest region. In the 13-state northeastern region, 81 percent of the facilities that use pulp-type roundwood produce an energy-related product. For the three states that are part of the central hardwood forest, 75 percent of the facilities are in business to produce an energy product while the other 25 percent produce a more traditional product (e.g., pulp, oriented strand board). The 323 operations identified in the Northeast potentially consume 40.6 million tons (50-percent moisture content) of wood per year, of which 12.5 million tons is consumed in the three central hardwood states. Forest resource concentrations and levels of industrialization suggest that collection of data on future woody biomass demand be focused on five key states: Maine, New Hampshire, Vermont, New York, and Pennsylvania. Small-scale bioenergy projects pose no real threat of significantly reducing the general supply of woody biomass in the Northeast. The ongoing decline in pulp and paper production together with the apparent decline in other traditional woody biomass-using industries, will relieve the pressure on the woody biomass resource. Assuming a normal and expected evolution of events, woody biomass consumption in the region will increase by roughly 25 percent over the next decade. The variable that could most affect the woody biomass resource is the future direction of electricity production from wood and co-firing of wood in coal power plants, especially in Pennsylvania, West Virginia, and Ohio.

INTRODUCTION

The future energy landscape will be a product of current energy demographics within a given state and region, state and federal policies, renewable resource availability, and consumer demand. Our objective in this study was to estimate how wood-based energy might be a part of the future energy landscape in the northeastern United States. Knowledge of the current distribution of wood-based energy systems and regional trends in biomass energy adoption provide important background information and insight toward fulfilling this objective. This information can help us estimate possible shifts in biomass demand that will be of interest to resource managers, existing wood products industries, and prospective entrants into the region's wood marketplace.

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Unfortunately, to accurately characterize the current status and trends for wood-based bioenergy is challenging and to project the future state of affairs is extremely difficult. The uncertainty involved in this undertaking is due to a myriad of factors, including: 1) the diversity of biomass-based energy technologies; 2) the incompatibility and incompleteness of state and regional forest products flow databases; 3) the wide array of regional and state policies toward energy production, emission controls, and forest management; and 4) the rate of change of policies and opinions.

Recognizing these challenges, we have included considerable discussion of the uncertainty factors that affect the following assessment of the current and future scenarios for wood use in bioenergy markets in the northeastern United States. The information in this report was obtained through extensive discussions and Web searches carried out over several months in 2008 and 2009.

RENEWABLE ENERGY GENERATION IN THE NORTHEAST

Five energy-consuming sectors must be considered in discussing energy supply and demand. These sectors are: 1) commercial, 2) residential, 3) industrial, 4) transportation, and 5) electric power generation. These sectors are listed in reverse order in terms of their size. In 2008, these sectors consumed 4, 7, 21, 28, and 40 percent of the energy used in this country, respectively (Energy Information Administration [EIA] 2009a). As the most important sector, electricity generation is a key focus area for energy policy-makers.

It is important to recognize, however, that the other four energy-consuming sectors, as the end-consumers of the product produced by the electric power generation sector, can influence the production and distribution of electricity. This is less the case for the commercial and transportation sectors than it is for the residential and industrial sectors. Not only do these two sectors consume electricity at greater rates, but some of the generated electricity that is included in statistics is generated by the industrial sector for in-house use. In some cases, industrial generators produce excess electricity that enters the grid. Thus, if these sectors shift to alternate heating sources or if they lend strong support to policies that promote greater use of renewable resources as inputs by the electric power generation sector, they can have a substantial effect on electrical energy generation.

The overall average percentage of electric power generation that was derived from renewable sources in the United States in 2007 was 8.5 percent (EIA 2009b). In the northeastern United States, renewables produced a higher percentage of the electricity generated—9.2 percent (adapted from EIA 2009b; Fig. 1)—despite the fact that only 4 of the 13 northeastern states met or surpassed the national average for percentage of in-state electricity generation derived from renewable sources (EIA 2009b). Substantially higher levels of renewable use in the New England states is the reason the Northeast exceeds the national average. Maine leads the region in renewable energy generation and exceeds the national average by a very large margin. In fact, the proportion of Maine's energy that is non-hydroelectric renewable energy is the highest in the nation. Nationwide, hydro-electricity is the number one form of renewable electricity and wood is the second greatest contributor of renewable electricity (Fig. 1).

In 12 of the 13 northeastern states, hydro-electricity is either the most important or the second most important renewable energy source for the electric power industry. In eight of these states, landfill gas ranks as one of the top two renewable energy sources. Only in Maine, New Hampshire, Vermont, and Ohio is wood one of the two most important renewable sources of electricity. In 2007, wood constituted 23.9, 4.2, 7.8, and

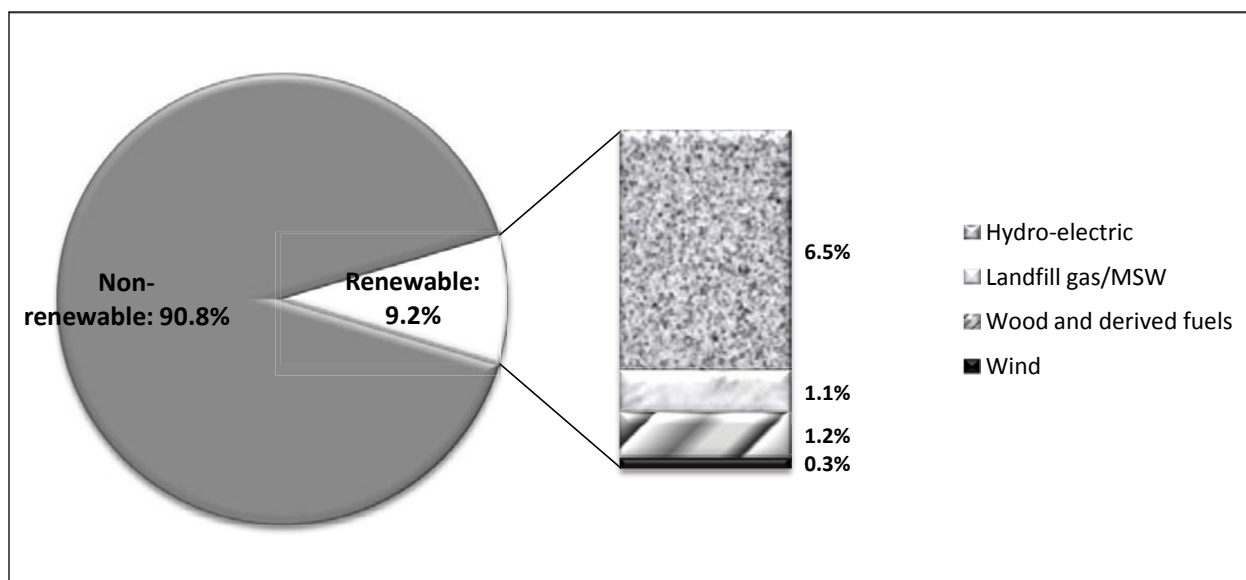


Figure 1.—Resources used to generate electricity in the 13 northeastern states in 2007 (EIA 2009b).

0.3 percent of total energy generation for Maine, New Hampshire, Vermont, and Ohio, respectively (EIA 2009b). Except for Ohio, the proportional use of wood rose significantly between 2003 and 2007 in these states.

CONSUMPTION OF RENEWABLES

Total U.S. consumption of renewable energy rose 7 percent in 2008 at the same time total energy consumption declined by 2 percent (EIA 2009c). In 2008, energy sourced from renewable resources provided 7.3 percent of total U.S. energy consumption. Among the five market sectors for energy (electric power, transportation, industrial, residential, and commercial), the electric power and industrial sectors consume the largest proportion of renewable energy—51 and 28 percent, respectively (EIA 2009c). The transportation sector's share of renewable consumption (e.g., biodiesel and ethanol) rose significantly in 2008 and now stands at 11 percent. Residential markets consume 8 percent of U.S. renewable energy and the commercial sector consumes the remaining 2 percent (EIA 2009c).

The proportional breakdown of sources of energy consumed (in all sectors) in the Northeast is rather similar to the breakdown for the entire United States (EIA 2009a). The Northeast consumed a slightly lower proportion of energy derived from natural gas and a slightly higher proportion of energy derived from nuclear generation than the rest of the United States. However, when we look at a slightly broader landscape to include Virginia, Kentucky, and Tennessee, and reconfigure the groups being analyzed, important distinctions appear.

For the group of states in the area referred to as the “Central Appalachian Region” (Pennsylvania, West Virginia, Ohio, Virginia, Kentucky, and Tennessee), coal is the predominant source of total energy consumed (32 percent; Fig. 2) while the percentage derived from natural gas is lower than for the rest of the United States (EIA 2009a; Fig. 2). With the removal of Pennsylvania, West Virginia, and Ohio from the group of 13 northeastern states, the energy consumption profile for the remaining 10 states changes. Most notably, petroleum-based energy consumption levels are substantially higher than for the rest of the United States (Fig. 2) because a large proportion of households in the region use petroleum-based heat.

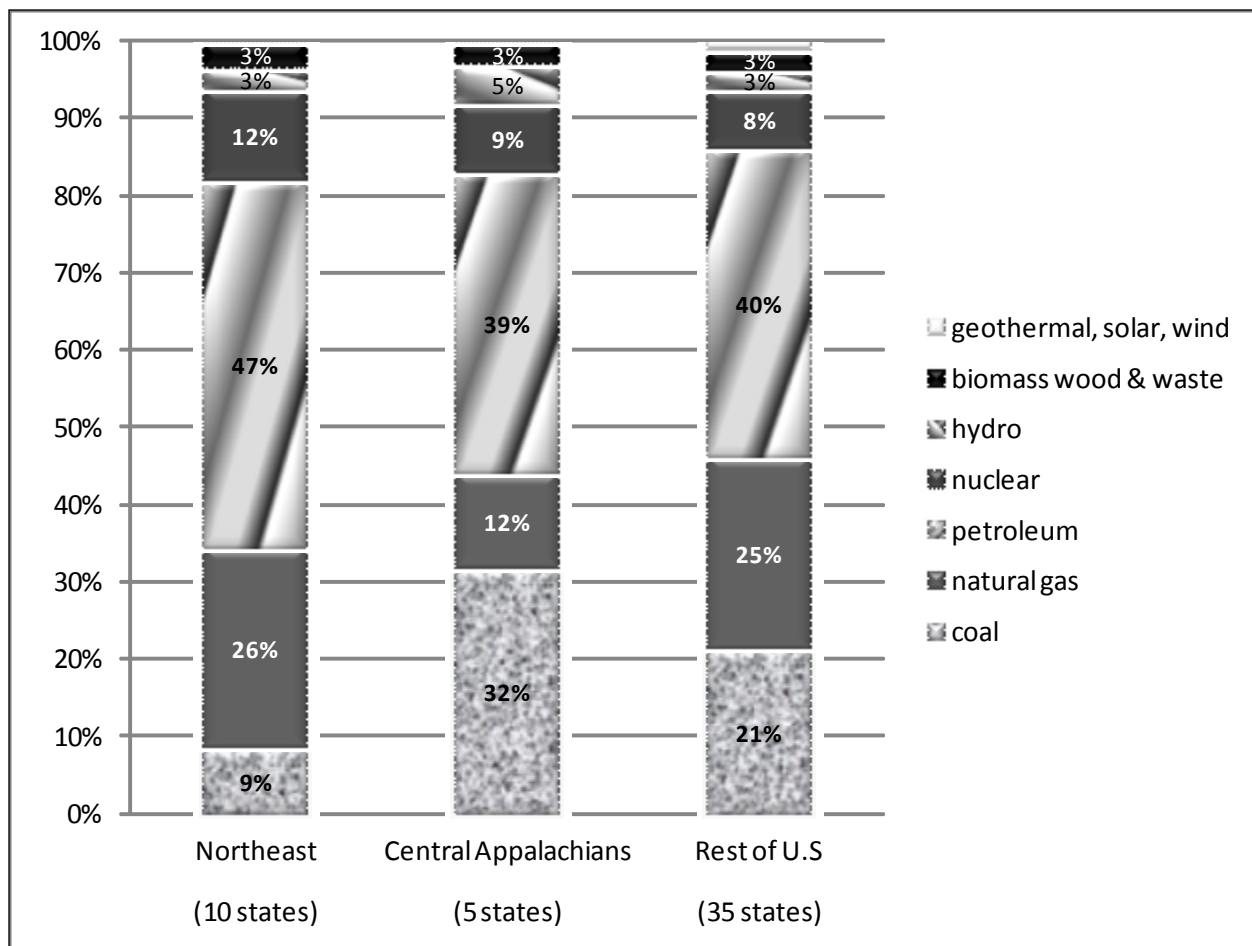


Figure 2.—Renewable energy consumption by all five sectors in the 10 northeastern states, 5 central Appalachian states, and the rest of the United States, 2007 (EIA 2009a).

USES OF PULPWOOD AND ENERGY WOOD IN THE NORTHEAST

CURRENT LANDSCAPE

To date, we have identified 347 current and planned woody biomass conversion facilities for the 13 northeastern states. Of these, 323 are currently in production and consume an estimated 40.6 million tons of wood per year (at 50-percent moisture content - wet basis), when operating at capacity. The total amount of wood consumption could rise to an estimated 53.6 million tons in 2015 as the 24 planned facilities that are not yet operating are brought on-line.

Figure 3 shows the distribution of production facilities in the northeastern United States that consume woody biomass of the sort that might be used to produce wood-based bioenergy. These biomass types include: pulpwood, logging residues, mill residues, whole-tree chips, limb wood, and urban forest residues. Some of the facilities shown on the map are producers of wood energy while others produce non-energy products (“traditional products”). Very generally, we can see that the New England States have been more inclined to utilize wood for electricity and pulp and paper production; ethanol and wood pellets projects have been more directed at the Pennsylvania and New York area; and wood combustion for heat is common throughout the entire northeastern region, although less so along the coastal plain.

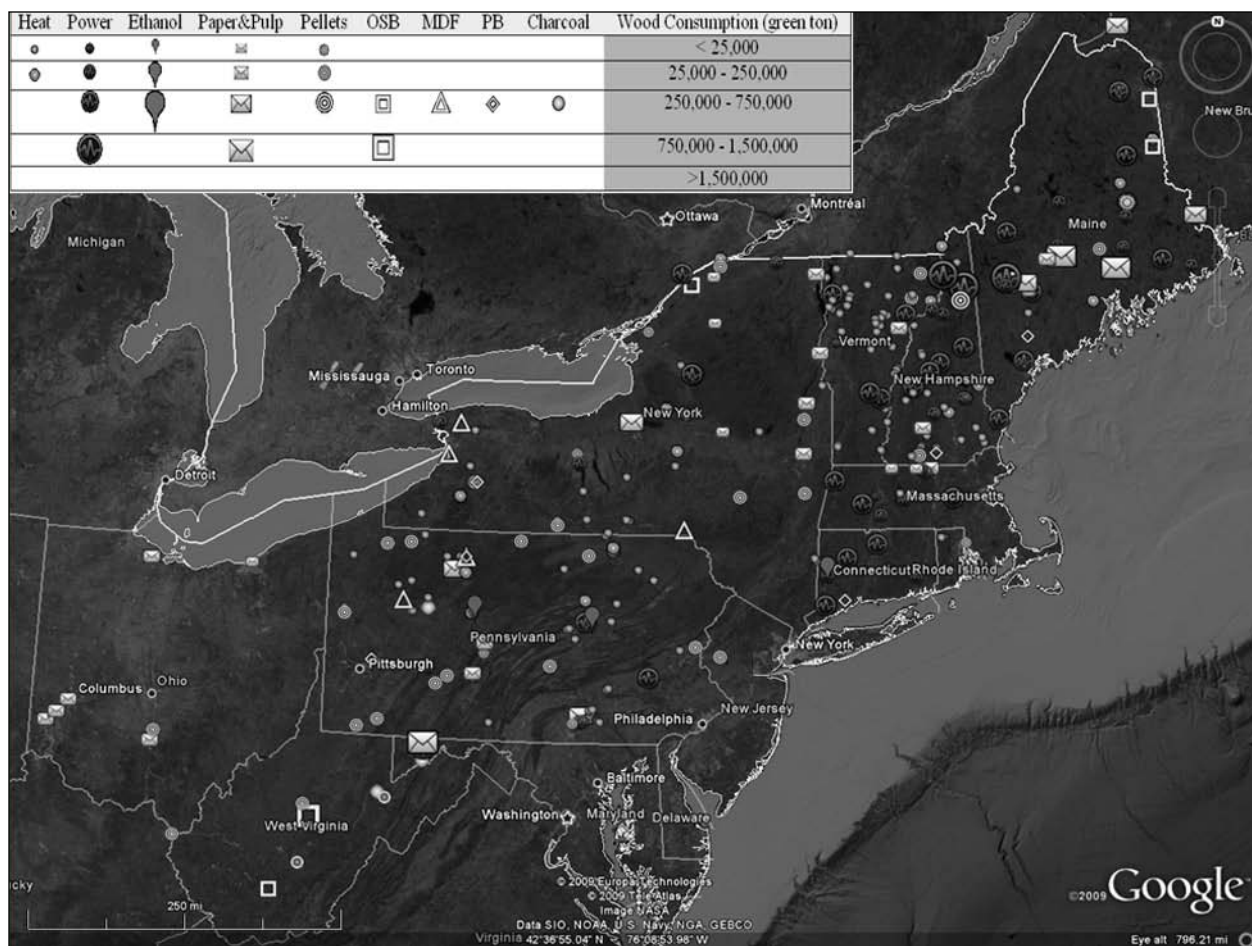


Figure 3.—Location, type, and size of woody-biomass consuming facilities, both existing and planned, in the northeastern United States. (OSB = oriented strand board; MDF = medium density fiberboard; PB = particle board.)

Focusing on the wood and energy industry demographics for the 4 states (Pennsylvania, Maryland, Ohio, and West Virginia) in this 13-state region that are part of the central hardwood forest landscape, 82 woody biomass-using facilities were identified. Of these, 89 percent are in business to produce an energy product while the other 11 percent produce a more traditional product (e.g., pulp, oriented strand board [OSB]) that might be competing for some of the same types of resource.

Inventory of woody biomass-using facilities by product type

Figure 4 gives the percentage-based distribution of the number of facilities of different types converting woody biomass into wood or energy products in the northeastern United States in 2009. This figure also shows the relative wood consumption (based on green weight) for the different biomass conversion sectors considered in this analysis.

The concern is commonly expressed that new wood-based heating and pellet manufacturing enterprises are going to lead to over-utilization of the forest biomass resource in the northeastern United States. Wood-based heating systems now represent roughly half the total number of operations that utilize pulp and energy wood in the Northeast. Figure 4 shows that these 179 operations utilize only 9 percent of all the woody biomass consumed in the region. Extrapolating, it appears that even if the number of biomass heating facilities

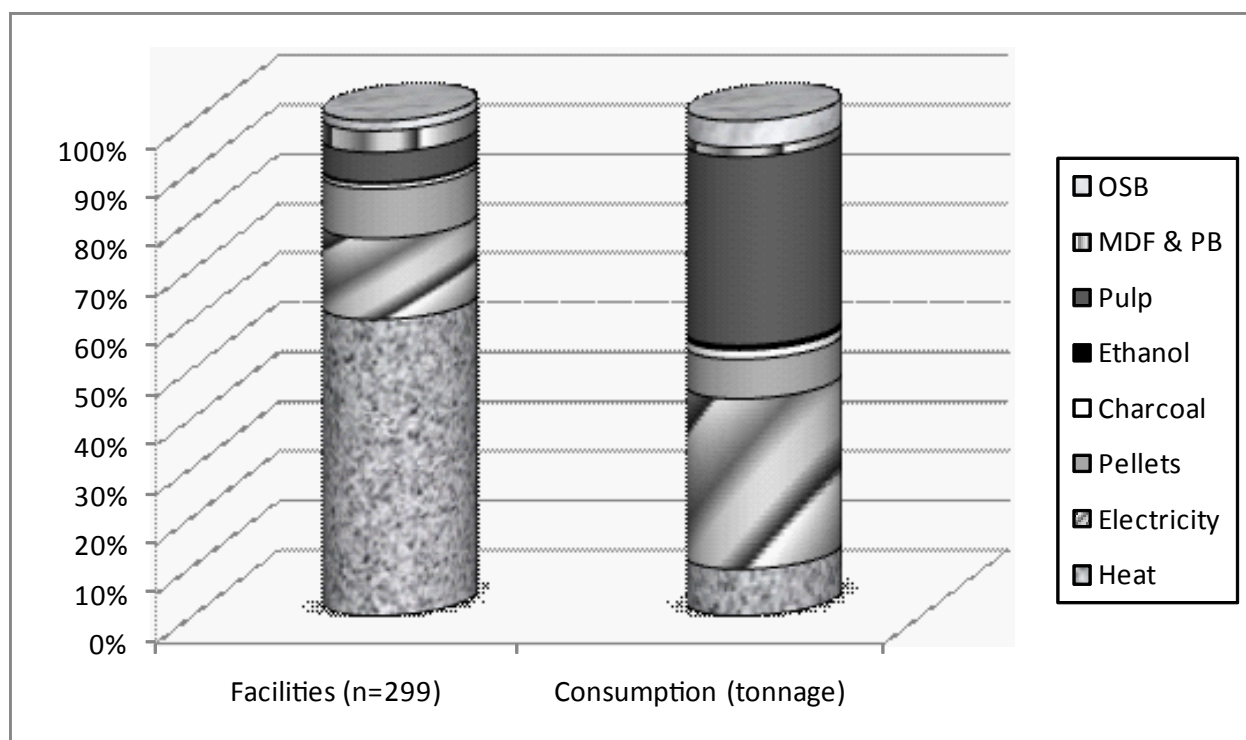


Figure 4.—Distribution of types of conversion operations utilizing pulp-type wood in the 13 northeastern states in 2009 in terms of numbers of facilities and wood tonnage requirements.

quadruples over the next decade, these district heating projects, schools and hospitals, and industrial wood boilers will still only consume roughly the equivalent of one-fourth of the tonnage of woody biomass currently demanded in the Northeast. Therefore, fears that expansion of current initiatives such as *Fuels for Schools* and community district heating projects will quickly impose hardship on the forest resource appear unfounded. Expansion of wood-based heating in the Northeast nonetheless may well have an impact on localized biomass roundwood and residue markets. However, significant shifts in energy policy could ratchet up demand for wood-based energy sources (such as pellets) such that large-scale wood-energy processing facilities could become a part of the region's wood utilization landscape.

Another important observation from Figure 4 is the much larger impact, per facility, that pulp mills and power plants have on the forest resource compared to other types of operations. Pulp mills constitute only 6 percent of the number of woody biomass utilizing operations, but they currently consume 38 percent of the total woody biomass (by weight). Power plants, in comparison, account for only 17 percent of the number of facilities, yet consume 34 percent of the woody biomass. The implication of these data is that each pulp mill and power plant has a significant impact on overall woody biomass availability.

Of the 40.6 million green tons of woody biomass currently consumed in the 13 northeastern states, 45 percent is consumed by traditional, non-energy industries. For the three central hardwood region states among this group of 13 northeastern states (West Virginia, Ohio, Pennsylvania), an estimated 11.1 million green tons of woody biomass were consumed in 2009. Traditional, non-energy industries consume 62 percent of the input volume in these states. Demand for pulpwood by large pulp and OSB operations and the current lack of wood-fired electricity plants explain the higher percentage of biomass going to traditional, non-energy industries in these three states compared to the rest of the northeastern region.

The siting of future woody biomass energy facilities in the vicinity of traditional wood products operations that have closed or curtailed production allows new ventures to capitalize on existing supply chain infrastructure, thereby minimizing costs associated with biomass acquisition. However, as new wood bioenergy facilities are built in coming years, to the degree that they increase demand for biomass faster than traditional wood consumption declines, there will be upward market pressure on wood prices. This price pressure will make it even more difficult for the traditional woody biomass-based facilities to remain competitive.

FUTURE LANDSCAPE

A portion of our database contains information on announced but not yet permitted and/or built and operating, wood-based bioenergy installations. Based on data sources, future woody biomass demand (considering both the traditional and energy sectors) in the Northeast is projected to see a net increase of 32 percent by the year 2015. The increase in consumption is most dramatic over the next 2 years, after which the rate of growth slows. This growth curve is largely a function of the fact that the planning and start-up cycle for many wood bioenergy facilities is not much longer than 2 years—many facilities that may open in the more distant future have not yet been publicly announced and in some cases not yet conceived.

This projected 32-percent growth in woody biomass consumption for the region is based on four important assumptions: 1) all announced facilities will be built to their announced capacity; 2) no other unannounced facilities will start up during this time; 3) the consumption of woody biomass by all current operations stays the same; and 4) no new wood co-firing by existing power plants, aside from those that have already been announced, will occur. Of course, all of these assumptions are likely to be violated to a greater or lesser extent. Therefore, this portrayal of the future biomass utilization landscape informs our current understanding but is subject to annual revision.

Potential impact of electricity production from wood

Use of wood for electricity generation is the major use of wood for bioenergy in the northeastern United States. Currently, wood-fired electricity capacity ranges from 0 megawatts (MW) in five states (Connecticut, Rhode Island, New Jersey, Ohio, and Delaware) to 600 MW in Maine. The increased demand in woody biomass required to meet the anticipated increase in electricity generation from wood will be significant, particularly in the New England states and Ohio. In terms of biomass consumption, the most significant projected increase is expected for power plants in Ohio (from 0 tons per year currently to 3.7 million tons/year), Massachusetts (from 0.3 million tons/year to around 2.25 million tons/year), New Hampshire (from 2 million tons/year to 4.5 million tons per year), and Connecticut (from 0.5 million tons/year to 1.3 million tons/year). Since Massachusetts' traditional wood products industry is relatively small, existing wood residues will not be sufficient to supply these new electricity-generating facilities. Feedstock requirements for the new facilities in New England are projected to result in a demand increase in the region of 3.5-4.0 million tons per year. This sharp increase in demand could cause considerable tightening of the local woody biomass market.

The Energy Information Administration (EIA 2009a) is projecting a large increase in biomass demand for energy production over the next decade and it projects this increase will come both from dedicated biomass facilities and from coal-fired production facilities beginning to co-fire with wood. To analyze the potential impact of this co-firing scenario, a sensitivity analysis was performed at varying levels of conversion to co-firing.

The sources for electricity production are unmistakably different across the Northeast. Wood-based electricity production is common in the New England states while coal-fired production dominates in the southern tier of the region (Fig. 5). Though nuclear, hydroelectric, and other forms of electrical capacity are important sources of power generation in the region, they have been omitted from Figure 5 so that wood and coal-based operations can be readily distinguished. This distinction is desired because, unlike nuclear and hydroelectric generation systems, coal-based systems can be partially or completely converted to wood-based systems.

What is not so apparent in looking at Figure 5 is the relative size and capacities of the wood-fired and coal-fired energy industries; this information is crucial for estimating the potential impact of future conversions to co-generation by the coal industry. In fact, the wood-fired electricity capacity of states such as Maine, New Hampshire, and Vermont is dwarfed by the total coal-fired power capacity in the other states of the Northeast. This comparison holds true even when the power plants that are planned and permitted but not yet on-line are included. It would be a mistake to focus solely on these new woody biomass energy projects because the potential impact of converting a coal-only plant to one that co-fires with wood and coal is probably the most important component of any discussion surrounding future forest sustainability in the Northeast.

Just a few years ago, the idea that coal-fired power plants would convert even a tiny percentage of their production to wood co-fired capacity seemed implausible. Today, state-mandated renewable portfolio

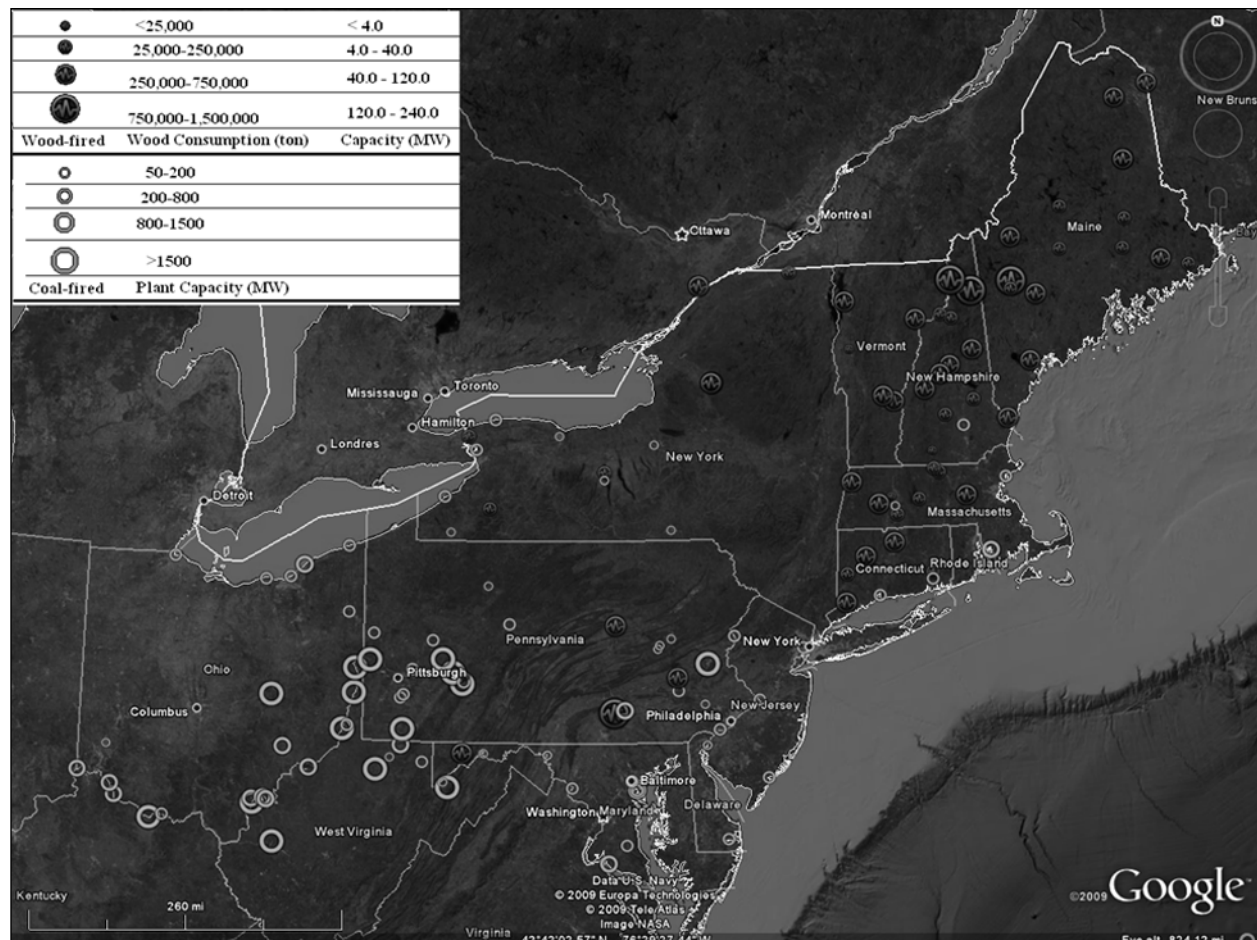


Figure 5.—Occurrence, size, and location of wood-fired and coal-fired electricity production in the northeastern United States – current and planned facilities.

standards and incentives for renewable energy (such as cap-and-trade or carbon tax schemes) are provoking many companies that own coal-based power plants to evaluate the costs and benefits of co-firing. Some of these evaluations have led to the decision to begin to do so.

Co-firing scenario assessment

Recognizing that future co-firing of coal power plants with wood is likely the biggest determinant of future demand for woody biomass in the region, we had to estimate the degree of impact associated with different co-firing scenarios.

The assumptions used to develop the wood volume estimations in these scenarios were:

- Energy content of biomass – 8,600 Btu/lb (dry).
- Efficiency of power plant – 33 %.
- Hours of operation – $365 \times 24 \times 80\%$ per year. 1 kwh = 3,412 Btu

Thus, to estimate the weight (bone-dry tons) of woody biomass required to produce X trillion Btu's:

$$\frac{X \times 10^{12} \text{ Btu}}{8,600 \text{ Btu/lb}} \times \frac{1 \text{ ton}}{2,000 \text{ lb}} \times \frac{1 \text{ million ton}}{10^6 \text{ ton}} = \frac{X}{17.2} \text{ million bone-dry tons}$$

Then, to estimate wood biomass requirements for a power plant with capacity of α MW:

$$\text{Woody biomass demand} = \frac{\alpha \times 10^3 \times 7,008 \times 3,412}{0.33} \div (17.2 \times 10^6) \text{ bone-dry tons}$$

Using these calculations and the information in our database on existing coal-fired power plants, we estimated the potential impact of different levels of wood use in co-firing these plants.

If wood co-firing of existing power plants reaches only a 2-percent level across the industry, the impact is relatively minor—only Ohio, Pennsylvania, West Virginia, and Maryland would see a resource impact of more than 1 million green tons annually. Once wood co-firing reaches 3 percent, however, the impact on wood demand in Maryland (1.7 million green tons), Ohio (7.3 million green tons), Pennsylvania (6.4 million green tons), and West Virginia (4.6 million green tons) begins to reach a level where co-firing facilities will negatively impact the competing wood-using industries. Policies and incentives that promote large-scale wood-fired electricity co-generation (e.g., ≥ 5 percent) could raise roundwood harvest in Maryland, Ohio, Pennsylvania, and West Virginia such that the growth-to-removals ratio approaches or dips below one. Currently, Maine is the only state in the region with a tenuous growth-to-removals ratio. While a growth-to-removals ratio of less than 1:1 may be justified on a short-term basis, it is unsustainable if continued for the longer term that would be necessary to support these large wood co-firing projects.

Therefore, policy incentives that promote wood co-firing of existing power plants could have a negative impact if they result in co-firing these facilities at a rate in excess of 5 to 6 percent (BTU basis), on average. If utilities are able to pass on to consumers the fuel cost increases associated with higher demand for woody biomass, they may be in a position to out-compete other woody biomass users. The inefficiencies in the generation of electricity by co-firing with wood also need to be considered; power generation in power plants typically achieves less than 30 percent efficiency, which will be decreased with the introduction of wood into the feedstock. By contrast, combined heat and power plants typically can achieve efficiencies of 60 percent.

Potential impact of ethanol production from woody biomass

Even under the most bullish of biomass-based energy projections of the Energy Information Administration, production of ethanol from woody biomass is projected to be only a small portion of the overall demand on biomass. Current technology limitations on the conversion of woody biomass to ethanol are clearly constraining industry development. Even assuming eventual technological breakthroughs that make cellulosic ethanol a sensible and profitable use of the resource, significant impact on the woody biomass resource from ethanol production appears to be at least a decade away.

Most current research is focused on cellulosic feed stocks such as corn stover, bagasse, and switchgrass, or single-species wood chips such as aspen, willow, or hybrid poplar. These fast-growing crop or plantation species have physical and chemical properties that make the chemical conversion from cellulose to sugars theoretically attainable in an efficient and cost-effective manner. However, *woody biomass from natural forests has more diverse and “contaminated” properties*; this woody biomass has yet to be proven in the laboratory, much less at the pilot scale, as economically feasible to process.

Currently, there are six ethanol operations in the database that have announced plans to use wood as a feedstock. These six facilities would consume 11 percent of the current woody biomass demand if they were to reach announced production levels. This scenario is highly doubtful, especially given the volatility (and decline in 2008) in oil and gasoline prices, and the de-emphasis during 2009 of policies to promote ethanol production.

CONCLUSIONS

Pulp and paper operations, as well as OSB, medium density fiberboard, and particle board plants, currently account for roughly 50 percent of the woody biomass consumption in the northeastern United States. The ongoing decline in pulp and paper production in the Northeast, together with the apparent decline in other traditional woody biomass-using industries, will work to relieve the pressure on the region's woody biomass resource. It is possible that future consumption of woody biomass by bioenergy facilities can be supplied by biomass that will become available as a result of the declining consumption by traditional wood products manufacturers. Unfortunately, in some regions, it may be that upward price pressure on the woody biomass market owing to increases in the needs of the bioenergy industry could hasten the decline of the traditional wood industries.

Small-scale bioenergy projects pose no real threat of significantly reducing the general supply of woody biomass in the Northeast, but areas of heavily concentrated projects could experience local demand pressure.

The future direction of electricity production from wood and co-firing of wood in coal power plants is the variable with the highest potential impact on woody biomass consumption. Government policies and incentives that encourage use of woody biomass by large electricity producers, either through dedicated use of wood or through co-firing of coal power plants at a 5-percent or higher level of co-firing, can be expected to put pressure on raw material availability for other industry segments and, over time, wood resource sustainability in the affected regions.

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

CHANGES IN U.S. HARDWOOD LUMBER EXPORTS, 1990 TO 2008

William Luppold and Matthew Bumgardner¹

Abstract.—The volume of hardwood lumber exported from the United States grew by 63 percent between 1990 and 2006 before decreasing by 29 percent between 2006 and 2008. Canada is both the largest export market for U.S. hardwood lumber and the largest source country for hardwood lumber imported into the United States. In the last 19 years China/Hong Kong has displaced Japan as the second most important export market for U.S. lumber. Nearly 60 percent of the hardwood lumber exported in 1990 was red or white oak, but exports of these species had decreased by 32 and 22 percent, respectively, by 2008. By contrast, exports of yellow-poplar, black walnut, and maple have increased by 252, 242, and 141 percent, respectively. In 1990, exports accounted for 6.8 percent of domestic production compared to 9.8 percent in 2008. However, the export of individual species as a percentage of domestic production varies widely. The change in export trading partners and the species exported between 1990 and 2008 is the result of a shift of secondary hardwood manufacturing capacity from the United States to China/Hong Kong, Mexico, and Vietnam, and changes in furniture fashion trends from open-grained to closed-grained species.

INTRODUCTION

The volume of hardwood lumber exported from the United States grew by 63 percent between 1990 and 2006 but decreased by 29 percent between 2006 and 2008 (Fig. 1). During this period Canada has consistently been the largest export market for U.S. hardwood lumber, but volumes shipped to other trading partners have changed considerably. In 1990 exports to Japan and Taiwan were the second and third most important markets for U.S. hardwood lumber, respectively, and European demand accounted for 35 percent of the export market (Fig. 1). By 2008, China, Japan, and other Asian nations accounted for 35 percent of U.S. exports while Europe accounted for 20 percent (Fig. 1). China/Hong Kong² was the second most important market after Canada (Table 1).

As hardwood lumber exports have grown, the species exported have changed. Nearly 60 percent of the lumber exported in 1990 was red oak (*Quercus rubra* L.) or white oak (*Q. alba* L.); red oak was the most commonly exported product (Table 2). The volume of lumber exported as a percentage of domestic hardwood lumber production increased by nearly 70 percent between 1990 and 2006 but decreased by 2008 (Table 3). In 2008 exports were second only to the kitchen cabinet industry in the volume of graded hardwood lumber consumed (Hardwood Market Report 2009). The fact that most of the lumber that is exported is kiln-dried means that the value of export sales rivals the value of lumber sold to the kitchen cabinet industry.

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²We treat China and Hong Kong as one market in this study because most of the lumber entering Hong Kong is shipped to mainland China.

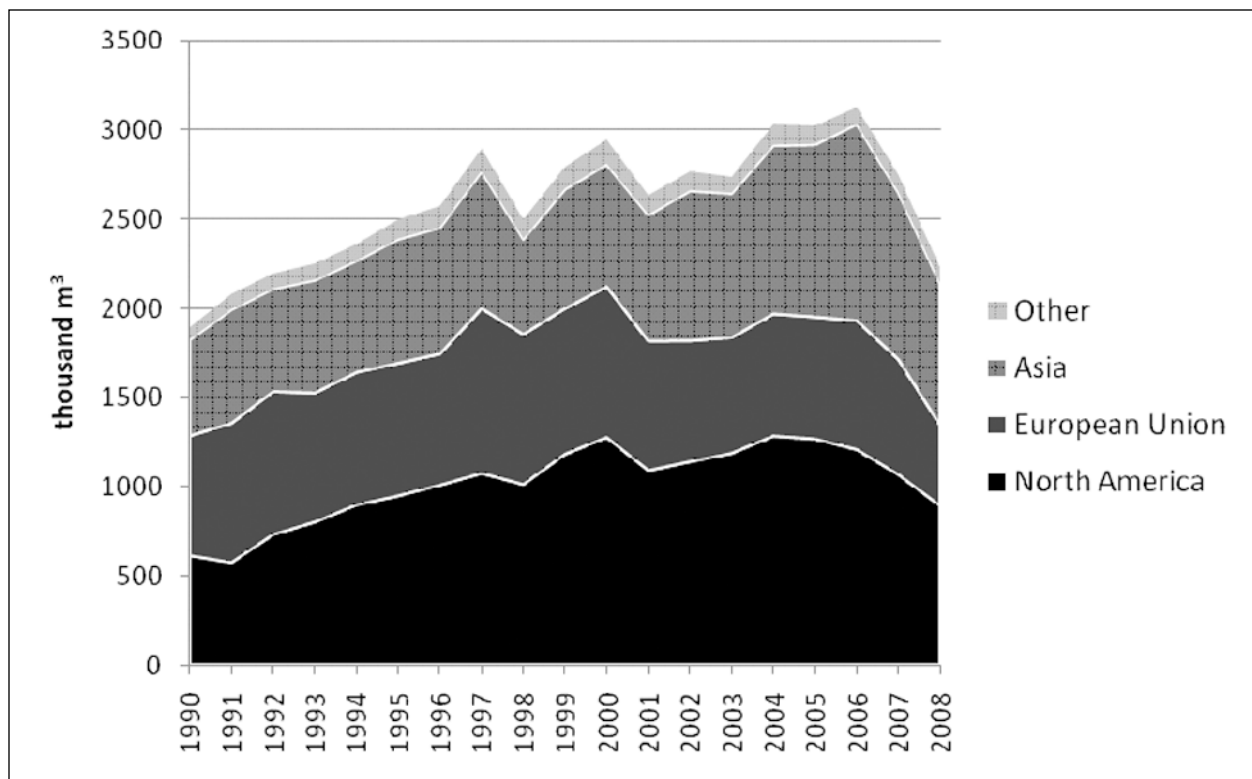


Figure 1.—U.S. hardwood lumber exports to major world regions (USDA FAS 2009).

Because exports have become a major market for hardwood lumber, it is important that we understand changes that have occurred in this market in terms of individual species, trading partners, and the reasons for these changes. It also is useful to understand the importance of exports as a percentage of domestic production for specific species or species groups. The objective of this paper is to review changes in hardwood lumber export by species and country, and to examine changes in exports as a percentage of domestic hardwood lumber production.

METHODS

We analyzed changes in Canadian and U.S. trade, given the mutual importance of this trade relationship. We then examined changes in the species exported and destination of hardwood lumber exports from the United States. These analyses were made for four time intervals (1990 to 1997, 1997 to 2001, 2001 to 2006, and 2006 to 2008) and utilized data developed from the U.S. Department of Agriculture, Foreign Agricultural Service (USDA FAS) (2009) and U.S. Department of Commerce (USDC), Census Bureau (2000 to 2009). These intervals represent periods of increasing and decreasing levels of export activity (Fig. 1). By examining changes between these intervals, we could determine the direction and speed of shifts and changes in lumber exports for the various hardwood species.

RESULTS

U.S.-CANADIAN HARDWOOD TRADE

While Canada has been the largest market for U.S. hardwood lumber exports, it also has been the largest source country for U.S. hardwood lumber imports (Table 4). Between 1990 and 2008, 80 to 90 percent of the lumber imported by the United States from Canada consisted of unspecified species, maple (*Acer* spp.),

Table 1.—Top three export markets for major hardwood species in 1990, 1997, 2001, 2006, and 2008.*

Species	Year	Rank 1		Rank 2		Rank 3	
		Country	%	Country	%	Country	%
Red oak	1990	Canada	41.8	Taiwan	17.5	Japan	10.2
	1997	Canada	54.0	Mexico	9.7	Taiwan	8.2
	2001	Canada	54.9	China/HK	12.1	Mexico	9.0
	2006	Canada	55.4	China/HK	22.1	Mexico	10.1
	2008	Canada	55.1	China/HK	21.1	Mexico	9.3
White oak	1990	Canada	12.9	U.K.	11.0	Spain	10.2
	1997	Spain	18.8	Canada	12.5	Japan	9.5
	2001	Spain	27.5	Canada	14.5	U.K.	8.1
	2006	Spain	16.5	Canada	15.9	China/HK	10.9
	2008	Canada	19.5	China/HK	13.9	Spain	10.9
Yellow-poplar	1990	Japan	40.3	Italy	33.9	U.K.	6.0
	1997	Italy	33.4	Japan	16.6	Mexico	16.3
	2001	China/HK	30.6	Italy	25.8	Mexico	17.6
	2006	China/HK	36.7	Italy	18.9	Mexico	8.6
	2008	China/HK	25.8	Vietnam	23.2	Italy	17.6
Maple	1990	Canada	41.2	Japan	15.1	Taiwan	11.5
	1997	Canada	40.0	Korea	7.4	China/HK	6.8
	2001	Canada	56.8	China/HK	10.9	Mexico	6.4
	2006	Canada	55.4	Mexico	22.1	China/HK	10.1
	2008	Canada	55.3	China/HK	13.3	Mexico	12.4
Red alder	1990	Japan	42.4	Italy	17.5	Taiwan	15.4
	1997	Germany	33.7	Italy	16.1	Taiwan	12.7
	2001	China/HK	34.7	Italy	13.1	Mexico	10.1
	2006	China/HK	42.7	Mexico	16.2	Canada	13.8
	2008	China/HK	41.3	Italy	15.2	Mexico	12.4
Ash	1990	Japan	35.8	Canada	22.0	U.K.	15.5
	1997	Canada	30.3	Japan	18.4	U.K.	10.7
	2001	Canada	27.3	China/HK	15.6	U.K.	14.8
	2006	Canada	23.2	China/HK	19.0	Italy	8.4
	2008	Canada	21.4	China/HK	21.2	U.K.	7.9
Walnut	1990	Japan	21.2	Canada	19.8	Italy	17.8
	1997	Canada	30.3	Italy	19.5	Taiwan	9.9
	2001	Canada	31.4	China/HK	22.4	Italy	11.0
	2006	Canada	36.8	China/HK	13.8	Japan	5.3
	2008	Canada	48.6	China/HK	9.2	Japan	5.8
Cherry	1990	Canada	28.9	Japan	15.6	Taiwan	8.6
	1997	Canada	31.3	Japan	8.5	Italy	7.2
	2001	Canada	34.5	China/HK	14.3	Mexico	7.1
	2006	Canada	29.5	China/HK	20.6	Mexico	9.3
	2008	Canada	31.2	China/HK	20.6	Mexico	9.0
Total	1990	Canada	28.9	Japan	15.6	Taiwan	8.6
	1997	Canada	31.3	Japan	8.5	Italy	7.2
	2001	Canada	34.5	China/HK	14.3	Mexico	7.1
	2006	Canada	29.5	China/HK	20.6	Mexico	9.3
	2008	Canada	31.2	China/HK	20.6	Mexico	9.0

*Source: USDA FAS (2009).

Table 2.—Volume of U.S. hardwood lumber exported by species in 1990, 1997, 2001, 2006, and 2008.*

Species	1990	1997	2001	2006	2008
	thousands of cubic meters				
Red oak	600.1	649.0	548.6	486.5	407.3
White oak	526.0	665.3	537.8	602.7	412.6
Yellow-poplar	107.0	203.8	224.5	388.1	376.8
Maple	97.9	390.8	364.8	404.4	235.7
Red alder	103.0	302.7	187.1	259.2	157.3
Ash	140.3	151.7	130.6	180.1	135.7
Walnut	22.4	19.7	60.4	95.9	76.4
Cherry	47.3	112.3	159.6	164.5	73.9
Other†	244.2	397.9	419.0	541.8	353.7
Total	1,895.8	2,893.3	2,632.4	3,123.1	2,229.4

*Source: USDA FAS (2009).

†Includes birch, hickory, beech, tropical, and unspecified species.

Table 3.—Volume of U.S. hardwood lumber exported as a percentage of domestic lumber production by species in 1990, 1997, 2001, 2006, and 2008.*

Species	1990	1997	2001	2006	2008
	percent				
Red oak	6.7	6.8	6.0	5.6	5.6
White oak	13.1	14.9	12.7	14.1	10.6
Yellow-poplar	3.2	4.5	5.5	9.4	10.0
Maple	3.6	11.7	10.6	11.8	8.5
Red alder	7.7	18.1	11.0	16.3	11.8
Ash	13.4	16.4	16.2	21.3	18.9
Walnut	12.8	16.4	30.7	31.4	30.1
Cherry	5.7	12.0	17.1	15.8	9.7
All species	6.8	9.9	9.5	11.5	9.8

*Developed from USDA FAS (2009), USDC Bureau of the Census (1991 to 1999), USDC Census Bureau (2000 to 2009), and Luppold and Bumgardner (2008). Volume of lumber reported by Census as not specified by kind and as mixed hardwood were allocated to the various reported species by the proportion they were produced relative to one another. Alder was assumed to be two-thirds of western production.

Table 4.—Canadian imports, exports, and re-exports of hardwood lumber to the United States, Western Europe, and other countries.*

Item	1990	1997	2001	2006	2008
	thousands of 1982 U.S. dollars				
Imports from:					
U.S.	149.2	185.5	206.5	199.9	155.5
Europe	0.2	0.4	0.5	1.3	1.6
Other	5.7	5.5	9.2	13.8	15.7
Total	155.1	191.4	216.2	215.0	172.8
Exports to:					
U.S.	36.9	101.2	142.9	137.2	60.4
Europe	82.9	104.8	85.1	57.6	35.0
Other	15.6	38.4	23.7	51.0	42.5
Total	134.4	244.4	251.7	245.8	137.9
Re-exports to:					
U.S.	0.3	0.2	1.7	2.2	1.6
Europe	51.4	13.4	5.7	1.0	0.1
Other	4.2	1.4	0.6	1.0	0.2
Total	55.9	15.0	8.0	4.2	1.9

*Source: Industry Canada (2009).

and birch (*Betula* spp.). Red oak has consistently been the most important species exported from the United States to Canada, but the second most important species has varied over time from white oak in 1990 to maple species, starting in 1994 and continuing through 2008.

Re-exports represent products shipped to one country then reshipped to a third country without having any value added. Nearly one-third of the lumber exported to Canada in 1990 was re-exported to Europe (Table 4) with half of the re-exports being oak species. The reason for these re-exports included long-term relationships between Canadian and European traders and possibly lower shipping costs from the port of Quebec in the early 1990s. By 1997, re-exports to Europe had declined by nearly 75 percent and had all but disappeared by 2008.

If green lumber is shipped to Canada and then dried before being reshipped to another country, it is considered an export and not a re-export. Because of this definition, the volume of U.S. lumber exported to Canada that is dried or remanufactured then reshipped to the United States or other countries is not known. However, it appears that such reshipments do occur because the United States imports lumber of species not commonly found in Canada, including yellow-poplar (*Liriodendron tulipifera* L.), white oak, hickory (*Carya* spp.), and black walnut (*Juglans nigra* L.).

THE OAKS

In 1990, red oak accounted for nearly 32 percent of hardwood exports and 70 percent of this material was shipped to Canada, Taiwan, or Japan (Table 1). The volume of red oak exported in 1990 amounted to 6.7 percent of domestic production of red oak on a volume basis (Table 3). Canadian demand was associated with furniture and cabinet production (Armstrong and others 1993). Taiwan imported red oak to produce furniture and other secondary products for export primarily to the United States, while Japan's demand for U.S. red oak appeared to be for the production of products consumed within that country (USDC, International Trade Administration 2009).

Between 1990 and 1997 red oak exports increased in absolute terms but decreased in relative terms because of increased exports of most other hardwood species (Table 2). While exports to Canada increased, exports to Taiwan and Japan declined by 50 and 70 percent, respectively. The decline in exports to Taiwan was apparently the result of reduced furniture production in this country as exhibited by the decline in furniture imports from Taiwan to the United States (USDC, International Trade Administration 2009). The decline in Japanese demand was in part the result of the prolonged recession in that country, which started during the 1990s and continues in the Japanese housing market today. Mexico was the second largest purchaser of red oak in 1997, and like Taiwan, most of the products manufactured from this material were exported back to the United States. What was largely unnoticed between 1990 and 1997 was the tenfold increase in exports to Hong Kong and China as Taiwan's furniture industry slowly shifted manufacturing to mainland China (USDA FAS 2009).

Between 1997 and 2001 red oak exports declined by 15 percent, which was greater than the 9 percent decrease in total exports during the period. During this time, exports to Taiwan decreased by nearly 70 percent. By contrast, exports to China/Hong Kong increased by 37 percent as the Asian furniture industry continued to migrate from Taiwan to mainland China. However, total exports of red oak to Asia decreased during this period as maple and other closed-grain species displaced red oak in the U.S. furniture market.

By 2006 red oak accounted for only 16 percent of hardwood lumber exports. In 2008 red oak exports were distributed among three major markets, Canada, China/Hong Kong, and Mexico (Table 1).

White oak was the second most important species exported from the United States in 1990 and accounted for 28 percent of the market in that year. As demonstrated in Table 1, the market for white oak is considerably more diffuse than the market for red oak; the top three trading partners account for 34 percent of the total market. While Canada imported a slightly greater volume of white oak than the United Kingdom and Spain, Europe collectively accounted for 36 percent of export markets for this species in 1990 (USDA FAS 2009). Most of the white oak shipped to Europe is used for appearance applications, including furniture and millwork, but the white oak shipped to Spain is used in wine barrel production.

Between 1990 and 1997 white oak exports increased 27 percent but decreased in relative terms because of increased exports of most other hardwood species (Table 2). While exports of white oak to Spain increased by more than 135 percent, exports to other major European markets decreased. Canada was the second most important market for white oak during this period, but overall shipments to this country decreased by 15 percent. Exports to Japan increased by 50 percent between 1990 and 1997 as U.S. white oak was substituted for indigenous white oak, which was becoming more expensive.

Between 1997 and 2001 white oak exports declined by 19 percent because of reduced shipments to nearly every major market other than Spain. Even though exports of white oak to the United Kingdom decreased during this period, this country became the third largest market for this species because Japan's demand declined by 66 percent. The decline in demand from Japan for U.S. white oak coincided with a decline in all lumber imports into this country from all sources other than Canada and China (e-STAT 2009).

Exports of white oak to Spain decreased between 2001 and 2006 while exports to other major European markets and Canada increased. However, the greatest change in the export market for white oak during this period was the 700-percent increase to China/Hong Kong. Exports of white oak to Spain declined by more than 50 percent between 2006 and 2008 while exports to Canada and China/Hong Kong declined by much smaller amounts. As a result of these changes, China became the second most important export market for white oak (Table 1).

While the volume of red oak exported has been considerable, exports of this species as a percentage of domestic production have been relatively low (Table 3). Red oak is the most commonly produced hardwood lumber species in the United States and accounted for approximately 28 percent of domestic production in 2007. By contrast, exports of white oak as a percentage of production have consistently been about twice the level of red oak (Table 3). White oak is the second most produced U.S. hardwood species.

YELLOW-POPLAR

Yellow-poplar exports increased 250 percent between 1990 and 2008 and this species has become the third most important export species on a volume basis (Table 2). In 1990, yellow-poplar exports accounted for only 3.2 percent of domestic production. Since that period the volume of yellow-poplar exported as a percentage of domestic production has tripled and by 2008 exports accounted for more than over 10 percent of domestic production (Table 3).

In 1990, 80 percent of the yellow-poplar exported went to three countries—Japan, Italy, and the United Kingdom (Table 1). Between 1990 and 1997 exports of yellow-poplar to Japan decreased while exports to Italy and Mexico increased. By 2001, China/Hong Kong became the largest market for yellow-poplar, but Italy and Mexico also were major markets for this species. In 2006, nearly 37 percent of the yellow-poplar exported went to China even as exports to Italy and Mexico increased by 50 and 15 percent, respectively. Vietnam became the second most important market for yellow-poplar in 2008, rivaling the volume exported to China/Hong Kong. As in the case of China, the yellow-poplar exported to Vietnam was exported back to the United States and other countries in the form of secondary products such as furniture.

MAPLE

Between 1990 and 2006 maple exports increased by 313 percent but decreased by more than 40 percent between 2006 and 2008 (Table 2).³ The volume of maple exported as a percentage of domestic production also increased between 1990 and 1997 and remained around 11 percent before declining between 2006 and 2008. Canada has traditionally been the largest market for U.S. maple (Table 1) but actually exports more maple to the United States than it imports. In 1990 the second and third most important markets for maple were Japan and Taiwan, respectively. By 1997 Korea became the second most important market for maple, which was used primarily in the production of pianos and other musical instruments that were exported. China and Mexico became important markets for maple in the 21st century as maple became a major furniture species.

RED ALDER

Red alder (*Alnus rubra* Bong.) is a West Coast species which can be finished to resemble more expensive black cherry (*Prunus serotina* Ehrh.) and currently is one of the most exported hardwood species as a percentage of production (Table 3). In 1990, 75 percent of the alder exports went to Japan, Italy, and Taiwan (Table 1). Between 1990 and 1997 alder exports to Japan declined while exports to Germany increased by 1,600 percent. Much of the alder exported to Japan in 1990 was in the form of rough dimension.⁴ Similarly, initial exports to Germany also were rough dimension. In recent years, however, most of the alder has been sold in board form under proprietary grading rules.

China/Hong Kong became the most important market for red alder in the 21st century. As in the case of maple exports, there was a large reduction in alder exports between 2006 and 2008 that primarily occurred because shipments to China/Hong Kong and Mexico declined by 54 and 42 percent, respectively. Again this decline corresponded with a decline in furniture exports to the United States. during this period.

ASH, WALNUT, AND BLACK CHERRY

In 1990 ash (*Fraxinus* spp.) lumber exports represented nearly 14 percent of domestic production (Table 3) and were shipped primarily to Japan, Canada, and the United Kingdom (Table 1). While some of the secondary products produced from ash in Canada may have been shipped to the United States, the products manufactured from this species in the United Kingdom and Japan were primarily for internal use. Between 1990 and 1997 ash exports to Canada increased by nearly 50 percent while exports to Japan and the United

³Export statistics have not consistently and unambiguously separated maple into the hard and soft species groups.

⁴This observation is based on knowledge gained by one of the authors, who toured West Coast sawmills in the 1990s.

Kingdom decreased by 44 and 26 percent, respectively. Although Canada and the United Kingdom remained important markets for ash, China/Hong Kong became the second most important market for this species by 2001.

Ash exports increased by 38 percent between 2001 and 2006 as higher volumes of this species were shipped to several countries, including a 70-percent increase to China/Hong Kong (USDA FAS 2009). In 2006 ash exports represented 21 percent of domestic production of this species. While ash exports to all major markets declined between 2006 and 2008, the proportion of ash shipped compared to domestic production has remained high.

Walnut was the only hardwood species examined in Table 2 that had a decline in lumber exports between 1990 and 1997. This decline was primarily the result of a 50-percent reduction in shipments to Japan (USDA FAS 2009). During this period, apparent domestic demand and price of this species fell, resulting in a 34-percent drop in production. This large decrease in production caused the exported proportion of domestic production to increase (Table 3) even though the total volume of walnut exported decreased (Table 2).

Walnut's fortunes began to change in the 21st century as China/Hong Kong started to import this species. Unlike most other hardwood species, walnut was used to produce products for domestic consumption. By 2001 more than 30 percent of the walnut produced in the United States was exported; Canada and China/Hong Kong accounted for more than half the market. While walnut exports have declined since 2006 primarily because of a 47-percent decrease in shipments to China/Hong Kong, it still remains the most heavily exported U.S. species as a proportion of total production (Table 3).

The export of black cherry lumber nearly quadrupled between 1990 and 2006 but has declined by more than 50 percent between 2006 and 2008. In 1991 nearly 95 percent of the cherry exports went to Canada and Europe. More than 80 percent of the cherry continued to be exported to Canada and Europe in 1997 and 2001 even though China/Hong Kong was the second largest market for this species by 2001. Cherry exports peaked in 2004 and 2005 but have declined 60 percent since then. The greatest decline was the 80-percent decrease in exports to Europe.

DISCUSSION

In the past two decades exports have become a major market for hardwood lumber, but the species mix and countries of destination have changed. In 1990 oak accounted for nearly 60 percent of the lumber exported by U.S. companies. Exports to Europe accounted for 35 percent of the total volume and Canada, Japan, and Taiwan were the three most important individual markets. While exports were an important market for hardwood lumber on a value basis in 1990, they accounted for only 6.8 percent of domestic production. In the case of white oak, ash, and walnut, however, exports represented approximately 13 percent of domestic production.

The volume of hardwood exported peaked in 2006 at 3.1 million cubic meters (1.3 billion board feet). While the oaks remained the most important species exported, they accounted for only 35 percent of the market while the combined exports of maple, yellow-poplar, and red alder accounted for 34 percent of the market. In 2006 exports accounted for 11.5 percent of total domestic production, but exports of walnut and ash amounted to 31 and 21 percent, respectively, of the production of these species.

Exports declined by nearly 30 percent between 2006 and 2008, but cherry, maple, and red alder exports declined by 55, 42, and 39 percent, respectively. By contrast yellow-poplar exports declined by only 3 percent. In 2008 yellow-poplar was the third most exported species and 50 percent of yellow-poplar export went to China and Vietnam.

While Canada has continually been the most important market for U.S. hardwood lumber exports, China/Hong Kong and Mexico have displaced Japan and Taiwan as the second and third most important markets in 2008. Only 20 percent of the lumber exported in 2008 went to Europe as a result of large declines in cherry, maple, and white oak shipments. While exports accounted for nearly 10 percent of domestic hardwood lumber production in 2008, they represented 30 and 19 percent of walnut and ash production, respectively. By contrast, red oak is currently the least exported major hardwood species as a percentage of the volume produced.

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CHARACTERISTICS OF WEST VIRGINIA LOGGERS DURING ECONOMICALLY DIFFICULT TIMES

Ben Spong, Jinxing Wang, and David Summerfield¹

Abstract.—West Virginia licensed loggers have been surveyed over the years regarding their business, operational, and personal characteristics. An update of this study was completed in 2009 to illustrate the impacts of the current economic conditions on the forestry and forest products industries. Additionally, this survey will help demonstrate the preparedness of the logging workforce to embrace emerging biomass markets that have potentially different workforce needs from traditional logging. A paper-based mail survey questionnaire, with an online option, was mailed to 625 licensed loggers; the questionnaire included questions regarding general harvesting, company arrangements, equipment and technology, and demographics of the owner/manager. These data will be summarized and compared to previous West Virginia logger surveys.

INTRODUCTION

As one of the most forested states in the United States, West Virginia has had a long history of land management, logging, and forest products production (Yarnell 1998). Loggers are a key component in the forest product supply chain as they fell trees, move the logs to the landing, load them onto trucks, and deliver them to the primary processing facilities. Current downturns in the forest products market (Buehlmann and others 2009) have decreased the demand for logs and logging services. Without reliable and profitable work, many loggers have moved away from forestry for opportunities in other industries, as illustrated by the drop in the number of West Virginia timber licenses from 1,030 in 2007 to 622 in 2009.

As logging businesses adjust to the current market conditions, those that are still working have downsized and found a different mixture of jobs, such as land clearing and oil and gas site work, or merely parked their equipment to await an industry turnaround. These conditions are very different from those in the mid-1990s, when demand increased based on large, new investments in the processing industry (Luppold and others 1998). In the early part of this decade the industry struggled somewhat (Milauskas and DeGeorge 2003), but the current economic downturn has far exceeded the previous downturn.

Logger studies have been conducted in West Virginia and other eastern states over the years (Luppold and others 1998, Milauskas and Wang 2006). To assess the impacts of current market conditions and continue to develop a better understanding of the demographic, economic, and operational characteristics of the logging population, periodic surveys have been planned and implemented. This project was designed to characterize many of the components of logging businesses in West Virginia. These results are a critical tool for forestry educators, researchers, and industry professionals to understand current conditions and develop programs and opportunities to foster healthy logging businesses.

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METHODS

A current list of licensed West Virginia loggers was obtained from the West Virginia Division of Forestry in April 2009. An initial survey questionnaire was mailed out in the spring of 2009 to 625 licensed loggers. Included with the questionnaire was a cover letter informing the potential participants of their rights regarding human test subjects. The West Virginia University Institutional Review Board and the U.S. Department of Health and Human Services requirements were followed. Potential survey respondents were informed that the survey was voluntary and that the results from any individual would be kept confidential. Postage-paid return envelopes and a sticker with the words “WV Logger and Proud of It” were included with the questionnaire. The sticker was a token of appreciation for spending the time and effort to fill out and return the survey, as suggested in the Tailored Design Method (Dillman 2000). Follow-up postcards were sent to all loggers who did not respond to the survey to attempt to further encourage their participation. The participants were also given the option of filling out and submitting the survey online. A Web-based duplicate survey was designed to provide a fast and simple method for them to provide their information. Participants could choose between the online and the paper survey questionnaire.

The questionnaire contained 51 questions, which were based on the survey used in 2002. Many of the questions concentrated on obtaining detailed information on West Virginia logger business, educational, and operational characteristics in order to better understand the typical logging contractor. Survey responses were collected into digital files for analysis. Summary statistics were calculated for the questions pertaining to general harvesting, company organization, equipment and technology, and owner demographics.

RESULTS

After the first mailing of the survey, 70 responses or 11.2 percent of the surveys were returned. Of these, 22 responses were from licensed loggers who do not own or manage independent logging businesses, but were required to hold logging licenses by the state of West Virginia. These organizations are typically processing facilities, log yards, and other companies that do not actively harvest timber. The remaining 48 responses (for a 7.7-percent response rate) were completed by independent logging business owners—the target survey respondents.

Owners of logging businesses averaged 49.8 years old and had been in business for 18 years (Table 1). Twenty-two percent of the owners were 40 years or younger in age, and more than 70 percent have been in business for over 10 years. The majority graduated from high school (54.3 percent), 17.4 percent have completed some college courses, and 17.4 percent graduated from college (Fig. 1). These businesses had an

Table 1.—Summary statistics for logging firms in West Virginia.

	Median	Mean	Standard deviation	Minimum	Maximum
Owner's age (yr)	50.5	49.8	11.8	24.0	73.0
Years in business	17.5	18.0	12.1	0.0	50.0
Work hours per week	40.0	41.6	9.5	10.0	60.0
Work weeks per year	51.0	49.4	5.5	30.0	55.0
No. of workers in the woods	2.0	2.9	2.3	1.0	12.0
No. of truck drivers	1.0	1.8	1.4	1.0	5.0
Total no. of employees	3.0	4.6	4.8	1.0	23.0
No. of certified loggers	2.0	2.1	1.3	1.0	8.0
No. of crews	1.0	1.3	1.0	1.0	7.0

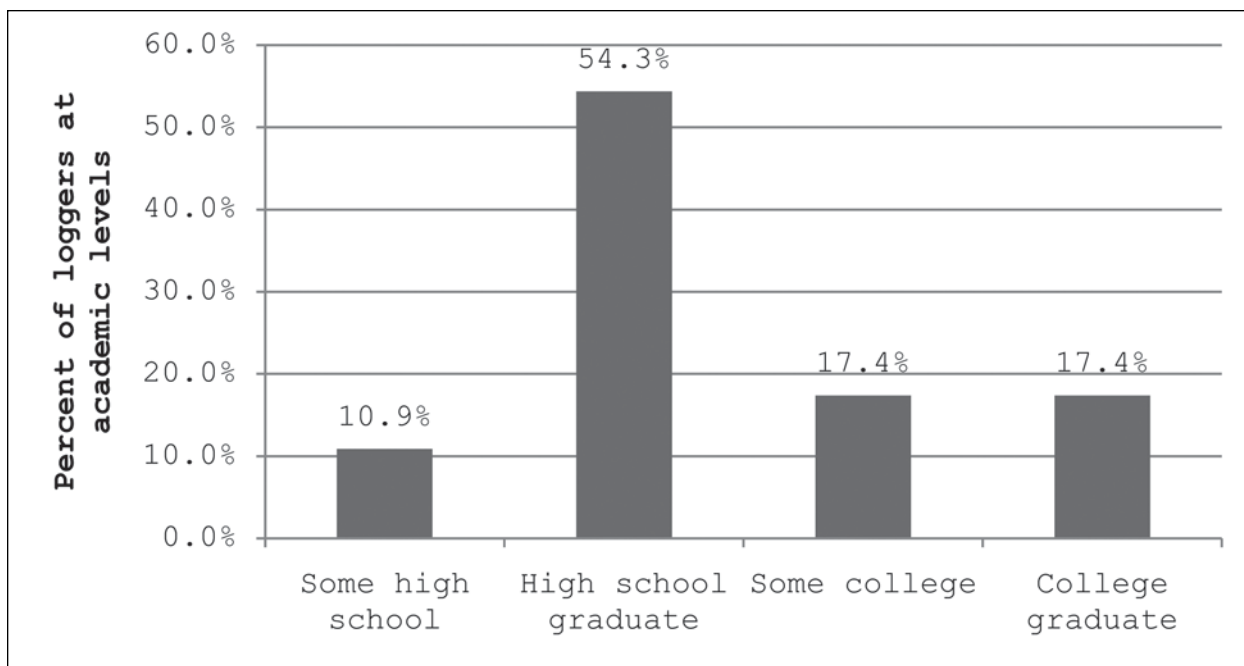


Figure 1.—Academic level of owners of West Virginia logging businesses.

average of 2.9 workers in the woods, 1.8 truck drivers, and 4.6 total employees; they averaged 1.3 crews per respondent. These crews had been trained in CPR/first aid and many (an average of 2.2 per business) were family members. Additionally, an average of 2.1 employees were loggers certified by the state of West Virginia. The West Virginia Forestry Association was the most popular professional organization with 47.9 percent of loggers possessing membership, while 14.6 and 2.1 percent of loggers held memberships with the Mountain Loggers Cooperative Association and the Forest Resources Association, respectively.

Loggers reported that they purchased their own stumpage 34.3 percent of the time (Table 2). The trucking and hauling of forest products was primarily completed with logger-owned and -operated trucks (61.0 percent). Survey responses also reported that loggers attempted to negotiate additional payment for their logging services 72.9 percent of the time in order to cover rising fixed and variable costs of their operations. Twenty percent reported that they were successful all of the time and 77.1 percent were successful some of the time. If logging rates, or the amount loggers were paid per MBF, were increased, half (50.0 percent) responded that they would purchase new equipment (Table 3). Fifty-eight percent would repair existing equipment, and smaller percentages of 27.1 and 12.5, respectively, would hire additional employees or make improvements to shop and office facilities. Logging within the framework of forest certification programs is low; the largest number of loggers (37.5 percent) provide logs to Sustainable Forestry Initiative (SFI) buyers, and fewer (8.3 percent) to Forest Stewardship Council (FSC)-certified facilities.

Unfortunately, increasing logging rates and certification programs have not been enough to save a large number of employees in the logging workforce from losing their jobs. Twenty-five percent of loggers responded that they laid off more than half of their employees (Table 4). Another 10.0 percent reduced labor between 25 and 50 percent, and 4.0 percent cut between 10 and 25 percent. While 27.0 percent of

Table 2.—Response percentage values for timber purchasing and logging rates.

	Percent ^a
Timber purchased by logger	34.3
Timber purchased by others	71.2
Timber hauled by owned trucks	60.1
How often do loggers negotiate logging costs	72.9
Always successfully negotiate	20.0
Sometimes successfully negotiate	77.1
Never successfully negotiate	2.9

^a Percents do not add up to 100% as these results were averages of all responses and each question was asked individually.

Table 3.—Response percentage values for business preferences if logging rates were to increase.

	Percent
Repair existing equipment	58.3
Buy new equipment	50.0
Hire additional employees	27.1
Shop or office improvements	12.5

the loggers cut back 10 percent or less, many included in this category still had job losses. Another difficulty loggers face is the requirement to make workers' compensation payments. West Virginia privatized its workers' compensation insurance system in 2006, creating an opportunity for new programs and potential savings to the logging business. However, this potential savings has not yet arrived for most loggers: only 20.8 percent reported a decrease in their workers' compensation rates (Table 5). Some 18.7 percent of loggers reported increases in their rates. The average workers' compensation rate was 31.7 percent; the median was 31.5 percent (minimum of 0 and maximum of 85 percent).

The profitability of logging businesses during 2009 was reported as very poor or poor by 50 percent of the respondents (Fig. 2). Only 13 percent reported results that would be considered good or excellent. When participants were asked about expected profitability in 2010, the percent of loggers responding good or excellent was 4.0 percent, while the number expecting poor or very poor results was 57.0 percent.

DISCUSSION

The 2009 survey had a higher usable response rate (7.7 percent) than the previous survey of 2002 (Milauskas and Wang 2006, with rates of 4.9 and 6.2 percent, respectively). The number of licensed loggers in West Virginia decreased dramatically from the previous surveys, so while the percentage of loggers replying increased, the raw number of surveys returned was slightly lower. The logging business owners in this survey also are slightly older on average and have been in business for an amount of time similar to those in the previous surveys. The largest number of owners are now in the 51- to 60-year-old group, up from the 41- to 50-year-old group reported in the previous survey, and the proportion of younger loggers remained steady.

Table 4.—Response percentage of logging businesses in downsizing category as a result of current economic conditions.

	Percent
Less than 10 percent	27.1
Between 10 and 25 percent	4.2
Between 25 and 50 percent	10.4
Over 50 percent	25.0

Table 5.—Response percentage values for workers' compensation insurance rates after privatization.

	Percent
Increased significantly	10.4
Increased	8.3
Did not change	20.8
Decreased	20.8
Decreased significantly	0.0

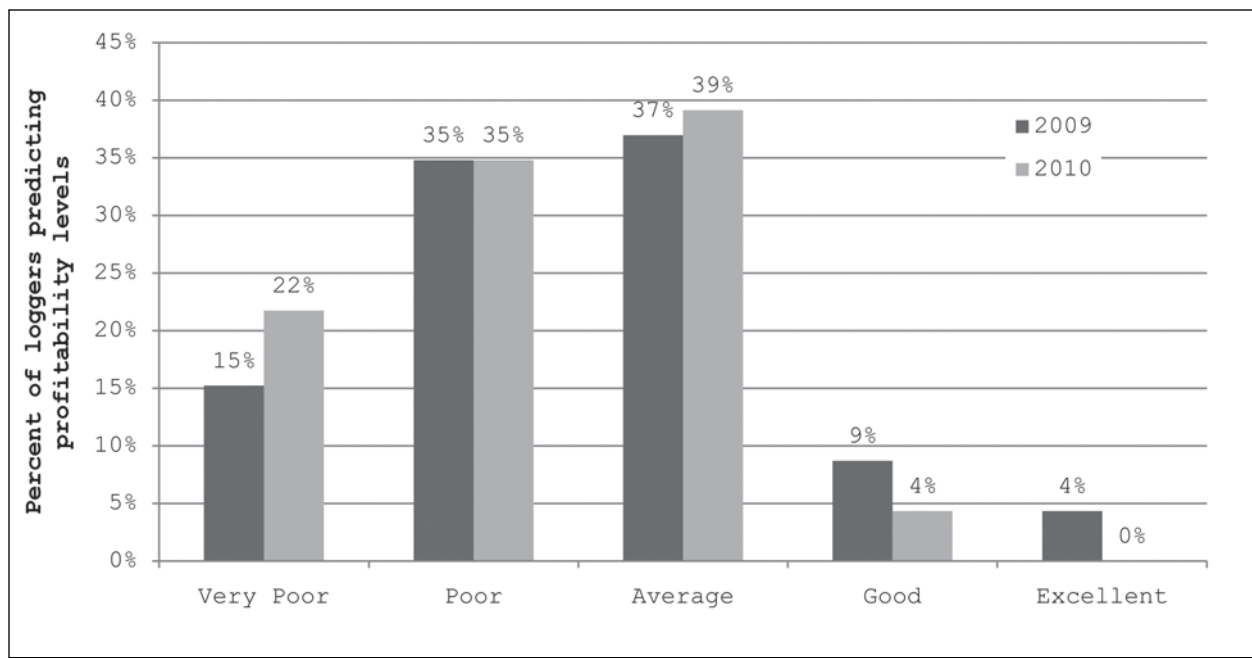


Figure 2.—Profitability in 2009 and expected profitability in 2010 for West Virginia logging businesses.

Comparing the education levels between the two time periods indicates nearly a 10-percent drop in the number of owners with a high school diploma, but a 20.8-percent increase in the number of loggers with some college credit or a college degree. This increase in the educational level of loggers could be attributed to greater efforts to increase higher education enrollment (Chenoweth and Galliher 2004).

The configuration of logging businesses also has changed in the past few years. With a large number of logging businesses and employees no longer actively logging, the drop in the number of workers in the woods (from 4.8 to 2.9) and total employees (from 8.0 to 4.6) between the 2002 and the 2009 surveys is not a surprise. Many loggers run family businesses and the employees that are often first eliminated from the payrolls are not the family members during these difficult financial times. Illustrating this point is a slight increase in family or related employees per logging business, from 1.4 in 2002 to 2.2 in 2009.

Another difference between the two surveys is the decrease in the amount of timber being directly purchased by the loggers. In 2002, loggers indicated that they purchased 41.0 percent of the timber they harvested, as compared to 34.3 percent in 2009. Timber purchases tie up capital needed to maintain operations, so many loggers have increased their reliance on the timber industry and other purchasers to finance the timber purchase. Unfortunately, many of these purchasers also face very difficult markets and financial situations. To complicate the relationship between the logger and timber purchaser, 72.9 percent of loggers reported attempting to negotiate payment rates for their logging services, up from 58 percent in 2002. The success of these negotiations, from the loggers' perspective, has improved as they reported not being successful only 2.9 percent of the time, improving from 15.0 percent from the previous surveys. As the number of loggers keeps decreasing, the demand for their services increases, allowing for some negotiating leverage; however, the scale and true relevance of these additional concessions by the purchaser are not currently known. Trucking activities by logger-owned trucks has remained constant at 61.0 percent.

Half of loggers reported that if the base or negotiated logging rates increased, they would purchase new equipment (54.0 percent in 2002 and 50.0 percent in 2009) and the number of those reporting they would hire additional employees increased from 10 to 27.1 percent between surveys. This large change in the number of loggers looking forward to hiring additional labor is an indicator of the very lean staffing many loggers have maintained during the current economic downturn. This outlook may reflect the start of a shift to increase staffing, contrary to the previous trend of decreasing staffing and replacing workers with advanced equipment when economic conditions improve. While the potential for productivity improvements with new equipment are still important, the reluctance to hire additional employees has decreased.

Loggers have begun to incorporate practices that comply with forest certification programs. While 2009 is the first year these data were collected, the SFI program was much more widely used (29.2 percent more) than the FSC program. Certification programs have benefited some loggers as the demand for certified wood has been slightly better over the last couple of years than for noncertified products. Given the much smaller size of the certified market in comparison to noncertified markets, the benefits of the certification programs to the broad logging population have not yet become ubiquitous.

Healthy logging businesses are a crucial component of the forest products industry as they provide the raw material to the primary and secondary processing facilities. The depressed forest products market has had an impact on the logger population in West Virginia. Results from this study offer valuable current and comparable information to the industry and other parties interested in developing programs that can specifically support and strengthen this portion of the industry. With a strong pool of professional loggers and abundance of raw materials, West Virginia is poised for a recovery in the broader wood products industry.

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ENTREPRENEURIAL ORIENTATION OF EASTERN WHITE PINE PRIMARY PRODUCERS AND SECONDARY MANUFACTURERS: A REGIONAL PHENOMENON?

Delton Alderman¹

Abstract.—Eastern white pine (EWP) and red pine make up nearly 8.5 percent of the total sawtimber volume in the New England, Mid-Atlantic, and Lake States regions. The majority of white pine growing stock is found in the Mid-Atlantic and Lake State regions; however, the center of eastern white pine production and markets is in New England. EWP is produced in both hardwood sawmills and EWP mills using the same equipment and sawing methods as common hardwood species. The quality of Mid-Atlantic and Lake States eastern white pine is arguably equal to or better than that found in New England. Given transportation and transaction costs, why does this competitive advantage exist? In part, this research was pursued to discern whether producers in New England had a greater level of *entrepreneurial orientation*; if so, then these findings (i.e., competitive advantage attributes) could be presented to the industry as a whole. To assess entrepreneurial orientation, a mail survey of EWP primary and secondary manufacturers was conducted in 2004 in the New England, Mid-Atlantic, and Lake State regions. Data were collected from more than 300 respondents and findings indicate that New England manufacturers were not more entrepreneurially oriented than Mid-Atlantic or Lake State producers. Frequency of implementing new technology and/or equipment and the active search for new markets are relevant indicators of entrepreneurship among eastern white pine producers, as opposed to the hypothesized main effects of manufacturing type and region.

INTRODUCTION

Eastern white pine (EWP) and red pine compose nearly 8.5 percent of the total sawtimber volume in the New England, Mid-Atlantic, and Lake States regions; only a minority of this estimate is red pine (USDA Forest Service 2004). EWP is dealt with very much like hardwood species: the same equipment and sawing methods as for common hardwoods are used to produce it in both EWP mills and hardwood sawmills (J. Easterling, pers. comm., May-Aug. 2003), and it is sold primarily on an aesthetic or appearance-grade basis, very similar to hardwood lumber. In New England, both primary manufacturers and secondary producers of EWP lumber and products have long been noted for their business acumen. It is generally demonstrated by their penchant for sustaining and advancing EWP markets, while other regions' manufacturers and producers labor to obtain and/or retain similar markets. The majority of EWP growing stock is found in the Mid-Atlantic and Lake States regions, yet the center of EWP production and markets is New England (Duvall 2004; Alderman and others 2007a, b). Why does this market success exist?

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There is some evidence for a regional preference; for example, research discerned that a preference for New England EWP was found among manufacturers in the New England and Mid-Atlantic regions (Duvall 2004). Other factors also contribute to EWP's success in the New England market. The first is history: New England has been the traditional market for EWP. The region has an extensive record of producing EWP lumber dating to colonial times (Alderman and others 2007a, b). A second factor is the presence of an industry trade association, the Northeastern Lumber Manufacturers' Association (NeLMA), which is the main promotional advocate for EWP. NeLMA's main constituency is in New England and the extreme northern Mid-Atlantic region. Third, NeLMA administers EWP grading, and its certification representatives currently do not serve most of the Mid-Atlantic and Lake States regions due to a lack of market penetration. Fourth, most EWP mills in New England saw EWP exclusively, without significant production of any other species (Duvall 2004). Furthermore, in New England, the retail market consumes nearly half of the total production. While retail markets are present in the Mid-Atlantic and Lake States regions, these markets are not nearly as large as in New England (Alderman and others 2007a, b). One could argue that New England EWP manufacturers have a type of "brand-loyalty" that has developed during the past three centuries. While it is beyond the scope of this paper to examine brand-loyalty, Oliver's (1999) paraphrased definition of brand loyalty may well apply to EWP markets in New England: "A deeply held commitment to repurchase or re-patronize a preferred product/service consistently in the future – resulting in repetitive identical-brand or same brand-set purchasing, in spite of situational influences or marketing efforts that potentially could result in a switching behavior." Anecdotally, and from previous research (Duvall 2004; Alderman and others 2007a, b), EWP originating from New England does possess certain aspects of brand-loyalty. Finally, another explanation for preferring New England EWP is based on perceived wood quality; however, the quality of EWP originating from the Mid-Atlantic or Lake States regions is arguably at least as good as that found in New England (Alderman and others 2007a, b).

In an effort to explain New England's competitive advantage in the face of transportation and transaction costs, this research in part asked whether New England manufacturers were imbued with greater *entrepreneurial orientation* (EO) than those from other regions. If these producers were found to have this attribute to a greater degree, then these attribute findings (competitive advantage attributes) could be presented to the entire industry to assist all regions' producers in the utilization and marketing of the EWP resource. To begin, the foundations of what is meant by "entrepreneur" are explored.

The French verb *entreprendre*, meaning "to undertake," is the origin of the term "entrepreneur." Cantillon (1755) defined "entrepreneur" as a businessman's method of risk-bearing to manage production factors in order to deliver a product or service to the market. In the early 20th century, Schumpeter (1934) described an entrepreneur as a person who is willing and able to transform a new idea or invention into a successful innovation; he concluded that entrepreneurship is a critical driving force for change. More specifically, he posited three typologies of entrepreneurship. The first is *entrepreneurial behavior*, which is the focus of this paper, and includes: 1) Introducing a new product and/or service; 2) new production method(s); 3) new market(s); 4) conquering a new source of raw materials; or 5) reorganizing an industry in a new manner. Schumpeter's other typologies are *entrepreneurial motivation* and *entrepreneurial action*. Twenty-first century definitions emphasize a strong link between entrepreneurship and innovation and differentiate entrepreneurship from typical forms of business management. More concretely, Miller (1983) summarized the characteristics of an entrepreneurial firm as "one that engages in product market innovation, undertakes somewhat risky ventures, and is first to come up with proactive innovations . . ." Entrepreneurship also is

viewed as a critical link between new knowledge and economic growth – as it facilitates knowledge transfer. Entrepreneurs incorporating new technology or equipment may indicate that technology is a business opportunity for those entrepreneurially oriented (Jelinek 1996).

The most widely used operationalization of EO or strategic posture (Table 1) was formulated by Covin and Slevin (1989); they theorized that innovation, proactiveness, and risk-taking behave in concert to “comprise a basic, unidimensional strategic orientation” and therefore should be aggregated when investigating entrepreneurship. The construct of EO is a combination of these three dimensions: *innovativeness* (to revitalize current products or services), being more *proactive* than competitors, and *taking risks* by introducing new and indeterminate (in regards to whether the introduction will be successful) products or services (Miller 1983; Covin and Slevin 1989, 1990, 1991; Zahra 1993; Zahra and Covin 1995). Typically, we evaluate entrepreneurship by measuring the entrepreneurial orientation construct. A further elucidation of EO follows.

ENTREPRENEURIAL ORIENTATION

The EO scale’s theoretical basis is the assumption that entrepreneurial firms differ from other firm types. Initial EO research posits that entrepreneurial firms tend to take more risks and proactively search for new opportunities (Mintzberg 1973, Khandwalla 1977). Similarly, one avenue by which entrepreneurs can differentiate their products or services is new product innovation, as entrepreneurs are noted for their motivation “to take considerable risks” (Miller and Friesen 1982). Covin and Slevin (1988) suggested that the level of entrepreneurship was an aggregate of three subconstructs. They then developed an EO scale measuring: *Innovation* – action to gain a competitive advantage for the firm, *Proactiveness* – competing aggressively with other firms, and *Risk-taking* – taking business-related risks. Firms are considered entrepreneurial if they have high scores for each dimension. Entrepreneurially oriented firms take risks and are proactive by introducing innovative products or services that change market boundaries and market behavior.

SUB-DIMENSIONS OF ENTREPRENEURIAL ORIENTATION

Innovation

Schumpeter (1934) defined “innovation” as the introduction of something new (idea, method, technology), and it is the elemental task of an entrepreneurial firm. Covin and Miles (1999) claimed that innovation was the most critical factor in defining entrepreneurship and Stevenson and Gumpert (1985) noted that it is the “heart of entrepreneurship.” Schumpeter (1934) further defined innovation as the creation and development of new products or processes, and Fagerberg (2004) described it as the first attempt to put an invention or process into practice.

Risk-Taking

McClelland (1960) wrote that entrepreneurship “involves, by definition, taking risks of some kind...” and entrepreneurs by definition are not risk-averse. Entrepreneurs often view business circumstances as being less risky than do non-entrepreneurs (Palich and Bagby 1995). Similarly, Busenitz (1999) argued that entrepreneurs tend to view situations more favorably than do non-entrepreneurs. *Risk-taking* is not the same as recklessness, as entrepreneurs are proficient at risk-assessment. As risk-taking is a salient constituent of entrepreneurial behavior, many entrepreneurs succeed by risk-avoidance and allowing others to assume risk; for instance, “My idea of risk and reward is for me to get the reward and others to take the risks” (cite from Di-Masi 2005).

Proactiveness

Stevenson and Jarillo (1990) conceptualized *Proactiveness* as the “willingness to pursue opportunity.” Likewise, Lumpkin and Dess (2001) viewed proactiveness as an “opportunity-seeking, forward-looking perspective[,]” where new products and/or services are introduced before the competition and where future demand is identified to create change.

OBJECTIVES

The research questions were framed with the following objectives:

1. To assess whether entrepreneurial orientation was region-specific (i.e., stronger in New England than in the Mid-Atlantic or Lake States region) and to assess the EO of primary producers and secondary manufacturers.
2. To determine whether entrepreneurial orientation was affected by gross sales, total production or consumption of EWP, an active search for new markets, frequency of technology or equipment implementation, and number of employees.
3. To develop a parsimonious model of entrepreneurial orientation.

METHODOLOGY

SAMPLE AND DATA COLLECTION

Research data were collected from primary and secondary manufacturers in the New England, Mid-Atlantic, and Lake States regions by a mail survey questionnaire. The EO segment of the questionnaire was designed based on relevant literature and then reviewed by faculty from Virginia Tech, the University of Wisconsin, and U.S. Department of Agriculture, Forest Service personnel. The survey instrument was pretested with 30 industry representatives. EWP manufacturers in three regions of the eastern United States were the population of interest: the Lake States (Illinois, Indiana, Michigan, Minnesota, Ohio, and Wisconsin), the Mid-Atlantic states (Kentucky, Maryland, North Carolina, Pennsylvania, Tennessee, West Virginia, and Virginia), and the New England states (Connecticut, Massachusetts, Maine, New Hampshire, New York, Rhode Island, and Vermont). The sample frame for each region was developed on a state-by-state basis from state and industry directories, national databases, trade association membership lists, and the NeLMA membership directory. Firms then were randomly selected through the use of these resources.

Consistent with previous EO studies, surveys were addressed to high-level executives (for instance, the owner or general manager). Numerous researchers have argued that an owner/general manager is the primary decision-maker and consequently decides on strategic orientation (Lumpkin and Dess 1997, Miller 1983).

After development of the sample frame and pretesting, the initial mailing was sent to 2,741 primary producers and secondary manufacturers in April 2004. A census was attempted for NeLMA members. The first mailing was followed by a reminder postcard approximately 3 weeks later, a second questionnaire was mailed 2 weeks after the reminder postcard was sent, and a second reminder postcard was sent after an additional 3 weeks. In addition to the mail survey, personal interviews were conducted after analysis of the mail survey. These interviews were conducted to validate mail survey results and gather information that may not have been addressed in the questionnaire.

Nonresponse Bias

The assessment of nonresponse bias was patterned on the Armstrong and Overton (1977) methodology. Respondent survey data (30 subjects) were contrasted with 30 nonrespondents contacted by phone (after completion of data collection). Nonrespondents were asked four questions and then asked to estimate their annual EWP consumption. These questions assessed two EWP product attributes and two business service attributes. The following attributes received the highest or lowest overall mean ratings and were selected for analysis: *supports local industry*, *rustic look*, *consistent prices*, and *flexible payment*. It was assumed that these attributes would be robust indicators of bias. Analysis of variance (ANOVA) was performed and there were no significant differences in four of the five contrasts: *rustic look*, *consistent prices*, *flexible payment*, and *production volume*. There was strong evidence for the statistical significance of *supports local industry* as respondents rated this factor higher than did nonrespondents (mean: 5.30 vs. 4.07; $P = 0.01$). *Supports local industry* may be more relevant to smaller mills, and the respondent sample was skewed toward these firms.

Measures – Dependent and Independent Variables

Entrepreneurial orientation was assessed by the following main or fixed effects: 1) Primary or secondary firm, 2) region, 3) gross sales, 4) total production or consumption of EWP, 5) actively searching for new markets, 6) frequency of new technology and/or equipment implementation, and 7) number of employees.

EO was measured using items posited by Covin and Slevin (1988, 1989). EO has been utilized in several research settings and has demonstrated more than acceptable reliability and validity (Becherer and Maurer 1997, Dickson and Weaver 1997, Barringer and Bluedorn 1999). Each questionnaire item asked the respondents to indicate their perception on a seven-point Likert-type scale, ranging from ‘strongly disagree’ to ‘strongly agree’ with scale points on each side and a ‘neither’ point in the center (Brinberg and Axelsson 2002).

Cronbach’s alpha, correlation analysis, principal components analysis (PCA) with varimax rotation, multivariate analysis of variance (MANOVA), and F-tests at $\alpha = 0.05$ were utilized. SPSS® 12.0 (SPSS, Inc., Chicago, IL, 2003) was used for all analysis. PCA was used to detect structure in the relationships between variables. PCA reflects both the common and unique variance of variables, as the algorithm computes a linear combination of variables so that the maximum variance is extracted from the variables.

Cronbach’s alpha (Cronbach 1951) is a statistical method used to measure the internal consistency reliability of item(s) and the extent to which items are interrelated to one another (Churchill 1979). Alpha-levels above 0.70 are considered acceptable for organizational research (Nunnally 1978). Cronbach’s alpha was 0.88 for the aggregated EO constructs. Cronbach’s alphas also were calculated for individual constructs: *Proactiveness* ($\alpha = 0.68$), *Risk-taking* ($\alpha = 0.79$), and *Innovation* ($\alpha = 0.86$). As the results were reliable, all three EO constructs are reported.

PCA also was utilized to assess whether the dimensions of EO represented discrete constructs and to assess reliability. The factor loadings for the EO scale indicated that all three sub-dimensions loaded on the appropriate factors, which accounted for 55 percent of the total variance. *Risk-taking* loaded on factor one (overall factor loading – 0.90; low-loading item = 0.89); *Proactiveness* – factor two (overall factor loading – 0.89; low-loading item = 0.71), and *Innovation* loaded on factor three (overall factor loading – 0.87; low-loading item = 0.81) (Table 1).

Table 1.—Entrepreneurial orientation factor loadings by principal components analysis^a.

EO Constructs	Components		
	1	2	3
Innovation	.336	.348	.875
Proactiveness	.300	.894	.333
Risk-taking	.902	.295	.316

^aVarimax rotation.

Multicollinearity was assessed by using variance inflation factors in accordance with Belsley and others' (1980) recommendations, and the results indicated that the independent variables were not significant and therefore were not confounded with each other. Additionally, the data were mean-centered; that is, the mean was subtracted from all observations. Centering variables may eliminate multicollinearity problems in the data because correlation with other independent variables may be reduced. Interaction terms are subject to multicollinearity and centering is appropriate for variables whose interactions are being modeled (Jaccard and Turrisi 2003).

ANALYSIS AND RESULTS

DEMOGRAPHICS

The adjusted response rate for all regions and both industry segments was 17.0 percent; by region it was as follows: New England – 21.4 percent, the Mid-Atlantic – 12.8 percent, and the Lake States – 17.6 percent). By industry segment, the adjusted response rate was: Primary – 21.7 percent (New England – 48.9 percent, the Mid-Atlantic – 15.8 percent, and the Lake States – 17.2 percent) and Secondary – 12.5 percent (New England – 17.0 percent, the Mid-Atlantic – 14.9 percent, and the Lake States – 7.2 percent). The adjusted response rate was calculated by dividing the number of returned and completed questionnaires by the total number of questionnaires mailed (after subtracting unusable questionnaires).

Analysis indicates that 92 percent of the respondents were high-level executives: Owner (39 percent), president (32 percent), company manager (12 percent), and vice-president (9 percent). Total sales, by category were the following: 49 percent having less than \$1 million in sales, followed by \$1-5 million (27 percent), \$5-15 million (14 percent), more than \$50 million (4 percent), \$15-25 million (3 percent), and \$25-50 million (3 percent). Firm size (employee basis) was dominated by those employing less than 25 people (73 percent), followed by 25-50 (14 percent), 51-100 (7 percent), 101-200 (4 percent), and 200 or more (2 percent).

MODELING ENTREPRENEURIAL ORIENTATION

To test construct relationships in a simultaneous manner, analysis was conducted using a MANOVA general linear model. The main effects were entered individually to assess their potential impact on EO: Primary or Secondary Firm, Region, Gross Sales, Total Production, Number of Employees, Frequency of New Technology or Equipment Implementation, and Actively Searching for New Markets (Table 2). The following main effects were not significant and were not retained in the model: Primary or Secondary Firm ($P = 0.65$) (Wilks' α reported for all MANOVA results), Gross Sales ($P = 0.22$), Total Production ($P = 0.26$), and Number of Employees ($P = 0.22$). Region ($P = 0.05$), Frequency of New Technology or Equipment Implementation ($P < 0.01$), and Actively Searching for New Markets ($P < 0.01$) (Fig. 1) were significant and were retained for the EO model.

Table 2.—Means and standard deviations of entrepreneurial orientation construct items.

Construct Items		N	Min	Max	Mean	Std. Dev.
Management philosophy	(I1)	272	1	7	4.10	1.87
New products/services - A	(I2a)	273	1	7	4.52	1.88
New products/services - B	(I2b)	265	1	7	3.86	1.72
Deal with competitors	(P1a)	261	1	7	4.32	1.63
Management response	(P1b)	265	1	7	3.86	1.44
Deal with competition	(P1c)	270	1	7	3.44	1.80
High- vs. low-risk projects	(R1)	269	1	7	3.79	1.69
Company management	(R2)	267	1	7	3.63	1.66
Respond to competitors	(R3)	266	1	7	4.05	1.71
Constructs						
Innovation		259	1	7	4.13	1.51
Proactiveness		255	1	7	3.86	1.28
Risk-taking		266	1	7	3.83	1.48

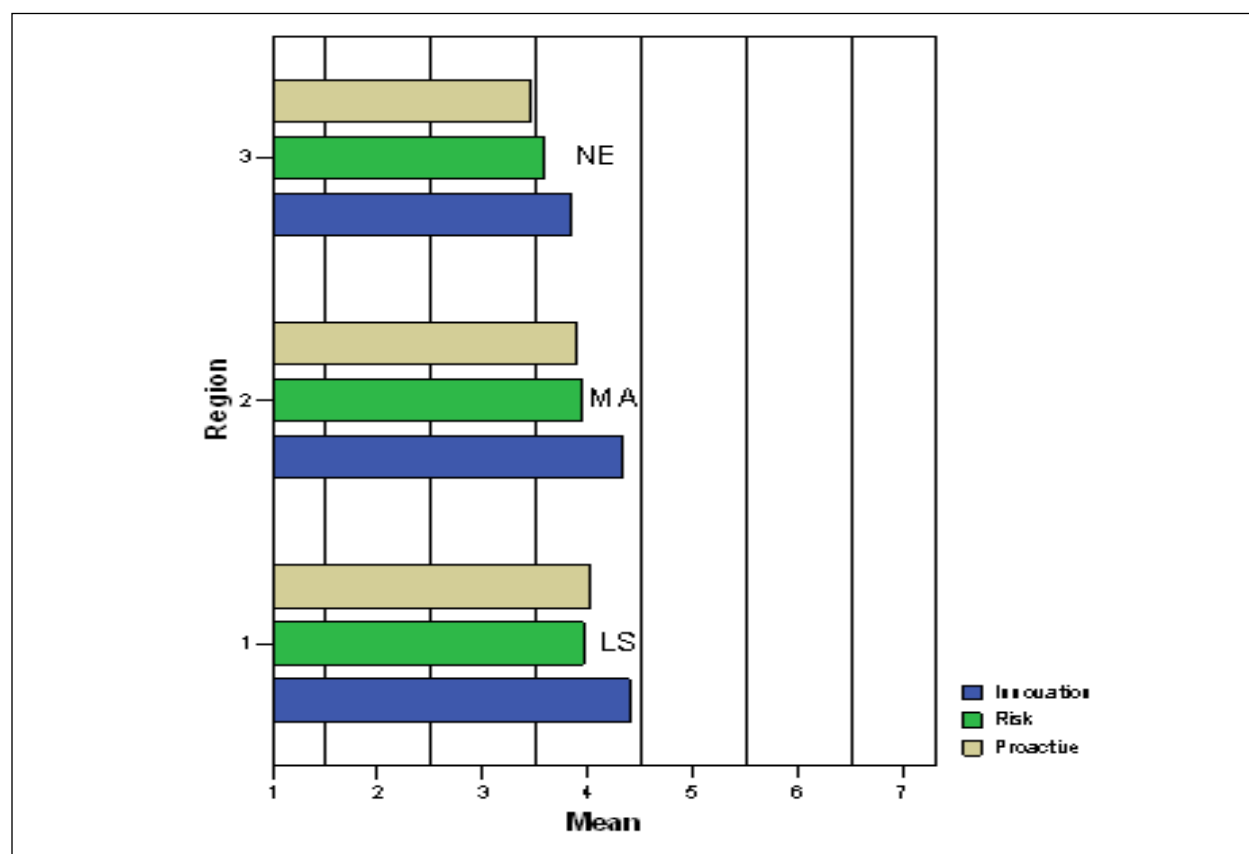


Figure 1.—Entrepreneurial orientation construct means. (NE = New England states, MA = Mid-Atlantic states, and LS = Lake States.)

The overall MANOVA model was significant, with $F_{(3, 209)} = 4.19$ and $P < 0.01$; however, only one main effect was significant in the overall model: *Actively Searching for New Markets* ($P < 0.01$). A between-subjects test was conducted to assess whether differences in the model existed between primary producers and secondary manufacturers. Differences were detected, leading to a corrected model of: *Innovation* ($F_{(3, 29)} = 2.02$ and $P < 0.01$); *Risk-taking* ($F_{(3, 29)} = 1.80$ and $P = 0.01$); and *Proactiveness* ($F_{(3, 29)} = 2.21$ and $P < 0.01$). Next, while most fixed effects were found to be non-significant, a between-subjects analysis was conducted for each main effect to ascertain whether differences existed between primary producers and secondary manufacturers.

In the between-subjects assessment of Region, only *Risk-taking* was significant ($P = 0.02$). In Tukey's HSD multiple comparisons, *Risk-taking* resulted in significant statistical differences between the Mid-Atlantic and New England regions ($P = 0.03$); the Mid-Atlantic region had a greater mean (0.22 vs. -0.39, respectively). This result may indicate that Mid-Atlantic firms are more aggressive in their approach in order to maintain or access new markets.

Regarding Frequency of New Technology/Knowledge Implementation, only *Risk-taking* was significant ($P = 0.04$). Tukey's HSD analysis for Frequency of Implementation resulted in statistical significance for: *Innovation*: Between 1 and 3 years ($P = 0.02$) and 3 and 5 years ($P = 0.01$) and *Proactiveness*: Between 1 and 3 years ($P < 0.01$) and 3 and 5 years ($P < 0.01$). Those who implement new technology or knowledge frequently could be categorized as entrepreneurs based on Jelinek's (1996) proposition that technology is viewed as a business opportunity for entrepreneurs.

For Actively Seeking New Markets, *Proactiveness* and *Risk-taking* were significant ($P = 0.01$ and $P = 0.04$, respectively). Results from Tukey's HSD analysis revealed differences in all three constructs: *Innovation*: Yes vs. No ($P < 0.01$); *Proactiveness*: Yes vs. No ($P < 0.01$); and *Risk-taking*: Yes vs. No ($P < 0.01$). Obviously, in all sub-constructs, the means for Yes were much greater and intuitively it would seem that the Active Search for New Markets is a hallmark of entrepreneurs.

Last, utilizing principal components analysis, all constructs were significant (Table 1): *Innovation* (0.875); *Proactiveness* (0.894); and *Risk-taking* (0.902).

DISCUSSION AND CONCLUSION

A myriad of exogenous and endogenous factors affect business success. In the context of EO posture, the research question was whether New England manufacturers and producers were more entrepreneurially oriented than other regions' firms—they were not found to be so. Second, no evidence was found to support the hypothesis that secondary producers possessed a greater level of EO than did primary manufacturers. Entrepreneurial orientation appears not to be the primary factor for New England firms' success.

In the assessment of interactions, several were not significant. Most notably among those variables were Region and Firm Size (sales or production). Two variables were strong indicators of EO—*Frequency of Implementing New Technology* and *Actively Searching for New Markets*, both yielding significant interactions. These findings indicate that firms searching for new markets and frequently implementing new technologies appear to be more entrepreneurially oriented. This result is not surprising, as anecdotal evidence throughout business history suggests that entrepreneurs are innovators and skilled risk-takers.

New England's competitive advantage appears to be historically based and hence a type of brand loyalty has developed over the centuries. With this historical perspective in mind, what can firms in the Mid-Atlantic and Lake States do to become more competitive? Managers will have to strengthen their entrepreneurial orientation. From model analysis, this approach would include incorporating *innovation*, whether product- or service-based; *taking risks* by introducing new products or services; and adopting a *proactive* posture, which would involve targeting the competition and/or seeking new markets. Mid-Atlantic and Lake States firms' marketing efforts have to address brand loyalty and regional preference. Favorable attitudes assist in creating brand loyalty, but consumers have to be exposed to a product before they are inclined to purchase it.

Many initiatives can be employed to attain a strategic entrepreneurial posture. Developing an EO posture does not result simply from circumstance, but rather from a combination of factors, the most important of which is firm managers' incorporating an entrepreneurial orientation.

LIMITATIONS

As with any research, this study had potential limitations, which can be attributed to several factors, including time and monetary constraints. The most significant limitation was the collection of data from only one individual in each organization. Two problems may ensue. The first possible problem is common method variance, when data are collected from the same source. The second problem could arise when trying to interpret and generalize results; as Podsakoff and Organ (1986) noted, "any defect in that source contaminates both measures, presumably in the same fashion and in the same direction . . .". The data suggest that the respondents may have been skewed toward smaller secondary producers; however, nonresponse bias analysis resulted in only one significant result, suggesting that the results could apply to the larger population. An additional limitation is that this study was a one-shot experimental design rather than longitudinal research.

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PERCEPTIONS OF FIRMS WITHIN A CLUSTER REGARDING THE CLUSTER'S FUNCTION AND SUCCESS: AMISH FURNITURE MANUFACTURING IN OHIO

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Abstract.—The Amish-based furniture manufacturing cluster in and around Holmes County, OH, is home to some 400 shops and has become an important regional driver of demand for hardwood products. The cluster has expanded even as the broader domestic furniture industry has declined. Clustering dynamics are seen as important to the success, but little information has been available to assess the perceptions of firms operating within the cluster. The present study asked manufacturers to rate the importance of several factors to making the Holmes County region a good place to be in business. Common actions taken by these firms, and the information sources most commonly used, also were evaluated. A reputation for high-quality products was rated as critical to the cluster's success, followed by access to suppliers and manufacturing services. It was somewhat common to outsource components from other local shops, to use other vendors for computer numeric control work, and to refer potential customers to other local shops when better equipped for an order. Firms within the cluster were close to their customers, as conversations with customers was rated the most common source of information to learn about furniture industry trends. Word of mouth from other local shops also was somewhat important. These results demonstrated the importance of repeated interaction among the numerous shops, and also showed the importance individual firms placed on maintaining the reputation for high-quality products coming from the cluster. The unique mix of cooperation and competition has resulted in a model of competitiveness in wood furniture manufacturing.

INTRODUCTION

Over the past decade, the United States has declined steeply in the manufacture of wood household furniture. From 1999 to 2008, production employment in the U.S. nonupholstered wood household furniture industry (NAICS 337122) declined by more than 62 percent, or nearly 70,000 employees (Bureau of Labor Statistics n.d.). Hardwood lumber consumption by the U.S. furniture industry has declined from 34 percent of total domestic appearance-grade production (excluding material used to produce pallets and rail ties) in 1999 to just 15 percent in 2008 (Hardwood Market Report 2009). This downturn in U.S. furniture manufacturing has been precipitated largely by increasing imports from low-cost sources such as China, Vietnam, and other locations in southeast Asia (Quesada and Gazo 2006). In 2007, approximately 60 percent of the nonupholstered wood household furniture sold in the United States was imported (Cochran 2008). In the absence of a viable domestic furniture industry, U.S. hardwood lumber demand becomes increasingly reliant

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on housing and remodeling markets (e.g., cabinets, flooring, millwork), particularly for the middle and higher grades (Luppold and Bumgardner 2008, Buehlmann and others 2009).

THE FURNITURE CLUSTER IN HOLMES COUNTY, OHIO

Despite these overall trends, one sector of the domestic furniture industry that has performed well during this period of globalization is the Amish-based manufacturing clusters located in northeastern Ohio and elsewhere. Clusters are defined as geographic concentrations of interconnected companies in a given field (Porter 1998). The Ohio cluster, located in Holmes County and portions of five surrounding counties, has grown as the broader domestic furniture manufacturing industry has contracted (Bumgardner and others 2007). Firms within this cluster seem to be employing many practices associated with a new “paradigm” for the industry (Schuler and Buehlmann 2003). For example, in many retail stores dedicated to Amish-made products, consumers are given options related to wood species, finish, and hardware for their furniture pieces. Specialized supply chains have been developed to facilitate this customization. Stains have been standardized across manufacturers to help enable production of semi-customized products to meet consumer specifications among the numerous small shops operating within the cluster.

Interdependence within a manufacturing cluster arises when few of the resident firms can produce the final product in isolation (Mottiar and Ingle 2007). Within the Holmes County cluster, this notion seems evident with the existence of smaller groupings of firms within the broader cluster. For example, very small shops may reach a point where they do not wish to grow beyond a customer base that can be served by the family members already employed. These shops might choose to work with a larger firm to manufacture specific products for the larger firm’s product line. In this way, they rely on the larger firm for marketing and distribution (Terreri 2008). This situation seems similar to findings by Mottiar and Ingle (2007), who found that two important motivations for small firms operating in a furniture cluster in Ireland were to 1) remain viable, and 2) continue living in the area. Unique linkages among firms are the result. According to Porter (1998), repeated market exchanges are common among proximal companies in clusters.

Economic clusters have been characterized as “critical masses – in one place – of unusual competitive success in particular fields” (Porter 1998, p. 78). The success of the Holmes County cluster can be measured in part by its importance to regional demand for hardwood lumber. A preliminary assessment of this sector in Ohio (Bumgardner and others 2007) found that this cluster, comprising roughly a two-county area, consumed 11 percent of the volume of hardwood lumber produced in Ohio, or 19 percent of the lumber used in appearance-based applications (i.e., excluding pallets and rail ties). The corresponding volume was approximately 43 million board feet, aggregated across more than 400 shops. Thus, the cluster has become an important regional source of hardwood lumber demand.

PROBLEM STATEMENT

Aguilar and others (2009) found that area-specific drivers often were important to the formation of successful forest products-based clusters. While clustering dynamics seemingly have been important to the growth in Amish-based furniture manufacturing, little information has been available concerning firm managers’ actual perceptions of the importance of operating within the cluster. This paper examines these perceptions to help explain what firms have done to be successful.

The present study asked manufacturers to rate the importance of several factors to making the Holmes County region a good place to be in business. In addition, the frequency with which several manufacturing-related actions were taken within the cluster was measured, as were the information sources used to learn of trends in the furniture industry. In addition, smaller firms (one to five employees) were compared to larger firms (six or more employees) on their answers to these questions.² There are possible perceptual differences by firm size based on the fact that, as noted previously, many smaller shops work with larger shops for marketing and distribution of products. Size of firm also could affect the information sources used and the actions commonly undertaken.

METHODS

DATA COLLECTION

A questionnaire was developed with input from several manufacturers and suppliers working in the cluster. While not formally pre-tested, the questionnaire was discussed line-by-line in two separate group meetings with these representatives. The final questionnaire was seven pages long and contained 40 questions. For the present study, the “Holmes County Region” was defined as Holmes County and portions of five surrounding counties in northeastern Ohio, representing an area of approximately 1,500 square miles. A map was provided on the questionnaire to make clear to respondents the geographic definition of the cluster for the purposes of the study.

A packet containing the questionnaire, a cover letter, and postage-paid return envelope was sent in May 2008 to 569 firms. The sampling frame was *The Furniture Book: A Complete Guide to the Furniture Manufacturers and Wholesalers in Ohio's Amish Country* (Anonymous 2005). A reminder postcard was sent to nonrespondents approximately 1 month after the initial mailing. Last, a second packet (containing a duplicate questionnaire, postage-paid return envelope, and updated cover letter) was sent to all nonrespondents approximately 2 weeks after the postcard. All mailings originated from (and were returned to) the Ohio Agricultural Research and Development Center in Wooster, OH.

Of the 196 usable questionnaires returned, 168 were primarily furniture manufacturers while 28 were primarily producers of components or dimension products. Forty packets were undeliverable due to incorrect addresses, and an additional 77 questionnaires were returned from respondents who were no longer in business or were not manufacturers (e.g., suppliers, finishers, distributors, outdoor furniture makers). The result was an adjusted response rate of 43.4 percent. More than 96 percent of respondents indicated that they were the shop owner or co-owner; thus respondents were very familiar with the specifics of their respective operations.

Potential nonresponse bias in the survey was assessed by comparing sample statistics to known population parameters developed from *The Furniture Book*, as discussed by Bumgardner and others (2007). The mean employment and establishment year in the sample was 7.6 and 1994, respectively, which compared favorably with mean values of 7.2 for employees and 1994 for establishment year for the population. Furthermore,

²While formation of a third category (i.e., “mid-sized” firms) might have yielded additional insights, the prevalence of small firms made further division among the larger firms difficult (i.e., only 15 percent of the firms in the sample had 10 or more employees).

Bumgardner and others (2007) reported 2,723 manufacturing employees in the Holmes County cluster; the sample included 1,433 employees or 52.6 percent of that total, which compares favorably with the survey response rate of 43.4 percent. Thus nonresponse bias was assumed not to be a factor when interpreting results.

DATA ANALYSIS

After noting the background characteristics of the sample, aggregated results (means) are presented graphically for each question of interest. For comparisons by firm size, multivariate analysis of variance (MANOVA) was used as a test of differences between small and large firms across multiple dependent variables. For example, when respondents were asked to rate the aspects that make the Holmes County region a good place to be in business, MANOVA was used to determine whether there was an overall difference in ratings between the two groups. When the MANOVA test was significant ($\alpha = 0.10$), one-way analysis of variance (ANOVA) was used to determine the significance of the individual dependent variables, i.e., to determine the variable(s) driving the overall difference.

RESULTS

BACKGROUND CHARACTERISTICS

The distribution of respondents by sales category for 2007 showed that respondents were fairly evenly distributed across categories, at least up to the \$500,000 mark (Table 1). Sixty-seven percent had gross sales of \$500,000 or less and about a quarter had sales of less than \$100,000, suggesting the small nature of most firms. Only slightly more than 5 percent of the sample had sales of \$3 million or more. The average shop operated 42.7 hours per week. For 66 percent of respondents, wood products manufacturing was their sole occupation. More than 80 percent of respondents indicated that they produced household furniture. Additionally, 36 percent produced office furniture and 6 percent produced institutional/contract furniture. Nearly 21 percent produced components (defined as ready to assemble) or dimension (defined as squares, blocks, and edge-glued products) to support manufacturing both within and outside the cluster. Last, 29 percent produced cabinets and 7 percent were manufacturers of millwork products (total sums to more than 100 percent because many firms produced multiple product types). Thus, while often referred to as a furniture manufacturing cluster, the region supports a variety of related wood products production, although furniture is clearly the primary final product. Red oak (*Quercus* spp., mostly *rubra* L.) was the dominant species used, accounting for nearly half of total consumption. To make comparisons based on firm size, 109 of the furniture firms were classified as “small” (1 to 5 employees) and 52 furniture firms were classified as “large” (6 or more employees).³

Table 1.—Breakdown of the sample by sales category for 2007.

Sales category	% of respondents
Less than \$100,000	23.4
\$101,000 - \$250,000	21.4
\$251,000 - \$500,000	22.4
\$501,000 - \$1,000,000	14.6
\$1,000,000 - \$3,000,000	12.5
\$3,000,000 or more	5.7

³Given the specific nature of some of the questions asked (e.g., importance of visiting retail furniture stores as an information source), results are presented for furniture firms only (excluding dimension and components firms).

WHAT MAKES THE HOLMES COUNTY REGION A GOOD PLACE TO MANUFACTURE FURNITURE?

Respondents were asked about the aspects of wood manufacturing in the Holmes County region that made it a good place to be in business (Fig. 1). A reputation for high-quality products stood out clearly as the highest-scoring attribute. Access to suppliers, manufacturing services, and skilled employees also were important, illustrating the importance of the clustering concept.

Some differences were noted based on firm size, as the MANOVA test was significant (Wilks' lambda = 0.844, $p = 0.07$). Based on subsequent ANOVA analyses, perceptions by small and large firms differed for three attributes: reputation for high-quality products, availability of skilled employees, and opportunities to manufacture products jointly with other shops (Table 2). While the last two differences seem reasonable (i.e., larger firms, which are more likely to be hiring outside the family, are more concerned with employee availability, and are in a position to develop products jointly with other shops), it was interesting that there was a difference in perceptions of the importance of quality by firm size. Still, quality was the most important overall attribute for both small and large firms.

HOW OFTEN DO YOU TAKE THE FOLLOWING BUSINESS AND MANUFACTURING ACTIONS?

Respondents were asked to indicate the extent to which they undertook several manufacturing-related actions within the cluster. As shown in Figure 2, the investigated actions were not frequently undertaken, and some

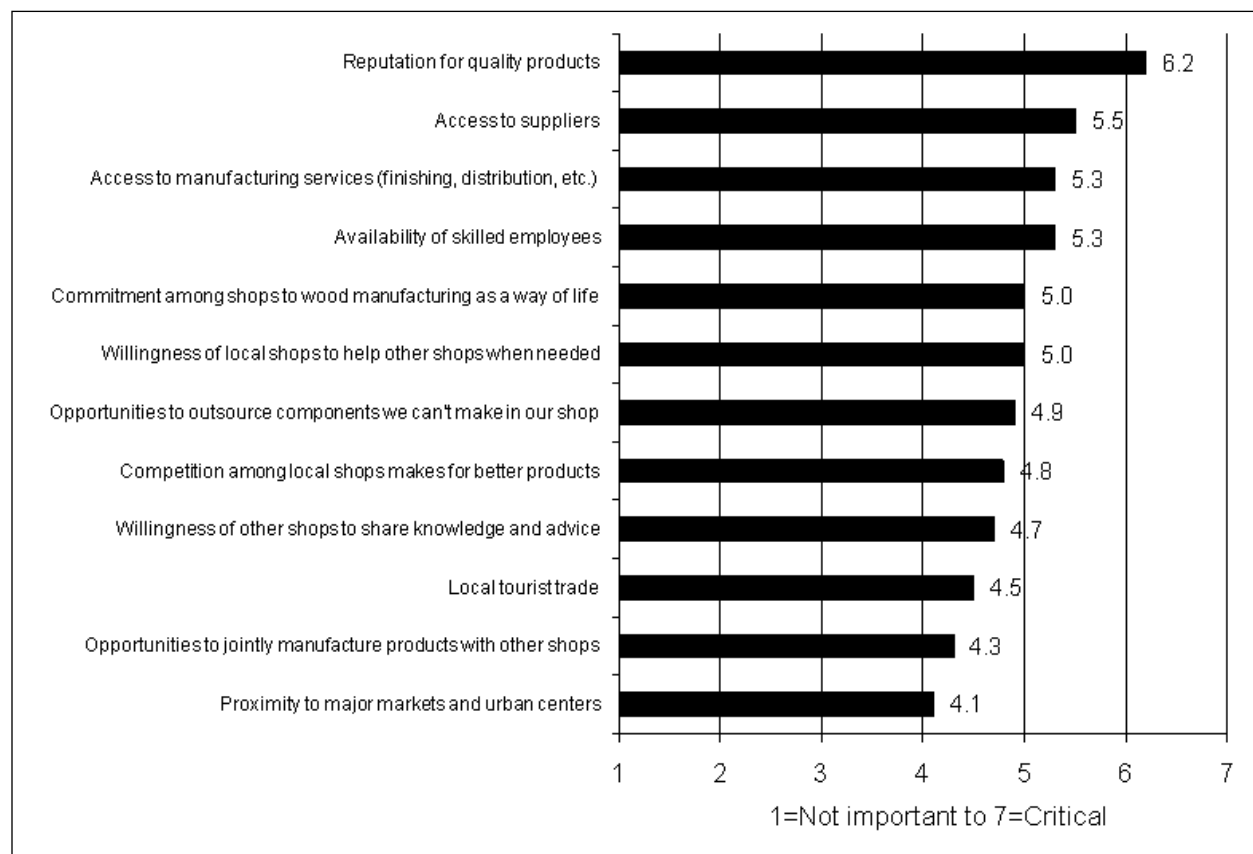


Figure 1.—Responses (means) to the question, “What aspects of wood manufacturing in the Holmes County region make it a good place to be in business?”

Table 2.—Differences in perceptions by small firms (one to five employees) and large firms (six employees or more) regarding the aspects of wood manufacturing in the Holmes County region that make it a good place to be in business.

Attribute	Small firms (mean) ^{a,c}	Large firms (mean) ^{b,c}	F	Sig.
Reputation for quality products	6.0	6.4	3.38	0.07
Availability of skilled employees	4.9	5.8	8.26	<0.01
Opportunities to jointly manufacture products with other shops	4.0	4.9	4.67	0.03

^a n=82.

^b n=42.

^c Scale anchored by: 1 = Not important to 7 = Critical.

were almost never undertaken. The three most common actions were outsourcing components from other shops, using other vendors for computer numeric control (CNC) work, and referring potential customers to other local shops better equipped for an order. The remaining actions were rated quite low, in particular borrowing and/or lending equipment. Thus, while there are numerous examples of cooperation among shops in the cluster, such cooperation does not generally extend to sharing equipment or shop resources. No statistical differences were detected between small firms and large firms among the actions investigated (Wilks' lambda = 0.932, $p = 0.36$).

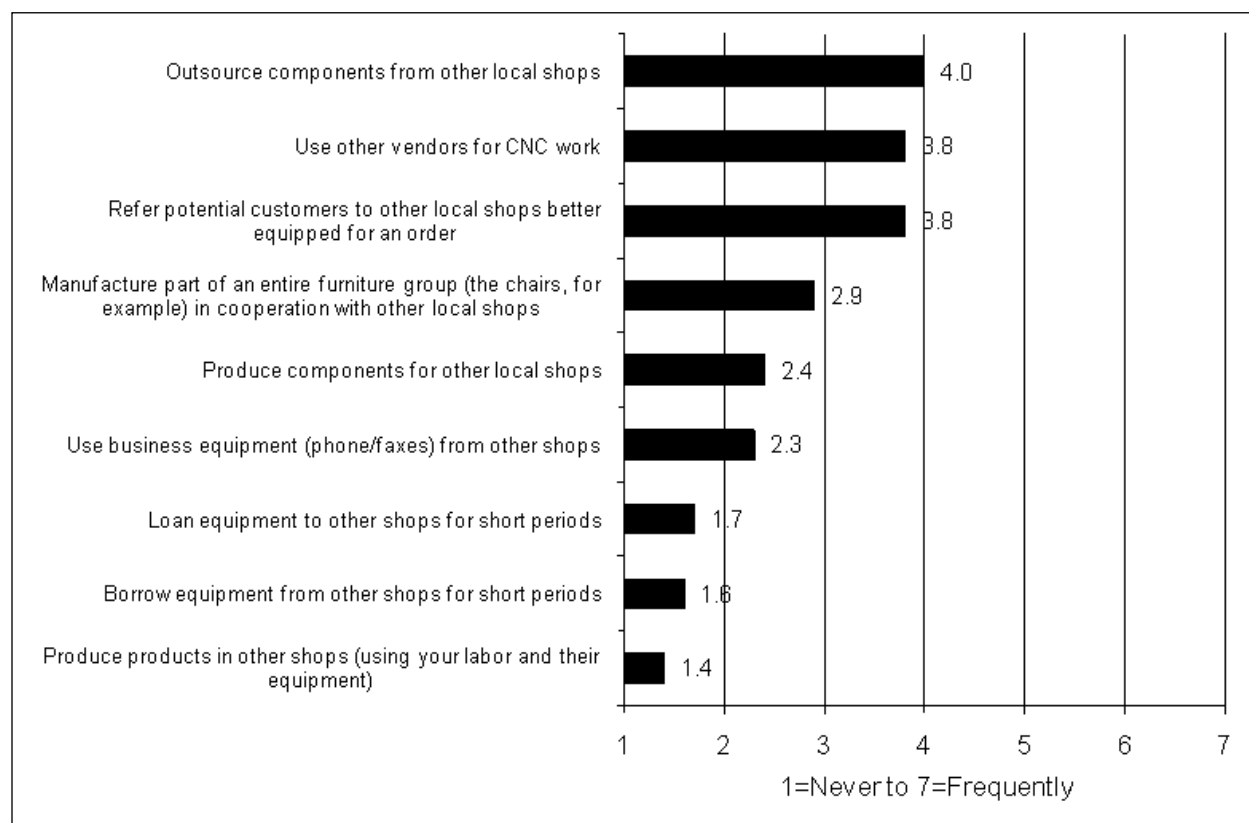


Figure 2.—Responses (means) to the question, “How often do you take the following business and manufacturing actions?”

HOW OFTEN DO YOU USE THE FOLLOWING INFORMATION SOURCES TO LEARN ABOUT TRENDS IN THE FURNITURE INDUSTRY?

Last, respondents were asked about the information sources used to learn about trends in the furniture industry. As shown in Figure 3, conversations with customers and word of mouth from other local shops were the most commonly used information sources. Visits to retail furniture stores and trade magazines also were used to a moderate extent, while workshop attendance, consultants/designers, and newspapers were more seldom used. These results suggest that closeness to customers is a key component of manufacturing within the Holmes County cluster, as is closeness to, and communication with, other local shops. Other studies have also suggested that closeness to customers is a key competitive advantage associated with smaller firms (Gilmore and others 1999). Such close communication allows for development of customized solutions to specific customer needs.

Several statistical differences were detected between small firms and large firms concerning their use of information sources (Wilks' lambda = 0.903, $p = 0.05$), as shown in Table 3. In general, larger firms utilized the information sources to a greater extent than did smaller firms. Specifically, attendance at workshops and use of designers/consultants exhibited the greatest disparities; larger firms also more commonly used trade magazines as information sources.

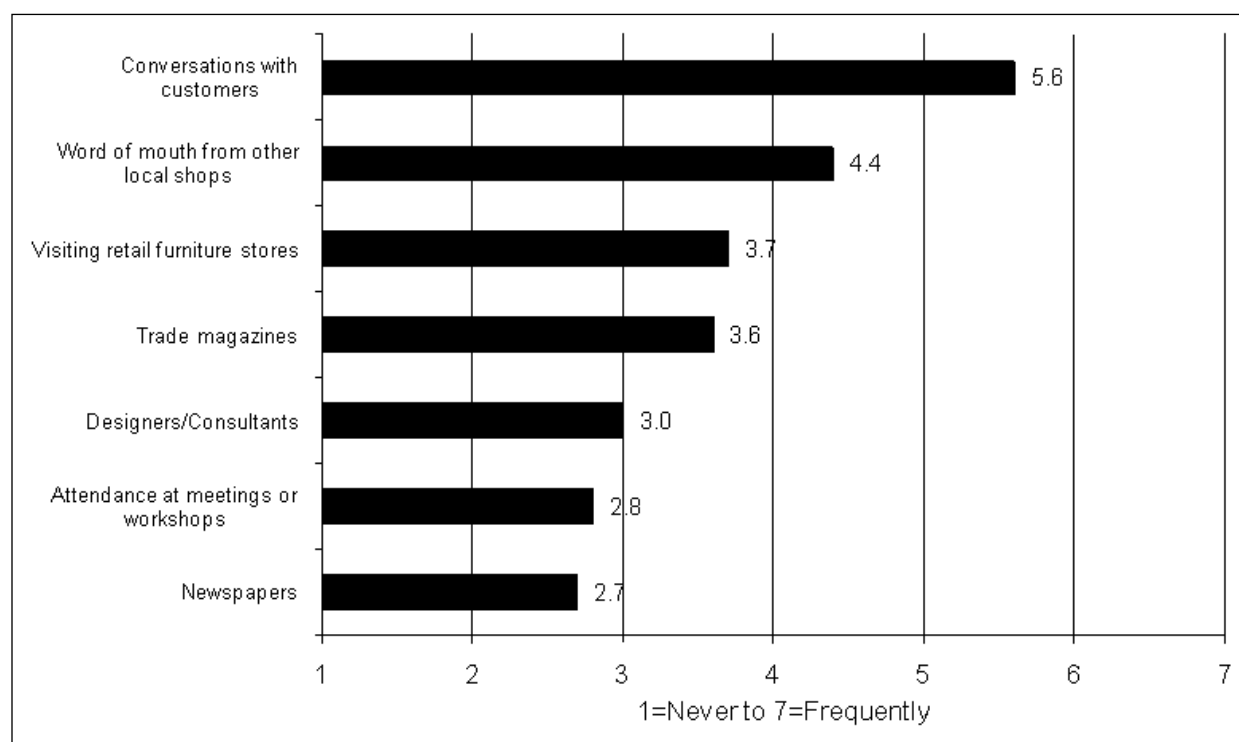


Figure 3.—Responses (means) to the question, “How often do you use the following information sources to learn about trends in the furniture industry?”

Table 3.—Differences in the use of information sources to learn of trends in the furniture industry by small firms (one to five employees) and large firms (six or more employees).

Attribute	Small firms (mean) ^{a,c}	Large firms (mean) ^{b,c}	F	Sig.
Trade magazines	3.3	4.0	4.47	0.04
Designers/Consultants	2.6	3.5	5.64	0.02
Attendance at meetings or workshops	2.6	3.2	5.22	0.02

^a n=94.

^b n=48.

^c Scale anchored by: 1 = Never to 7 = Frequently.

DISCUSSION

The responding firms perceived a reputation for high-quality products as especially critical to the cluster's success. Porter (1998) posits that "peer pressure" is a critical element of the competitive nature of clusters—in the case of Amish furniture manufacturing, firms appear to be especially focused on this aspect and try to maintain a quality level consistent with their peers. It also likely is a function of "embeddedness," which Mottiar and Ingle (2007) found to be an important characteristic of a furniture district in Ireland comprising many small companies; the relationships among firms are not purely economic. In the case of Holmes County, a commitment to quality might be part of the social fabric of the Amish community. Perhaps this characteristic explains why, in Figure 1, *competition among shops makes for better products* is rated somewhat lower than *reputation for quality products*; quality is about more than the effects of peer pressure.

Quality was followed somewhat closely by access to suppliers and manufacturing services, critical elements of the supply chains that are common to clusters. This response likely reflects, in part, the small size and specialized nature of most of the furniture firms. As stated by Porter (1998, p. 80), "A cluster allows each member to benefit as if it had greater scale or as if it had joined with others without sacrificing its flexibility." Availability of skilled employees also was rated highly, especially by the larger firms. The findings also confirmed the notion that larger firms take advantage of opportunities within the cluster to manufacture and market products jointly with other shops (Terrerri 2008), a practice found in other forest products-based clusters (Braden and others 1998). Unlike the findings from Braden and others (1998), however, proximity to major markets was rated relatively low, perhaps suggesting that the Holmes County cluster is less dependent on local markets than are many other forest-based clusters. Many firms reported distant states (e.g., California, Alaska, and Washington) when asked the farthest state in which they had sold their products.

It was somewhat common to outsource components from other local shops, to use other vendors for CNC work, and to refer potential customers to other local shops that would be better equipped for an order. However, most of the investigated actions were not common. Thus, in spite of the unique supply chains resulting from so many firms in close proximity, there appeared to be a degree of independence at the individual firm level when it came to such actions as borrowing or lending business and manufacturing equipment.

Firms within the cluster were close to their customers, as conversations with customers was rated the most common source of information to learn about furniture industry trends. This relationship could be an inherent advantage of the small nature of most Amish-based firms, as Gilmore and others (1999) claim an

advantage of small firms is their ability to stay close to their customers and thus better understand their needs. Word of mouth from other local shops also was somewhat important, demonstrating the importance of repeated interaction among the numerous proximal shops (Porter 1998). This finding also seems consistent with the notion of an “industrial district,” defined by Mottiar and Ingle (2007, p. 668) as “areas dominated by small firms that are geographically concentrated and exhibit strong inter-firm relations.”

While contact with customers and contact with other local shops were important information sources to all firms, larger firms also were more likely than small firms to seek information from sources outside the cluster. These sources included designers and consultants, as well as subscriptions to trade magazines. They also were more likely to travel to workshops, although workshops were not frequently utilized (Fig. 3).

The unique mix of cooperation and competition has resulted in a model of competitiveness in wood furniture manufacturing. The presence of this manufacturing cluster provides evidence that viable furniture production is possible in the United States, which helps sustain regional forest-based economies and provides diversification for hardwood lumber markets beyond those directly related to housing.

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

UNDERSTANDING CHAIN-OF-CUSTODY CERTIFICATION IN THE APPALACHIAN HARDWOOD REGION: PRIMARY MANUFACTURERS' PRACTICES AND PERCEPTIONS

Iris B. Montague¹

Abstract.—Many obstacles may deter hardwood manufacturers from obtaining chain-of-custody certification. Because the hardwood and softwood forest products industries have many differences between them, current certification systems may not fit the unique demographics of the hardwood industry. For this reason, it is important to understand chain-of-custody certification as it relates to hardwood forest products producers. The objectives of this study were to define the characteristics of Appalachian primary hardwood manufacturers and determine their attitudes towards chain-of-custody certification. The study included a questionnaire that was mailed to 1,239 primary hardwood manufacturers in the Appalachian region. The majority of the producers surveyed were small, noncertified manufacturers. Although certification levels were low and many of the producers held negative attitudes towards certification, many of the producers felt that they were environmentally conscious. Results indicate that many companies pursue certification for their customers and to gain some type of market advantage. Although most felt knowledgeable about the certification process, many producers still have little or no knowledge of chain-of-custody certification.

INTRODUCTION

Based on total value, lumber is one of the most important products derived from eastern hardwood forests (Luppold and Bumgardner 2008). In fact, 96 percent of all U.S. hardwood is produced in the eastern United States (U.S. Department of Commerce, Census Bureau 2006). However, the structural changes that have occurred in the domestic hardwood lumber market as well as the increase in globalization and the deterioration of the economy have led to the hardwood industry's decline (Schuler and Buehlman 2003; Luppold and Bumgardner 2006, 2008).

Adding to hardwood producers' concerns of foreign competition and economic decline is the concern of whether to pursue forest product certification. According to "Hardwood Review Weekly," certification is considered one of the four most important challenges currently facing the hardwood industry (Barrett 2008). Forest management certification and chain-of-custody (COC) certification were created with the goal of helping to sustain the forest resource, one of the earth's most valuable natural resources. Forest management certification is the process of verifying that forests are sustainably managed, and COC certification is the process of verifying that wood products come from sustainably managed forests. However, the complexity and expense of existing certification systems effectively exclude sustainable hardwood timber from green markets (Barrett 2008).

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Although the hardwood industry has a long record of sustainability (Barrett 2009), failure to pursue certification may prohibit manufacturers from entering some markets. For example, the U.S. Green Building Council has a policy of allowing only Forest Stewardship Council (FSC)-certified wood to receive the certified wood credit from the Leadership in Energy and Environmental Design (LEED) (Barrett 2009). If manufacturers refuse to certify their products, they are essentially excluding themselves from LEED projects and other markets that accept only certified wood. Because COC certification has been in existence for a couple of decades and appears to be an important issue to the industry, it is worthwhile to determine how to enhance the current systems to ensure they are efficient and beneficial to hardwood manufacturers. To achieve these goals, it is necessary to determine the dynamics of the hardwood industry and the certification movement. This information will inform company managers of the opportunities and challenges of COC certification so that they may make the certification adoption decision that is best suited for their situation. The objectives of this study were to: (1) define the characteristics of Appalachian primary hardwood manufacturers, and (2) determine the attitudes of Appalachian hardwood manufacturers towards COC certification.

STUDY AREA

To understand certification and the U.S. hardwood industry, this study focuses on the primary hardwood manufacturers in the Appalachian region. The Appalachian geographical region was set by the boundaries that the Appalachian Hardwood Manufacturers Association, Inc. (AHMI) uses to define its membership. This region contains 344 counties in New York, Pennsylvania, Ohio, West Virginia, Maryland, Virginia, Kentucky, Tennessee, North Carolina, Alabama, Georgia, and South Carolina (Fig. 1). A list of primary hardwood manufacturers in these regions and their contact information was created using listings from association bulletins, state directories, governmental documents, and other resources. The survey population comprised all primary hardwood solid wood products manufacturers identified in these resources.

METHODS

A mail-based survey was developed using Dillman's Tailored Design Method (Dillman 2000) and methods adapted from Churchill's "Procedures for Developing a Questionnaire" (Churchill 1999). Other published certification-related surveys also were used as templates.

The questionnaire was divided into three sections. Section 1 contained demographic questions about the responding company. Section 2 contained questions about the company's beliefs and attitudes on chain-of-custody certification. Section 3 contained questions relating to the decision process that a manufacturer uses when deciding whether to provide COC certified products.

In October 2008, the initial survey mailing was sent to 1,239 primary hardwood manufacturers. The questionnaires were mailed first class along with a cover letter and a stamped return envelope. The cover letter explained the importance of the survey and respondent participation. Four weeks after the initial survey was mailed, a follow-up questionnaire was mailed to those manufacturers who had not yet responded. After the

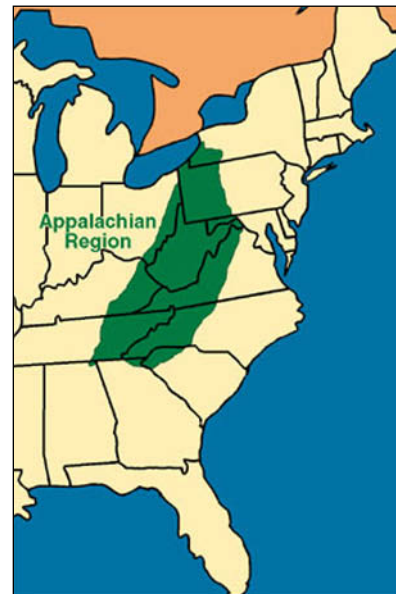


Figure 1.—Appalachian region as defined by the Appalachian Hardwood Manufacturers, Inc. (AHMI)

initial and follow-up letters were mailed, it was necessary in some cases to call manufacturers to determine if addresses were correct or if they were still in business.

Of the 1,239 questionnaires mailed to Appalachian primary hardwood manufacturers, 55 were returned with bad addresses or were otherwise undeliverable. The first question asked the respondents whether they were primary hardwood manufacturers. If the respondents answered “No,” they were thanked for their time and asked to return the questionnaire. One hundred and ninety-nine respondents either responded “No” to that initial question or phoned to say that they were not primary hardwood manufacturers (62 of these responded that they were closed or no longer in business). Of the remaining 985 questionnaires, 192 were returned completed and were deemed usable for the study. The returned questionnaires represent a 19-percent response rate for operating primary processors.

RESULTS

CERTIFICATION AND CERTIFICATION PRACTICES

Out of the 192 usable returned surveys, 189 indicated their COC certification status, of which only 43 (22.8 percent) indicated that they were certified. Of the remaining 77.2 percent, 68 respondents (46.6 percent) indicated they were interested in or actively seeking certification.

Certified respondents were asked to indicate what system was used to certify their products and noncertified respondents were asked to indicate what certification system would be their first choice for certifying their products. Of the 43 manufacturers indicating they were certified, slightly more than half (53.5 percent) had products certified by FSC. However, when it came to noncertified manufacturers, 28.2 percent indicated they would prefer the Sustainable Forestry Initiative (SFI) and 16.1 percent indicated a preference for FSC.

To understand the reasons that manufacturers pursue certification, respondents who were certified or considering certification were asked to indicate which entities most influenced them to consider certification. Respondents indicated that customers, followed by certification organizations, were the most influential. Third-party certifiers, suppliers, and other entities were cited as the least influential entities in respondents’ decision to pursue certification (Table 1).

Respondents were then asked to indicate the level of importance of 10 factors in their decision to pursue certification. The respondents were provided a Likert-style scale and were asked to mark the appropriate

Table 1.—Entities that influence the certification decision of the Appalachian hardwood manufacturers (n = 183).

Entities	Number of firms	Percentage of firms
Customers	65	35.5
Certification organizations	30	16.4
Trade organizations	24	13.1
News organizations	16	8.7
Third-party certifiers	5	2.7
Suppliers	5	2.7
Other	5	2.7

responses (1 = unimportant, 2 = somewhat unimportant, 3 = neither important nor unimportant, 4 = important, 5 = very important). To analyze the decision-making factors of the respondents, the mean and standard deviation of the responses to each statement were calculated. The respondents indicated that all of the listed factors were slightly important (rating above 3 but below 4) in their decision to pursue certification. Cost was the most important factor to the respondents in their decision to pursue certification. This factor was followed by the amount of resources needed to pursue certification and the complexity of pursuing certification. Improving the company's image was the least important factor to the respondent.

It also was important to determine whether any differences existed between certified and noncertified respondents in terms of certification decision-making factors. The mean and standard deviation of the responses for each statement and for each group were calculated. The responses from each group were analyzed using the Wilcoxon test to assess whether differences existed (Table 2). Results indicate that differences exist between the two groups. Of the 10 statements, differences existed between 3 statements at $\alpha = 0.05$ and one at $\alpha = 0.01$. Certified respondents indicated that improving the company's image was slightly important (mean = 3.58; median = 4) in the decisionmaking process, while noncertified respondents felt that it was unimportant (mean = 2.99; median = 3). Results indicate that both groups acknowledge that the amount of resources needed to pursue certification were important; certified respondents felt it was significantly more important (mean = 3.76; median = 3). Noncertified manufacturers ranked the complexity of pursuing certification higher than did certified manufacturers (mean = 3.73; median = 4).

To allow respondents the opportunity to address certification issues that were not included in the survey, three open-ended statements were included. Respondents were asked to list reasons they have pursued or will pursue certification, to list reasons they have not pursued or will not pursue certification, and to express any further comments and concerns that they had with certification.

There were 131 responses to reasons the respondents have pursued or will pursue certification. These responses were grouped into five categories and are shown in Table 3. From these responses it can be seen that many manufacturers (60 responses, 45.8 percent) pursue certification so that they may gain market access or a competitive advantage. These results can be compared with other studies that indicate gaining market access/competitive advantage as a key factor in the decision to pursue certification (Berg and Olszewski 1995, Haener and Luckert 1998, Jensen and others 2003, Vidal and others 2005). The second most frequently cited

Table 2.—Differences between certified and noncertified respondents in terms of certification decisionmaking factors.

Statement	Cert. mean	Non-cert. mean	Sig. diff.
The cost of the certification process	3.53	3.77	
Improving the company's image	3.58	2.99	*
The ability to charge a premium for certified products	3.30	3.15	
The ability to meet the demands of customers	3.74	3.40	
Being able to expand market access	3.79	3.39	
The availability of certified products/raw materials	3.33	3.22	
Gaining a competitive advantage	3.79	3.22	*
The ability to increase or maintain profits	4.00	3.51	
The amount of resources needed to pursue certification	3.76	3.36	**
The complexity of pursuing and maintaining certification	3.28	3.73	*

*significant at $\alpha = .05$, ** significant at $\alpha = .01$

Table 3.—Reasons primary hardwood manufacturers have pursued/will pursue certification (n = 131).

Reasons to Pursue Certification	Number of Responses
To gain competitive advantage/market access	60
To accommodate customers	41
To have a “green image”/good for industry/environment	19
Required to do business/company policy	9
To gain financial benefits	2

reason for pursuing certification is to accommodate customers’ needs/demands (41 responses, 31.3 percent). It is important to note that although most respondents consider themselves to be environmentally conscious, only 19 respondents (14.5 percent) listed “green” image or environmental benefits as their reason for pursuing certification. Nine respondents (6.9 percent) stated that certification is required to do business or that it is company policy. Only two (1.5 percent) respondents cited financial benefits as a reason to pursue certification.

One hundred-sixty reasons were provided by respondents to explain why they have not pursued or will not pursue certification (Table 4). These responses also were grouped into six categories. Of these responses, cost and the inability to receive financial benefits were cited 51 times (31.8 percent). Numerous respondents (42 responses, 26.3 percent) also believed that certification was not necessary and did not have environmental benefits. Twenty-six respondents (16.3 percent) believed that the demand for certified wood did not exist. One respondent stated “Our company will not get certified until our customers demand it.” Other respondents (18 responses, 11.3 percent) indicated certification systems are too complex and time-consuming. The last two reasons for not pursuing certification were the lack of sources (12 responses, 7.5 percent) and supply of certified raw materials and the distrust/dislike of the certification systems (11 responses, 6.9 percent).

CERTIFICATION ATTITUDES

Since the implementation of forest certification, there have been numerous arguments for and against certification. To understand how primary hardwood producers in the Appalachian region feel about chain-of-custody certification, respondents were asked to indicate their level of agreement/disagreement with 18 COC-related questions. A five-point Likert Scale was used to measure the respondents’ level of agreement/disagreement (1 = strongly disagree, 5 = strongly agree). The mean and standard deviation of each response was then calculated to analyze the data.

Table 4.—Reasons primary hardwood manufacturers have not pursued/will not pursue certification (n = 160).

Reasons not to pursue certification	Number of responses
Costly/no financial benefit	51
Not necessary/no environmental benefits	42
No demand	26
Too complex/time consuming	18
No supply	12
Distrust/dislike of systems	11

The results indicated that the majority of the respondents had negative attitudes towards most of the statements; for 15 of the statements the results indicated the respondents disagreed or strongly disagreed. Two of the statements had means indicating the respondents slightly agreed with the statements (a rating above 3 but below 4). For only one statement did the mean response level indicate a positive attitude: “Our company considered themselves to be environmentally conscious,” (mean = 4.37). Respondents disagreed most strongly with the statement “Our company always purchases certified wood” (a mean rating of 1.57). “Our company will pay a premium for certified wood products or raw materials” evoked a very similar set of negative responses (mean = 1.58).

To determine whether attitude differences existed between certified respondents and noncertified respondents, the attitudes of these groups were compared. The responses from each group were analyzed and compared using a Wilcoxon test to assess whether differences existed. Results indicated that differences existed between the two groups (Table 5). The certified respondents tend to be more positive towards the statements than the noncertified respondents. While the certified respondents indicated they slightly to strongly agreed with 10 of the statements, noncertified respondents slightly to strongly agreed with only two of the statements. Of the 18 statements, responses between the two groups are statistically different for all but two statements. No differences were found in the statements, “Our company believes that the chain-of-custody certification process is complicated” and “Our company cannot find an adequate supply of certified wood to justify our becoming certified.” Both certified and noncertified respondents slightly agreed to agreed (rating more than 3 but less than 4) with the first statement and disagreed (rating less than 3 but more than 2) with the second statement.

Although the respondents largely agreed with the statement “Our company is environmentally conscious,” giving it a 4.37 rating out of 5.00, 77.2 percent of the respondents were not certified. When asked to indicate important factors in deciding whether to pursue certification, noncertified respondents were concerned about the complexity and cost of certification and the ability to increase or maintain profits. Certified respondents were concerned about maintaining and increasing profits, gaining a competitive advantage, and expanding market access.

DISCUSSION

Although the majority of primary hardwood manufacturers indicated negative attitudes towards COC certification, 22.8 percent were certified and an additional 36.5 percent were actively pursuing certification or were interested in certification. Given these numbers, it is very likely that the number of certified primary hardwood manufacturers in the Appalachian region may increase in the next few years.

When it comes to choosing a certification system to certify their products, certified manufacturers had a preference for FSC, while noncertified manufacturers preferred SFI. These results are not surprising considering the current statistics on forest certification. According to Bowyer (2008), there are 144.7 million SFI-certified acres of land in Canada and the United States, compared to only 90.3 million FSC-certified acres of land. Because there is a larger supply of SFI-certified timber in North America, it is understandable that more U.S. manufacturers would choose to have their products certified by SFI. Until recently, however, SFI was considered an American-based program and was not accepted in foreign countries. It was not until 2005 that SFI was endorsed by the Programme for the Endorsement of Forest Certification (PEFC) and recognized by other countries (SFI 2008). This delay in global recognition may explain the difference in

certification system preferences between certified and noncertified manufacturers. The certified manufacturers may have based their system choices on the final destination of their products. Products that would remain in the domestic markets could pass with only SFI certification while products that were destined for foreign markets would require FSC certification (PEFC does not endorse FSC). The manufacturers that are currently certified may have chosen FSC when it was the only option for international acceptance and the noncertified manufacturers may prefer SFI now because of the abundance of SFI-certified raw material and the system's recent international acceptance.

Table 5.—Ranking of certification statements by certified and noncertified manufacturers and differences (1=strongly disagree, 5=strongly agree).

Statement	Cert obs. (n=)	Mean/ std. dev.	Non- cert. obs. (n=)	Mean/ std. dev.	Sig. diff.
Our company is environmentally conscious	42	4.62/0.58	144	4.30/0.84	*
Our company is familiar with the certification process	42	4.19/1.06	135	3.33/1.45	**
Our company has purchased certified wood in the past year	40	3.80/1.52	139	2.94/1.40	**
Our company plans to be certified next year	34	3.56/1.52	134	2.87/1.53	**
Our company believes that the COC certification is complicated	42	3.52/1.23	139	2.28/1.36	
Our company believes certification is necessary to be competitive	42	3.43/1.23	138	2.13/1.34	**
Our company believes certification has environmental benefits	42	3.31/1.15	138	2.05/1.15	**
Our company believes certification is necessary	42	3.14/1.33	140	2.01/1.61	**
Our company believes certification has financial benefits	40	2.98/1.37	138	1.98/1.27	**
Our company seeks suppliers of environmentally certified wood products or raw materials	40	2.95/1.43	140	1.92/1.51	**
Our company believes the benefits of certification are worth the costs	42	2.90/1.48	137	1.90/1.27	**
Our company feels pressured by our customers to supply certified wood	42	2.83/1.21	136	1.88/1.24	*
Our company feels pressured by outside groups to produce environmentally certified products	42	2.83/1.17	137	1.83/1.19	**
Our company cannot find an adequate supply of certified wood	37	2.46/1.35	140	1.78/1.19	
Our company only buys certified wood when there is a demand	40	2.40/1.10	136	1.71/1.23	**
Our company believes consumers will pay a premium for certified wood products or raw materials	42	2.36/1.19	137	1.58/1.24	*
Our company always purchases certified wood	40	2.15/1.27	139	1.43/1.04	**
Our company will pay a premium for certified wood products or raw materials	42	2.10/1.16	137	1.41/1.06	**

*significant at $\alpha = .05$, ** significant at $\alpha = .01$

Many factors affect a company's decision to pursue certification. However, it was not surprising that the cost of certification is still the most important factor to the manufacturers in their decision to pursue certification. The fact that these respondents felt certification is not financially or environmentally beneficial and that certification is not necessary may be the reason they chose not to pursue certification. Another important reason given by respondents for not pursuing certification is the lack of demand for certified wood products. Although prior studies have indicated that demand for certified products is weak (Lyke 1996, Ozanne and Vlosky 1996, Ozanne and Smith 1998), more recent studies have indicated that demand from wholesalers, architects, and builders is increasing (Anderson and Hansen 2004, Wickham 2004, Tikina and others 2008, Perera and others 2008).

When the relative importance of these decision-making factors is analyzed, differences in how certified and noncertified manufacturers ranked the factors may explain the difference in certification adoption between these groups. Noncertified respondents were concerned about the complexity and cost of certification and the ability to increase or maintain profits. Certified respondents were more concerned about maintaining and increasing profits, gaining a competitive advantage, and expanding market access. Although the respondents want to receive direct benefits from COC certification (e.g., increased profits), Miles and Colvin (2000) indicate that benefits associated with certification may be indirect. There is strong support for the concept that being a good environmental steward creates a reputational advantage that leads to enhanced marketing, a competitive advantage, and improved financial performance (Miles and Colvin 2000, Vidal and others 2005). Certified respondents may have a better grasp of the possible indirect benefits of certification than do noncertified respondents.

Although this study did not survey respondents on their understanding of the COC process, it is important to note that several respondents stated that they had limited understanding of the process. This response indicates that additional educational marketing of the COC process may be needed, with special emphasis on the indirect benefits of certification, to help increase certification adoption levels. However, respondents also indicated that the complexity of certification was an important factor in their decision to pursue or not pursue COC certification. Current COC certifying bodies must address the unique structure of the hardwood industry and determine a way to decrease the complexity and cost so that firms may be better able to handle the certification process.

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SMALL ENTERPRISES AS IMPORTANT HARDWOOD LUMBER CONSUMERS: EVIDENCE FROM THE CURRENT HOUSING DOWNTURN

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Abstract.—Housing fixtures (such as cabinets, flooring, moldings, and stairways) have become a major market for secondary wood products manufacturers in the United States and a major user of U.S. hardwood lumber. Thus, downturns in housing markets, with associated declines in the demand and price for fixtures, can pose significant challenges to the profitability of these manufacturers and have implications through the supply chain down to raw material suppliers (e.g., hardwood lumber suppliers). The current downturn in the U.S. housing construction industry segment, which began in 2006, provided an opportunity to investigate the attributes associated with successful secondary wood products companies. In particular, it was thought that smaller firms and those producing customized (semi-customized and fully made-to-order) products would be the most likely to have realized increased sales volume during the period in question. It was found that small firms do indeed seem to do better during turbulent times, due in part to the production of fully made-to-order products. Implications for hardwood lumber distribution are discussed.

INTRODUCTION

Seventy percent of dimensional softwood lumber (for example, 2x4's made from spruce, pine, or fir) and structural panels (for example, plywood or oriented strand board) consumed in the United States goes into housing, whether new construction or remodeling of existing homes (Buehlmann and others 2008). Hardwood lumber, although not frequently used for framing, is a raw material widely used to fabricate fixtures such as cabinets, floors, and moldings; therefore, the hardwood lumber industry is sensitive to the housing economy's fortunes. In 1982, hardwood lumber use by the construction and remodeling sector (defined as kitchen cabinets, flooring, millwork, and other miscellaneous building products) accounted for 1.4 billion board feet (bbf) or 28 percent of domestic consumption in appearance-based hardwood lumber products (excluding pallets and containers). This volume had increased to 4.2 bbf by 2002, accounting for more than 52 percent of domestic consumption in appearance-based products (Luppold and Bumgardner 2008). Thus, construction-related markets have grown in volume and relative importance for the domestic secondary wood products industry and for U.S. hardwood lumber producers, especially as the domestic furniture manufacturing industry has moved offshore (Schuler and Buehlmann 2003, Buehlmann and Schuler 2009).

The U.S. housing economy currently is undergoing its worst recession in decades (Buehlmann and others 2008, Dicks 2008). Robust growth earlier in the decade—driven largely by readily available credit, favorable

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demographics, lax lending standards, record low interest rates, a belief by many homebuyers that housing prices would continue to rapidly increase, and speculation by home buyers, builders, and lenders—has turned into a bust, with single-family housing starts at 0.53 million annualized (from a more typical 1.5 to 2.0 million annualized prior to the start of the recession²). Between 2006 and 2007, the value of U.S. single-family housing put in place shrank by 27 percent or \$113 billion (U.S. Census Bureau 2009).

It is often said that smaller firms manufacturing customized products are the most competitive in the current economic environment for secondary wood products. However, much of the evidence is anecdotal. While previous studies have investigated the roles of competitive strategy (Barth 2003) and owner-manager demographic factors (Kangasharju 2000) on firm performance in mature markets, less is known about whether small firms perform better than large firms under such market conditions. This study sought to confirm these suppositions and to determine the product characteristics of successful secondary wood products firms in the current downturn in single-family residential construction (SFRC).

Two hypotheses (or predictions) were tested in this study: 1) Firms having increased sales volume in the declining SFRC market will be smaller as measured by number of employees or by total sales; and 2) Firms having increased sales volume in the declining SFRC market will score higher on customization as measured by proportion of semi-custom products produced and by proportion of made-to-order products produced.

MATERIALS AND METHODS

An email was sent by *Modern Woodworking* magazine in February 2008 to all subscribers with an email address available (approximately 17,000), inviting them to participate in the study. The email included a cover letter explaining the study and a link to the questionnaire posted on the Internet. A reminder notice was sent via email 2 weeks after the initial email. The questionnaire contained 24 questions and was completed by respondents online. As an incentive for participation, respondents were entered in a sweepstakes to win a prize. After 4 weeks, 496 responses had been received. The usable sample of manufacturers was 430 after nonmanufacturing responders such as distributors, service providers, and educational organizations were removed. (The apparently low response rate, as traditionally defined, was partially a result of the heavy presence of nonmanufacturers in the list of email addresses on file). Respondents were active in markets across the United States; 44 percent of firms did regular business in the Midwest, and 21 percent of respondents did regular business in California. More than 69 percent of the respondents held positions in corporate or operating management or were owners, while another 16 percent were in production management.

It was possible to check for nonresponse bias by comparing the sample to known parameters for *Modern Woodworking* subscribers. The proportion of the sample with less than 50 employees was 82 percent, while this proportion was 72 percent for the subscriber population. Also, the sample consisted of 71 percent cabinet and household furniture manufacturing firms while firms associated with these products were 62 percent of the population. Despite some minor differences, the sample seemed to be a reasonable representation of the *Modern Woodworking* population. Given that differences might exist between subscribers and nonsubscribers, however, some caution is warranted in generalizing beyond companies subscribing to the magazine, although the subscriber base is large and likely representative of the broader secondary wood products industry. Another

²Sources: U.S. Bureau of Census – www.census.gov/const/newresales.pdf, and National Association of Realtors – <http://www.realtor.org/research/index.html>.

point to note was that the number of responders facing a declining market was only 15 percent greater than the number realizing increased demand between 2006 and 2007. Thus, there might have been a subset of respondents “motivated” to respond based on their success.

To test the hypotheses, respondents were placed into one of four market/performance groups based on:

1) The proportion of their production volume that was directly associated with the SFRC market in 2007 (that is, used in the building or trimming of new homes, excluding furniture or furnishings); and 2) Whether their sales volume increased or decreased from 2006 to 2007. Primary interest was with those firms who had more than 60 percent of their production volume associated with the declining SFRC market, and whose sales volume increased from 2006 to 2007. Sales growth has been used in other small business studies as the performance measure of choice (Barth 2003). Data were analyzed using Chi-square tests of independence.

RESULTS

The breakdown of the sample along the market and performance dimensions (percentage of business from the SFRC market and sales volume performance) is shown in Table 1. There was a fairly even distribution across the categories representing changes in sales volume from 2006 to 2007. Production volume directly associated with the SFRC market in 2007 was also fairly evenly distributed, but with a small spike in the category where sales volume was 81- to 100-percent associated with the SFRC market. Using Table 1, we classified the firms as follows:

- SFRC ≤ 60 percent of production; sales volume off; Group 1 ($n = 118$)
- SFRC ≤ 60 percent of production; sales volume up; Group 2 ($n = 127$)
- SFRC > 60 percent of production; sales volume off; Group 3 ($n = 112$)
- SFRC > 60 percent of production; sales volume up; Group 4 ($n = 73$)

A group breakdown by main product manufactured is shown in Table 2. As might be expected, the proportion of companies whose main product was kitchen/bath cabinets was substantially higher for those firms primarily in the SFRC market (Groups 3 and 4), while architectural fixtures and contract furniture as a main product were proportionally higher for those firms operating primarily in non-SFRC markets (Groups 1 and 2). The categories were very similar in terms of respondents’ job titles; 64 to 73 percent were corporate/operations managers or were the firm owner. Most of the remainder in each category worked in production management.

Table 1.—Summary results for variables used to categorize firms.

Change in sales volume, 2007 vs. 2006	Percent of respondents	Production volume directly associated with the SFRC market, 2007	Percent of respondents
Sales volume off		—— ≤ 60 percent of production ——	
Much worse (off by 20% or more)	15.6	0%	18.1
Somewhat worse (off by 10%)	11.4	1-20%	16.0
Slightly worse (off by 5%)	10.0	21-40%	10.0
Unchanged	16.5	41-60%	12.8
Sales volume up		—— > 60 percent of production ——	
Slightly better (up by 5%)	13.0	61-80%	16.5
Somewhat better (up by 10%)	19.1	81-100%	26.5
Much better (up by 20% or more)	14.4		

Table 2.—Profile of companies in each group.

Demographic variables	SFRC ≤ 60 percent of production; sales volume off (Group 1)	SFRC ≤ 60 percent of production; sales volume up (Group 2)	SFRC > 60 percent of production; sales volume off (Group 3)	SFRC > 60 percent of production; sales volume up (Group 4)
Main product				
Cabinets	27.1%	32.3%	57.1%	63.0%
Arch. fixtures	12.7%	15.0%	4.5%	4.1%
Molding/millwork	11.9%	11.8%	17.0%	11.0%
Contract furniture	10.2%	12.6%	1.8%	0.0%
Household furniture	31.4%	18.1%	11.6%	17.8%
Other	6.7%	10.2%	8.0%	4.1%
Average number of business regions per firm ^a	2.6	2.6	2.1	1.6

^a Respondents were asked to indicate all regions of the United States where they did regular business: Northeast, Mid-Atlantic, Southeast, South, Southwest, Midwest, Northwest, and California.

Another difference was that firms in Group 3 and especially Group 4 did regular business in fewer geographic regions on average than did firms in Groups 1 and 2 (significant at the $\alpha = .05$ level), which is not surprising given the local nature of housing construction.

Hypothesis 1 stated that firms having increased sales volume in the declining SFRC market would be smaller as measured by number of employees and total sales. As shown in Table 3, Hypothesis 1 was supported. The overall test for number of employees was significant ($p < 0.01$). Group 4 (SFRC > 60 percent of production; sales volume up) had proportionally fewer firms with 20 or more employees than did the other groups. Group 2 (SFRC ≤ 60 percent of production; sales volume up) seemed to be geared the most toward larger firms, with 39 percent of the firms having 20 or more employees.

Similar results were obtained for total sales in 2007, with an overall significant test ($p < 0.01$), though the strength of association was lower than for number of employees (Cramer's $V = 0.16$). Group 4 again had the largest proportion of firms in the lowest sales category (less than \$1 million) and the smallest proportion of firms in the highest sales category (more than \$10 million). Also following a pattern similar to number of employees, Group 2 had proportionally fewer firms in the lowest sales category and proportionally more in

Table 3.—Firm size by SFRC and sales volume group.

Size category	SFRC ≤ 60 percent of production; sales volume off (Group 1)	SFRC ≤ 60 percent of production; sales volume up (Group 2)	SFRC > 60 percent of production; sales volume off (Group 3)	SFRC > 60 percent of production; sales volume up (Group 4)
Number of employees ^a				
1-19	74.6%	61.4%	72.3%	86.3%
20 or more	25.4%	38.6%	27.7%	13.7%
Total sales ^b				
less than \$1 million	61.0%	48.8%	66.1%	78.1%
\$1 - \$10 million	24.6%	34.6%	16.1%	13.7%
more than \$10 million	14.4%	16.5%	17.9%	8.2%

^a Overall χ^2 statistic = 14.9; $p < 0.01$; Cramer's $V = 0.19$.

^b Overall χ^2 statistic = 22.4; $p < 0.01$; Cramer's $V = 0.16$.

the middle sales category (\$1 to \$10 million). There is thus some indication that firms with increased sales volume in 2007 tended to be smaller firms when operating within the declining SFRC market, and larger firms when operating outside of SFRC markets, which tended to be stronger.

Hypothesis 2 stated that firms having increased sales volume in the declining SFRC market will score higher on customization as measured by proportion of semi-custom and made-to-order products. As shown in Table 4, hypothesis 2 was partially supported. There was little evidence of group differences across the “semi-custom” product categories ($p = 0.15$), with a definition that for a given design, customers have a choice of wood species, finish, and hardware. However, there was a significant relationship for “made-to-order” products ($p = 0.04$), which were defined as products for which customers can specify all aspects of the product, including design. Here, Group 4 had proportionally fewer firms in the 0- to 20-percent made-to-order production category and proportionally more in the 61- to 100-percent category, as predicted. Nearly four out of five firms in Group 4 produced made-to-order products on the order of at least 61 percent of their overall product mix.

DISCUSSION

This study found support for the hypotheses that successful secondary wood products firms operating in the declining SFRC housing market were smaller and manufactured more fully customized products than their counterparts. Semi-customization nevertheless was not enough to warrant success as no statistical difference could be established for performance differences between firms manufacturing semi-customized products and firms who did not produce such products. Smaller firms with the capability to manufacture products to order seem especially well positioned to weather the current and future downturns in the housing market and possibly also to compete with larger offshore competitors that have inherent advantages in terms of economies of scale and price. Similar to our findings, other researchers also have found that smaller firms, by working closely with customers (e.g., to produce made-to-order products), seemingly have an important competitive advantage; as stated by Gilmore and others (1999, p. 33), “[our] findings suggest that the SME [small and medium-sized enterprise], by staying close to its customers, will have a keen understanding of their needs.”

With the relative success of smaller businesses and the demise of larger ones (at least in terms of domestic

Table 4.—Characteristics of the overall product type mix by group.

Product type mix	SFRC ≤ 60 percent of production; sales volume off (Group 1)	SFRC ≤ 60 percent of production; sales volume up (Group 2)	SFRC > 60 percent of production; sales volume off (Group 3)	SFRC > 60 percent of production; sales volume up (Group 4)
Semi-custom products ^a				
0 – 20%	14.4%	21.3%	17.0%	26.0%
21 – 60%	28.8%	25.2%	21.4%	13.7%
61 – 100%	56.8%	53.5%	61.6%	60.3%
Made-to-order products ^b				
0 – 20%	18.6%	14.3%	16.1%	4.1%
21 – 60%	23.7%	19.0%	15.2%	16.4%
61 – 100%	57.6%	66.7%	68.8%	79.4%

^a Overall χ^2 statistic = 9.4; $p = 0.15$.

^b Overall χ^2 statistic = 13.0; $p = 0.04$; Cramer's $V = 0.12$.

production) due to more global competition, smaller firms likely will become increasingly important to understanding consumption patterns for hardwood lumber and other raw materials, even though the availability of data to measure consumption by small firms is limited. It also could pose challenges related to distributional efficiency for lumber manufacturers and wholesalers, as they may need to reach proportionally more small customers in the future.

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HARDWOOD LUMBER DISTRIBUTION YARDS: OUTPUT, DEMANDS, AND PERCEPTIONS OF THEIR ROLE

Urs Buehlmann, Omar Espinoza, Robert Smith, and Matthew Bumgardner¹

Abstract.—Efficient and effective supply chains strengthen the entire forest products industry value chain. As the secondary wood products industry has been transformed by the decline of large manufacturers in some industry segments, the industry's supply chain has responded to these new realities. Remaining and new customers tend to be smaller and have unique needs and expectations of their supply chain. Hardwood lumber distributors are adapting to meet these needs by sourcing products in a way that facilitates their expanding role in the new domestic industry. This study assessed the sourcing needs and perceptions of hardwood lumber distributors and depicts parts of the ongoing transformation of the hardwood lumber supply chain.

INTRODUCTION

The primary hardwood lumber industry predominantly serves the secondary wood products industry, including the furniture, cabinet, flooring, millwork, pallet, and railroad tie manufacturing segments. Most of these domestic manufacturers have been affected significantly by increased imports, slowing housing starts, growth in repair and remodeling, and changing buyer habits (Buehlmann and Schuler 2009).

Wood species utilization also has changed during the last decades. For instance, oaks have lost market share. Assuming that the frequency of displaying a species at the High Point, NC, furniture market is positively correlated with consumer preference, large shifts in consumer preferences can be demonstrated. For example, in 1998, oak (*Quercus* spp., red and white combined) represented 22 percent of the wood species displayed in bedroom and dining room furniture, compared to 18 percent for cherry (*Prunus serotina* Ehrh.) and 10 percent for maple (*Acer* spp.). By 2008, however, oak showings had declined to 10 percent of the total while cherry was 12 percent and maple was 9 percent (Appalachian Hardwood Manufacturers, Inc. 2008). Luppold and Bumgardner (2007) discuss the design/fashion trends associated with the shift from red oak to more diffuse-porous species such as cherry and maple in the U.S. marketplace for appearance-based products.

The volume of lumber used by industry segments changed over the last 10 years as well. For example, while the furniture industry was the largest user of appearance-grade lumber in 1999, consuming an estimated 2.6 billion board feet (bbf), the segment's consumption had decreased to 1.2 bbf by 2005 (Hardwood Market Report 2007); by 2009 the furniture industry's lumber consumption had shrunk below 1 bbf (Buehlmann and Schuler 2009). Other segments experienced growth over the period, including flooring, cabinets, railroad ties, and exports (Hardwood Market Report 2007). Table 1 shows lumber use by industry segment.

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Table 1.—Lumber use by market segment 1999-2005 (billion board feet) (Hardwood Market Report 2007).

	1999	2000	2001	2002	2003	2004	2005
Pallets	4.5	4.4	4.0	4.0	4.0	4.0	3.8
Furniture	2.6	2.5	2.2	1.8	1.6	1.3	1.2
Exports	1.2	1.2	1.1	1.2	1.2	1.2	1.3
Millwork	1.3	1.3	1.3	1.3	1.3	1.2	1.3
Cabinets	1.2	1.2	1.3	1.4	1.4	1.5	1.6
Flooring	1.4	1.4	1.5	1.5	1.5	1.6	1.5
Railway ties	0.7	0.6	0.7	0.7	0.8	0.9	0.9

Manufacturers of hardwood products in the United States have lost considerable market share to overseas producers in many categories (Schuler and Buehlmann 2003, Buehlmann and Schuler 2009). One portion of the domestic industry that has fared well against cheaper imported products is the smaller, customized manufacturing sector. These manufacturers often offer customized services and products in niche markets that are insulated from imported products. Examples include small cabinet, millwork, or flooring companies that can design, manufacture, and/or install customized products, often with short lead times. Luppold and Bumgardner (2008) document the increasing relative importance of these construction- and remodeling-related markets to overall hardwood lumber consumption over the past 20 years. These firms frequently are located close to urban centers where new home construction and repair and remodeling markets are substantial. While collectively these producers account for significant use of hardwood lumber, the dispersed and small nature of these firms requires specialized systems of lumber delivery. Among the existing hardwood lumber channels through which sawmills can sell lumber, the distribution yard is emerging as a key supply chain player in the new dynamics of domestic manufacturing.

Distribution yards acquire lumber; add value through services such as sorting, drying, or pre-manufacturing; and are capable of delivering small orders of specialized lumber products to custom manufacturers. While ongoing research efforts track hardwood lumber use by sector in the hardwood industry (e.g., pallets, furniture, cabinets, flooring, and exports), less is known about the emerging role of distribution yards. Distribution yards appear to be playing an increasingly crucial role in strengthening the future competitiveness of hardwood-based manufacturing in the United States by providing an important link between lumber producers and lumber users. U.S.-based distributors also are well positioned to help interested manufacturers source certified materials or products otherwise documented as “green.” This study benchmarked the quantities of hardwood lumber being distributed by distribution yards in the United States, the types of services being demanded of distribution yards by the secondary industry, and the perceptions of distribution yard managers of their emerging and future role in the hardwood supply chain.

MATERIALS AND METHODS

QUESTIONNAIRE

A mail questionnaire using Dillman’s (1978) total design method was chosen as the best way of collecting data due to the large number of questions, some of which required internal company data collection (Lees-Haley 1980). The questionnaire was segmented into four sections: 1) demographics, 2) distributor’s operation questions, 3) customer questions, and 4) open-ended questions regarding the future of the hardwood distribution industry. Where historical trends were queried, data for 2003 and 2007 were collected.

DATA COLLECTION

A sample frame encompassing the hardwood lumber distribution industry in the United States was compiled using the databases of the National Hardwood Lumber Association (NHLA), the North American Wholesale Lumber Association (NAWLA), and the Hardwood Distributors Association (HDA). Use of a variety of associations helped to ensure more complete coverage of the overall distribution industry (Hollander and Wolfe 1999). We mailed 424 questionnaires, 388 in the United States, 32 to Canada, 3 to the United Kingdom, and 1 to Mexico. The foreign outlets' addresses were contained in the associations' directories, indicating that these companies do business in the United States. Twenty-six surveys were returned as undeliverable, and 10 respondents indicated they were not in the hardwood lumber distribution business, bringing the adjusted number of surveys mailed to 388. Seventy-nine usable questionnaires were returned within the survey time frame (August to October 2008), resulting in an adjusted response rate of 20.4 percent. The survey followed traditional surveying methods, with a pretest conducted with four selected companies. A first questionnaire mailing including a letter explaining the survey was mailed after some minor adjustments were made from the insights gained from the pretest. A reminder postcard followed this initial mailing after 2 weeks; another letter and questionnaire mailing printed on colored paper followed 2 weeks later. In a quest to enhance the response rate, a third postcard was mailed, followed by a third letter and questionnaire within 2-week intervals.

DATA ANALYSIS

Thirty-four nonrespondents were contacted after the closing of the survey to obtain answers to check for nonresponse bias (Dillman 1978). Using tests of proportions (Dimension Research 2009), we found no significant difference ($\alpha = 0.05$) between respondents and nonrespondents when asking, “*Is hardwood lumber distribution your only business,*” “*What was your company’s hardwood lumber volume input in 2007,*” and “*What was your company’s hardwood lumber volume input in 2003.*” When we asked nonrespondents and respondents, “*On average, what was the species distribution of the domestic hardwood lumber that you sold in 2003 and 2007 (bdfb basis)?*” (with nonrespondents answering for only five species, namely red and white oak, yellow-poplar [*Liriodendron tulipifera* L.], maple, and cherry), only the answer for cherry was found to be significantly different between respondents and nonrespondents, based on the results from the *t*-test. Likewise, when we asked, “*What percentage of your lumber went to each of the following market segments in 2003 and 2007 (bdfb basis)?*” (with nonrespondents answering only for four markets, namely cabinets, millwork, pallets, and furniture), furniture turned out to be the only significantly ($\alpha = 0.05$) different answer. Thus, no major differences in answers between respondents and nonrespondents were detected.

Fifty-three percent of respondents are single-facility companies and 37 percent are involved solely in the hardwood lumber distribution business. On average, 63 percent of respondents' business stems from hardwood lumber distribution (min. 10 percent, max. 100 percent). Thirty-nine percent of the businesses are located in the South, and 27, 14, and 13 percent are located in the Northeast, the West, and the Midwest, respectively. Seven percent are located outside of the United States.

STUDY LIMITATIONS

Several limitations apply to the results presented in this manuscript. As with most survey research, results are based on responses from a single person from each company surveyed. Though this person typically has roles in ownership or upper management, his or her responses may not necessarily reflect the perspectives of

other decision-makers in the firm. Also, respondents are members of at least one of the associations (NHLA, NAWLA, HDA) from whom addresses were obtained. While the membership base of these associations is large and likely representative of the U.S. hardwood distribution business, caution is warranted in generalizing results discussed in this manuscript.

RESULTS

HARDWOOD LUMBER INPUT AND OUTPUT VOLUME

The average volume of hardwood lumber purchased by U.S. hardwood lumber distributors increased 10 percent from 2003 to 2007, from an average of 12.0 million board feet (mmbf) in 2003 to 13.2 mmbf in 2007 (Table 2). Sales increased by 8 percent from a total volume of 565.7 mmbf in 2003 to 612.1 mmbf in 2007. On average, hardwood lumber distributors sold 12.0 mmbf of hardwood lumber in 2003 and 12.75 mmbf in 2007. However, as can be seen when comparing average company sales (which increased by 6 percent from 2003 to 2007) with the median company sales (which decreased by 6 percent from 2003 to 2007), the increase in sales was due to gains of a few, larger industry participants, while other participants experienced decreasing sales.

SPECIES DISTRIBUTION

When survey participants were asked, “*On average, what was the species distribution of the domestic hardwood lumber that you sold in 2003 and 2007 (bdft basis)?*,” the most noticeable responses were the large reduction in red oak sales (-33 percent), and the large increases in sales of ash (*Fraxinus* spp.; +88 percent), black walnut (*Juglans nigra* L.; +88 percent), hickory (*Carya* spp.; +37 percent), beech (*Fagus grandifolia* Ehrh.; +85 percent), and other species (+41 percent). Figure 1 shows the changes in sales by species from 2003 to 2007 as indicated by respondents.

MARKET DISTRIBUTION

Lumber purchases by market segment also changed substantially over the 5-year period. Sales to the furniture industry segment decreased by 37 percent, while millwork (+14 percent), exports (+29 percent), flooring (+34 percent), retail (+40 percent), and other (+23 percent, encompassing distribution and stair manufacturing) posted solid gains. The largest growth, however, came from railroad ties, which doubled sales over the 5-year period investigated (+103 percent). Only the cabinet markets (-3 percent), besides furniture, lost in importance, a decline that could have been a very early sign of the coming recession (Dicks 2008). Figure 2 shows the percentage of distributors’ lumber going into different market segments in 2003 and 2007 on a board foot-basis.

Table 2.—Average, standard deviation, median, and total lumber volume purchased and sold.

	Lumber Input			Lumber Sales		
	2003 mmbf	2007 mmbf	Change %	2003 mmbf	2007 mmbf	Change %
Average	12.02	13.22	10	12.04	12.75	6
Std. Dev	12.14	11.10	-9	8.77	9.61	10
Median	10.05	10.00	0	10.60	10.00	-6
Total	577.1	634.5	10	565.7	612.1	8

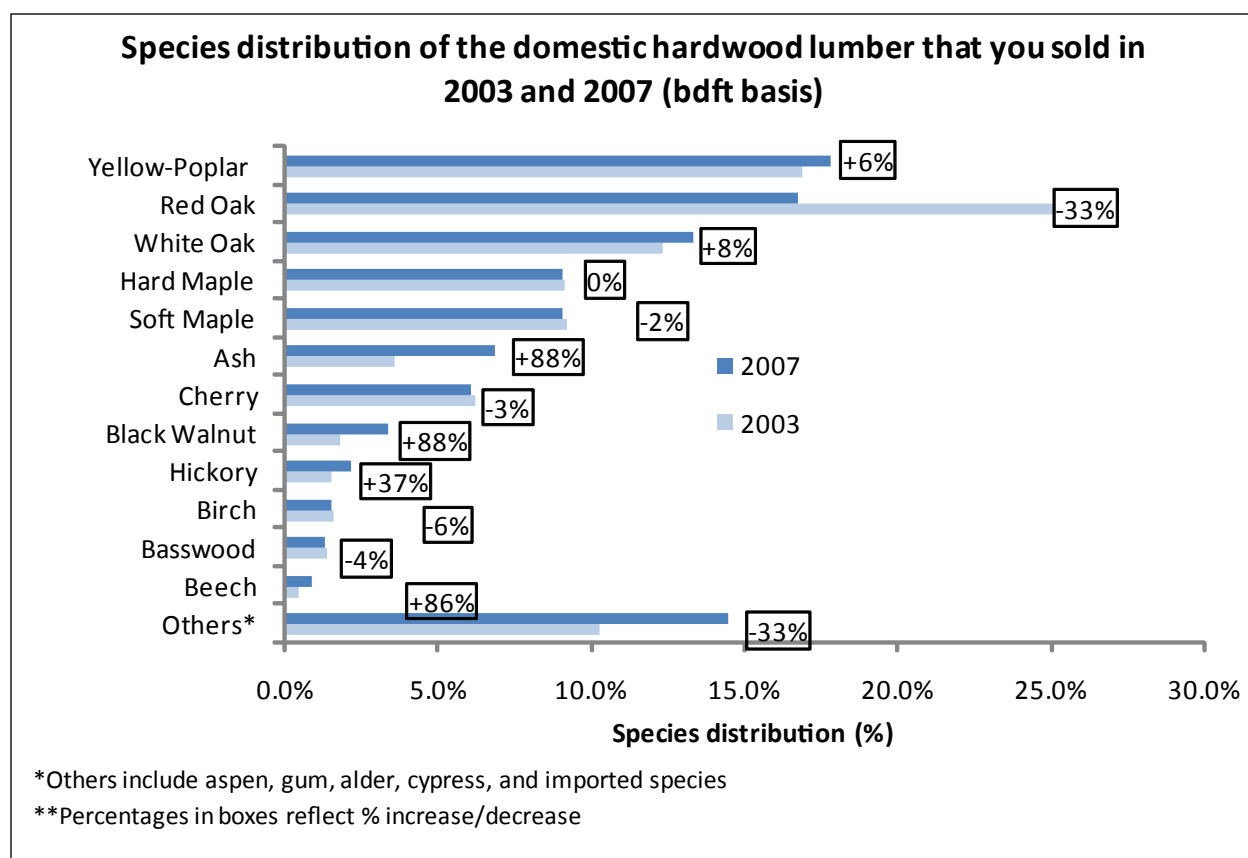


Figure 1.—Changes in sales by species from 2003 to 2007 (board foot-basis).

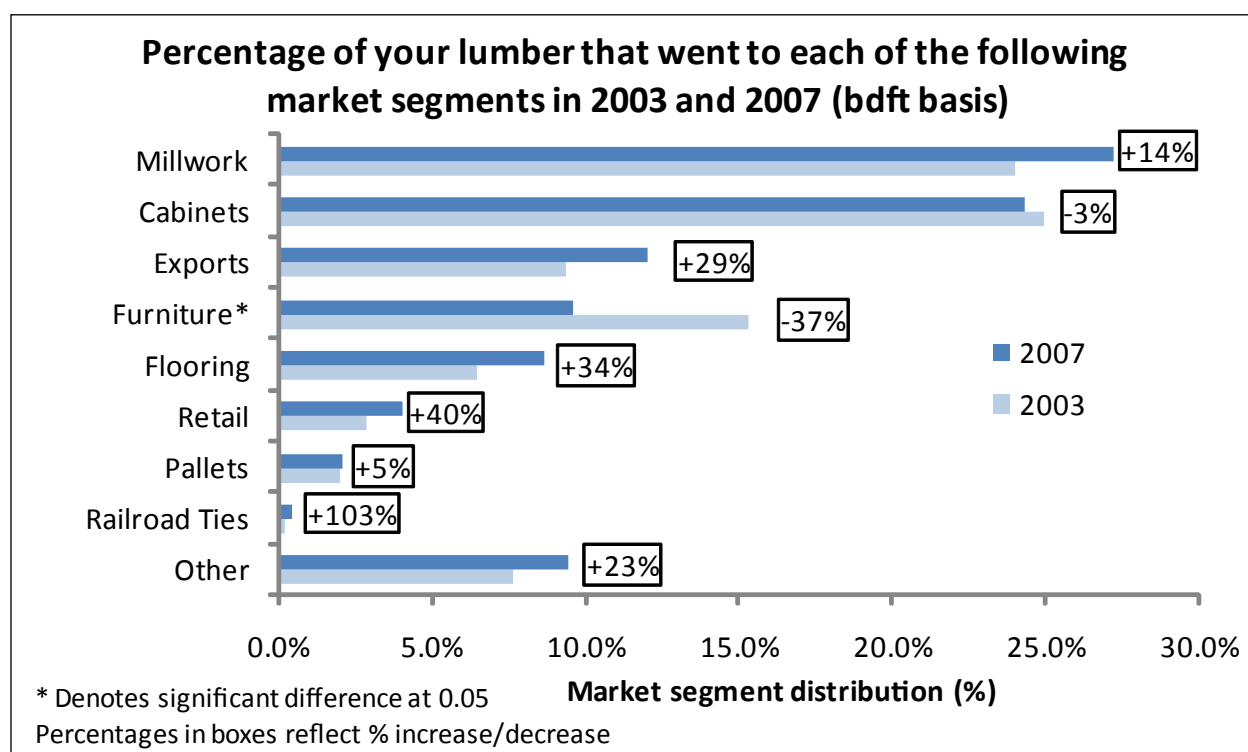


Figure 2.—Percentage of lumber going into different market segments in 2003 and 2007 (board foot-basis).

SIZE OF CUSTOMERS AND ORDERS

Almost half of all respondents to the survey indicated that their current average customer is smaller today than 5 years ago (47 percent). Thirty-nine percent of respondents indicated that their current average customer is about the same size today compared to 5 years ago while 12 percent indicated that the current average customer is larger. Two percent of respondents indicated that they are not sure about the size of their current average customer. A full 75 percent of respondents, however, indicated that average order size (in board feet) decreased over the past 5 years. Thirteen percent responded that average order size stayed about the same and 12 percent indicated that their average order size grew larger during the past 5 years.

While no direct link between decreasing average order size and the smaller size of the average customer exists, it can be hypothesized that both trends—smaller customers and smaller orders—are the consequence of the exodus of manufacturing capacity from the United States (Buehlmann and Schuler 2009) and the competitiveness of smaller shops who directly serve customers with customized products (Schuler and Buehlmann 2003). Both trends also suggest an increasingly important role for lumber distributors in the hardwood supply chain.

FUTURE OF THE HARDWOOD LUMBER DISTRIBUTION BUSINESS

Table 3 shows a summary of responses received to the open-ended question, “*How do you feel the role of the hardwood lumber distributor will change in the next five years?*” Survey administrators categorized responses as follows: “*Wider product range*,” “*Customization*,” “*Smaller orders*,” “*Improved services*,” “*Faster service*,” “*Concentration/consolidations*,” “*Certification*,” “*Customer-orientation*,” “*Diverse Inventory*,” “*Efficiency*,” and “*Flexibility*.”

The largest number of respondents to the questions about changes in the role of hardwood distributors over the next 5 years believes that they will have to offer a wider product range (11 responses). Customization of orders (8 responses), serving smaller orders (7 responses), and providing improved services (6 responses) followed in the frequency ranking

(Table 3). Four responses were obtained for each of the following: the need for faster service, the occurrence of industry concentration/consolidation, and the need for certification of their product. Three responses pertained to the need for more attention to customer-orientation and the importance of building and maintaining a diverse inventory. The increased need for efficiency and flexibility drew two responses apiece.

Table 3.—Number of responses received (in 11 categories) to the open-ended question, “How do you feel the role of the hardwood lumber distributor will change in the next five years?”

Category	Number of responses
Wider product range	11
Customization	8
Smaller orders	7
Improved services	6
Faster service	4
Concentration / consolidation	4
Certification	4
Customer-orientation	3
Diverse inventory	3
Efficiency	2
Flexibility	2

CONCLUSIONS

The average lumber distributor purchased 13.2 mmbf of lumber in 2007, up 10 percent from 2003. Sales of hardwood lumber distributors participating in the survey increased by 8 percent between 2003 and 2007 for an average volume of hardwood lumber sold in 2007 of 12.75 mmbf. Ash, black walnut, beech, hickory,

and some other species such as aspen, gum, alder, cypress, and imported species, saw their sales grow between 2003 and 2007. Red oak sales, conversely, declined substantially, as red oak lost 33 percent in sales from 2003 to 2007. The furniture market declined in importance as a customer to the hardwood lumber distributors, while millwork, exports, flooring, retail, and some other markets grew in importance. The largest increase, however, came from the railroad ties segment.

Almost half of the respondents to this survey indicated that their current average customer is smaller today than 5 years ago, and 75 percent of respondents indicated that the average order size in 2007 was smaller than it was in 2003. These are important trends in the industry and suggest an increasingly important role for the services typically supplied by distributors in the hardwood lumber supply chain. Correspondingly, respondents indicated that, going forward, they will have to offer a wider product range, offer more customized orders, serve smaller orders, and provide improved services.

Future research could address the increasing importance of green building and certified products to the hardwood industry, and specifically the role of distributors in meeting this demand, as well as related policy implications (e.g., the amended Lacey Act of 2009). Ultimately, satisfying customer demand and ensuring sustainability are ways to improve the competitive position of the U.S.-based hardwood industry.

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WILDLIFE

TALES OF A REPATRIATED MEGAHERBIVORE: CHALLENGES AND OPPORTUNITIES IN THE MANAGEMENT OF REINTRODUCED ELK IN APPALACHIA

John J. Cox¹

Abstract.—Reintroductions of large mammals are challenging and often controversial endeavors. Elk (*Cervus elaphus canadensis*) were successfully reintroduced to southeastern Kentucky beginning in 1997. Since then, elk have exhibited a decade-long irruptive growth pattern in the absence of mortality factors that commonly limit its abundance in more western portions of North America. As a result, elk impacts on local vegetation in this area are increasingly apparent and concomitant with population growth of the species, particularly on or near reclaimed surface mines. Despite 8 continuous years of hunting, state wildlife officials estimate the current elk population has grown to 10,000-11,000. This paper examines some of the current and future challenges that local wildlife managers and land stewards increasingly face in managing elk numbers within social and ecological carrying capacities in an area with little public land.

INTRODUCTION

The early-mid Pleistocene epoch was characterized by the evolution of an abundant diversity of large mammalian fauna (Kurtén and Anderson 1980). Fifteen Pleistocene predators larger than 15 kilograms hunted the North American landscape, and their prey consisted of approximately 56 herbivores larger than 30 kilograms (Van Valkenburgh and Hertel 1993). Twenty-nine of these herbivores exceeded 300 kilograms and are often referred to as “megaherbivores” (Van Valkenburgh and Hertel 1993). These long since disappeared giants had profound influences on the continent’s ecosystems during this period, yet the majority of plants they coevolved with (e.g., Kentucky coffeetree, *Gymnocladus dioica*) have persisted into modern times.

Rapid ecosystem change (Pielou 1991) and over-exploitation by humans (Martin 1984) during the late Pleistocene are thought to have caused the extinction of many habitat specialists, including many megaherbivores. Evolutionarily older, less specialized habitat generalists prevailed to colonize vacated niches in the landscape (Guthrie 1984). Archaeological data suggest that at least 18 species of large mammals in the Appalachians were extirpated during the Pleistocene and the range of dozens more was reduced (Guilday 1984). Geist (1992) refers to the remaining biota as “species-poor megafauna based on Siberian generalists and Rancholabrean survivors...poorly adapted to North America and to one another.”

Global reductions in megaherbivore diversity and abundance have especially characterized the past two millennia. Today, there remain 11 North American herbivores larger than 30 kilograms and only three that can exceed 300 kilograms (elk, *Cervus elaphus*; bison, *Bison bison*; and moose, *Alces alces*). “Ubiquitous and abundant to the point of dominating mammalian biomass over most of the globe for millions of years, megaherbivores have been so systematically persecuted that they have become almost irrelevant to today’s

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ecosystem and conservation concerns, except in dwindling portions of Africa and Asia” (Terborgh and others 1999). Despite this gloomy prognostication concerning these faunal giants, we must not ignore the realized nor potentially undiscovered influences remaining megaherbivores can exert on the landscape and ecosystem processes.

A “keystone species” has been defined as having a disproportionately great ecological effect on ecosystems relative to their biomass (Power and others 1996). A number of apex predators have been observed to exert keystone influences on ecosystems through top-down regulatory effects on species at lower trophic levels (*sensu* Smith and others 2003). Megaherbivores have been suggested as keystone species because they act as habitat modifiers through grazing, trampling, wallowing, and uprooting of existing vegetation. Ungulates are important modifiers of ecosystem structure and function because they can trigger trophic cascades (McNaughton 1979, Mattson 1997), increase spatial heterogeneity, accelerate successional processes (Hobbs 1996), and influence nutrient cycling and primary productivity (Augustine and McNaughton 1998). For example, elephants have converted entire woodlands to grasslands, which in turn become susceptible to invasion by ungulate grazers and fire; both agents suppress woody growth, thus maintaining grassland regimes (Laws 1970). In pre-European North America, herds of migratory bison numbering in the hundreds of thousands may have exerted similar keystone influences on the ecosystem (Knapp and others 1999). Through acts of grazing, trampling, and wallowing, bison probably maintained existing, and perhaps often created, large tracts of prairie. In more closed-canopy systems, bison activity may have been responsible for creating natural glades (Collins and Uno 1987).

Historically, herbivores were spatially and temporally dynamic modifiers of the bluegrass savanna-woodland of Kentucky (Wharton and Barbour 1991) and other areas within the Central Hardwoods Region. Species that persist precariously in only a few isolated areas of Kentucky, such as the federally endangered running buffalo clover (*Trifolium stoloniferum*) and Short’s goldenrod (*Solidago shortii*), apparently depended on the conditions created by the physical disturbances of large herbivores (Baskin and Baskin 1984, Campbell 1988). Whether the elk and bison historically functioned as keystones in this region or as ecological dominants (Odum 1971) is unknown.

ELK RETURN TO KENTUCKY

Measuring between 225 and 450 kilograms, the elk is the smallest extant megaherbivore in North America. Elk had a pre-European settlement distribution that encompassed southern Canada and much of the contiguous United States (Murie 1951), but it has experienced a drastic range reduction since. Two subspecies—the eastern elk (*C. e. canadensis*), which occurred in Kentucky, and the Merriam elk (*C. e. merriami*)—were extirpated due to habitat loss and overharvest (Bryant and Maser 1982). The eastern elk was extirpated by the mid-1800s (Murie 1951) and little information exists on its morphological distinctiveness or its ecological function in the landscapes of the east. Although early explorers of the Commonwealth provide accounts of elk, this information has been insufficient to infer its historic statewide abundance or relative distribution. The abundance and distribution of elk place names in Kentucky suggest a statewide distribution (Cox and others 2002). Because large generalist herbivores such as elk and white-tailed deer evolved in disturbed ecotonal landscapes where edge provides both food and cover (Geist 1982, 1998), eastern elk distribution and relative abundance may have coincided with areas of disturbance. As such, elk in Appalachia may have foraged primarily in or near river floodplains and glades, and seasonally within the forest for its herbaceous understory and mast. The relative importance the eastern elk played in shaping the biodiverse ecosystem of this region remains largely speculative.

Cervids are highly valued for consumptive and nonconsumptive reasons (Potter 1982, Conover 1997) and have been the subject of extensive and intensive restoration efforts in North America (Bergerud and Mercer 1989, Witmer 1990). Although attempts to reintroduce woodland caribou (*Rangifer tarandus*) largely failed (Bergerud and Mercer 1989), those involving elk and white-tailed deer have been more successful. Since 1900, restoration efforts in North America have facilitated population increases of white-tailed deer (*Odocoileus virginianus*) from approximately 500,000 (Downing 1987) to 26 million (Demarais and others 2000), and elk from less than 100,000 (Seton 1953) to over 1 million (Bunnell 1997). Limited post-release monitoring and scant documentation frequently hindered insight into factors responsible for elk reintroduction failures, although insufficient habitat, crop depredation, disease, poaching, and lack of sufficient funding for adequate management were often blamed (Witmer 1990).

In 1997, the Kentucky Department of Fish and Wildlife Resources (KDFWR) established an elk reintroduction program aimed at translocating approximately 1,800 elk from the western United States to Kentucky over 9 years. Initial public support for elk reintroduction was strong, although some expressed concerns that elk would depredate crops, cause automobile accidents, outcompete and cause population declines of white-tailed deer, and place additional herbivory pressure on plant communities beyond that caused by white-tailed deer (Maehr and others 1999). To address some of these concerns, KDFWR selected a 1.06 million-ha, 14-county area in the southeastern portion of the State for elk release and population establishment characterized by low densities of roads, row crops, and humans (Phillips 1997). The long-range population objective was initially 7,500, or 2.6 elk per square kilometer within the elk restoration zone (Phillips 1997). A few years later, the target population goal was increased to 10,000 and the elk zone expanded to include an additional two counties that bordered Tennessee's newly created elk reintroduction zone.

Management plans for reintroduced species should account for spatial and temporal variation in habitat use and availability at geographical scales appropriate to the species to enhance the likelihood of successful establishment. Griffith and others (1989) stated, "Without high habitat quality, translocations have low chances of success regardless of how many organisms are released or how well they are prepared for the release." Although less than 1 million people lived in the elk restoration zone, human settlement patterns closely mirrored the dendritic patterns of watersheds that were likely important historical foraging grounds of ungulates such as elk. With these areas occupied, alternative habitat of sufficient size, quality, and distribution through the restoration zone was a prerequisite for reintroduction.

Surface coal mining and subsequent reclamation have changed the physical and biotic character of the landscape in eastern Kentucky and other portions of Appalachia. Rugged, forested terrain has been converted into gently sloping mesas and plateaus that have been revegetated with grass and forb species that are typically exotic, yet provide an inexpensive means to bind soils and minimize erosion. Reclaimed mine lands are novel landscapes that sometimes harbor a bizarre combination of grassland and forest species not found in more intact lands surrounding them. Surface mines can exceed 5,000 ha in size, and uncannily resemble the rolling plains that elk inhabit in portions of its range in the western United States. As such, reclaimed coal surface mines were thought to provide a suitable habitat alternative for reintroduced elk. By March 2002, more than 1,500 elk had been released at eight sites in Kentucky, including seven on active or reclaimed coal surface mines.

DO DENATURED MOUNTAINS LIVE IN FEAR OF ELK?

Habitat selection by large herbivores can occur at multiple ecological scales as dictated by the morphophysiological adaptations of the species (Senft and others 1987). A number of North American ungulates, including elk, have foraging patterns that can operate at the landscape scale and can influence both single species and community dynamics in profound ways. Elk modify ecosystems directly through alteration of plant composition, diversity, and structure, and by serving as a nutrient and propagule carrier (Frank and others 1994, Singer and others 1994, Stewart and others 2009). These effects in turn can cause habitat changes at several spatial and temporal scales that have equally profound indirect impacts on other species, and sometimes lead to trophic cascades and alternative stable states within an ecosystem, particularly where elk predators are absent (Ripple and others 2001, Smith and others 2003). In the absence of wolves, elk in Yellowstone National Park (located in portions of Wyoming, Idaho, and Montana) exerted top-down herbivory influences that caused declines in willow (*Salix* spp.) and aspen (*Populus* spp.), which in turn influenced a host of other species dependent on these plant communities (*sensu* Smith and others 2003). Arguably, elk in Yellowstone functioned as ecological dominants in relation to their biomass rather than as ecological keystones before wolf reintroduction.

The release of elk onto minescapes posed several management questions, including whether these areas would retain elk at or near release sites, and whether habitat was of sufficient quality to foster population growth. Reintroduced elk in Kentucky quickly settled into these novel habitats and were found to select mine lands over others at all release sites at landscape and home range scales (Cox 2003). Although surface-mined areas represented only 10 percent of the elk restoration area, composite elk home ranges at each release site contained a minimum of 30 percent of this cover type. Reintroduced elk in Pennsylvania have also been found to extensively use reclaimed mine lands (Cogan 1996).

Elk in the western United States can exhibit migratory movements largely in response to the availability and quality of foods and winter severity (Craighead and others 1973, Irwin 2002). Murie (1951) suggested that the eastern elk subspecies was nonmigratory. Most elk populations that currently inhabit the eastern United States live in more temperate climates with longer growing seasons than their western counterparts. Therefore, assuming an adequate supply of food is available, eastern elk should show little to no tendency to migrate and have greater fidelity to local ranges, a pattern exhibited by elk in Pennsylvania (Cogan 1996) and Kentucky (Larkin and others 2003). In an area with little public land, fidelity to mine lands has thus far likely reduced both the number of elk-vehicle collisions and depredation of crops on smaller private parcels. Despite high fidelity to reclaimed mines, complaints about elk as nuisances persist in Kentucky, and primarily include use of cemeteries, gardens, golf courses, and orchards (KDFWR, data available on file, Frankfort, KY).

Although surface mines have severely impacted forest ecosystems in Appalachia (Hamel 2000, Holl and others 2001), these denatured lands have thus far proved ideal habitat for elk in Kentucky. Reclaimed mines have reduced topographic gradients, which allow elk to maximize forage intake, minimize energy expenditure, and remain close to forest edges and thermal cover (Coop 1973, Grace and Easterbee 1979). In addition, reclaimed mines have created vast viewscapes favorable to the gregarious elk. This herbivore is morphologically and behaviorally adapted to open areas, where it coevolved with cursorial predators such as the gray wolf (*Canis lupus*) (Geist 1998). Despite having been hunted for nearly a decade, many elk freely roam active and long-abandoned surface mines during diurnal times, conspicuously bed in open areas, exhibit little fear of humans, and thus appear to be operating as optimal foragers (MacArthur and Pianka 1966). In fact, many elk

monitored using global positioning system collars move only a few hundred meters per day as they travel from bedding to feeding areas (J. Cox, unpubl. data). Areas with a high concentration of elk activity have been humorously place-named “elk nirvana” by elk researchers.

The availability of thousands of hectares of reclaimed mineland, coupled with mild winters and the absence of large predators, appears to have provided ideal conditions for elk in Kentucky, as demonstrated by the their irruptive population growth in the past decade. Despite being hunted since 2001, elk have grown in population by an estimated average of nearly 25 percent since 1997, although population growth has slowed from an estimated high of 57.6 percent in 1998 to around 4.7 percent in 2009 (KDFWR, data available on file, Frankfort, KY).

Researchers (including the author) at the University of Kentucky have studied the habitat use, space use and movement patterns, and demographics of reintroduced elk in Kentucky for more than a decade (e.g., Larkin and others 2003). However, the initially rosy prognostication for elk population growth, viability, and compatibility with local systems belies disturbing new ecological patterns that have emerged in recent years as elk have dramatically increased in abundance and density. Concomitant with elk population growth have been more frequent observations of their impact on local plant communities and soils, particularly on mine lands and adjacent forests. These effects are perhaps most noticeable in forest-grassland edge that elk frequently traverse in large herds containing as many as 150 individuals, in forest remnant islands surrounded by mine lands, and in areas of reclaimed mines where reforestation has been initiated.

On the edge of surface mines, elk have created wide movement paths as they enter and exit forests during their daily activities. In these areas, erosion is readily visible, where soils have been excavated by trampling hooves, particularly on steeper slopes. Piles of elk feces frequently dot these trails and form concentrated pools of nutrients and plant propagules that are carried into surrounding forests where erosion channels have become sufficiently deep and furrowed. Additionally, elk browse lines in edge areas have become apparent, and tree saplings in mined areas frequently exhibit arrested growth patterns from having been repeatedly browsed. Elk have also been repeatedly observed browsing invasive exotic species, such as multiflora rose (*Rosa multiflora*), autumn olive (*Elaeagnus umbellata*), and crown vetch (*Coronilla varia*) in mined areas (Schneider and others 2006). Transport of propagules of these and other invasive, exotic plant species by elk into nearby forests has increased the likelihood that neighboring forests will become invaded and native species eventually displaced by these prolific and difficult-to-eradicate aliens. This threat is particularly serious from shade-tolerant invaders, such as multiflora rose.

Forest remnant islands and peninsulas adjacent to reclaimed mines are favorite day bed areas for elk. As early as 2 years post-reintroduction, the impacts of elk were readily apparent and manifested in overbrowsed vegetation. Most noticeable were the larger group beds characterized by hundreds of square meters of ground stripped of topsoil and vegetation and littered with prolific piles of elk feces and widespread urine deposition. Smaller forest remnants near favorite feeding grounds in particular were highly fragmented by both day beds and numerous trails that radiated from them.

Elk impacts on individual trees and shrubs also became evident early on during active restoration efforts. Elk were observed to impact entire hillsides of tree plantings to the point these areas were place-named by field researchers to demarcate elk impacts (e.g., “Rubout Ridge”). In particular, elk browsing of the nitrogen-fixing

black locust (*Robinia psuedoacacia*) has been high, constituting as much as 27 percent of woody browse in their seasonal diet (Schneider and others 2006). Much to the dismay of some of my fellow forestry researchers, entire experimental plantations of black locust have been browsed to extinction on these mines (J. Lhotka, University of Kentucky, pers. comm.). Additionally, black locust is frequently antler-rubbed and subsequently top-killed by bull elk during the rut. Repeated browsing and rubbing by elk in some areas have left thousands of black locust saplings deformed and in an arrested state of growth. Silver maple (*Acer saccharinum*) and yellow-poplar (*Liriodendron tulipifera*) are also commonly browsed by elk on reclaimed mines (Schneider and others 2006), while others such as American sycamore (*Plantanus occidentalis*) are virtually untouched.

These observations collectively suggest that elk in Kentucky are at minimum exerting direct local effects on plant composition and abundance, and perhaps are emerging as ecologically dominant species with landscape-level impacts concurrent with rapid population growth. Given the difficulty of natural vegetative succession and successful reforestation on compacted mine lands, and the unique species assemblage that occurs there, it may be possible that elk will exert ecological influence in these areas in ways that exceed what their numbers alone would suggest. Such keystone effects may alter ecological trajectories of these novel systems in ways that favor grassland and frustrate managers and reclamation bond holders alike. These potential impacts are only now stimulating interest in long-term ecological studies of elk in the region on both natural areas and reclaimed mines.

THE CHALLENGE OF REGIONAL ELK MANAGEMENT

The Kentucky elk restoration program was one of the most successful single-species reintroductions in history. Elk have at least partially filled an ecological niche vacated for nearly two centuries. However, the use of a surrogate elk subspecies coupled with its repatriation into a landscape altered by forest fragmentation, establishment of high-elevation exotic grasslands, and absence of important players such as large predators and American chestnut (*Castanea dentata*), has created an entirely new ecological dynamic vastly different from what existed immediately prior to pre-European settlement.

As of early fall 2009, it is estimated that between 10,000 and 11,000 elk roam the 16-county elk zone (T. Brunjes, KDFWR, pers. comm.), a number approaching wintering elk numbers in Yellowstone National Park. Given the scarcity of natural openings in the pre-European forests of eastern Kentucky, it is likely that elk numbers in this area were historically much lower than today, and that the regional carrying capacity for the species has been substantially increased as a result of surface mining. Widespread forest fragmentation caused by mining and timber harvest could act synergistically with high densities of elk and white-tailed deer to dramatically alter regional biodiversity, community composition, and ecosystem services as they have elsewhere (Coté and others 2004) .

The return of elk poses a conservation and management dilemma for many. Elk are a charismatic species that is popular with the general public and hunters, and thus a focal species that generates management income and tourism dollars that can ultimately benefit many other wildlife species. Concurrently, coal mining interests herald the elk as a symbol of the benefits surface mining has provided to wildlife and its users, while those opposed to surface mining often view the elk as the industry poster child associated with the ecological ills caused by mountain-top removal (J. Hardt and others, Kentuckians for the Commonwealth, pers. comm.).

Depending on the management goal of post-mined land, elk persistence and mine reclamation may be at odds. Remedies to control deer and elk overbrowsing are typically costly measures implemented at the local scale (e.g., fencing, tree guards) and not at landscape scales, where effects can be equally pervasive yet harder to detect unless carefully measured and monitored over longer time intervals. Mine managers that implement reforestation of post-mined lands will, if successful, ultimately create conditions less favorable to elk and more likely to encourage depredation of private yards, pastures, gardens, and fields of landowners with small parcels of land. In contrast, elk would likely fare much better and cause less depredation if mines were managed as grasslands, as improved pasture, or as a shrub-grassland mosaic with limited forest cover in all cases. If elk continue to increase and persist at high densities, surface mine managers will be increasingly challenged to maintain viable populations of common plant species used in reclamation. It is possible that elk depredation could lead to instances of delayed release of reclamation bonds if their impacts increased soil erosion or prevented revegetation of bare ground.

Despite the successful repatriation of several wildlife species to Appalachia during the past century, the region still contains 8 of the top 15 states with species most vulnerable to extinction (Kentucky Environmental Quality Commission 1997). Habitat loss and fragmentation, pollution, and exotic species now represent the major threats to regional biodiversity (Kentucky Environmental Quality Commission 1997). Managers of protected areas and public lands may also find the additional burden of elk management challenging, particularly in those areas that border large mines and are more likely to harbor high densities of elk. Overabundant elk could increase fragmentation and accelerate species loss, particularly along edge, but also in the interior of these lands if palatable grasses and other forbs sown on reclaimed mines become scarce and/or as elk approach local ecological carrying capacity.

Knowledge of the abundance and distribution of elk is paramount for successful management. These data allow managers to best allocate hunting and other elk management efforts to maintain population viability and minimize conflicts with humans and other species. For wildlife managers, just knowing how many elk are out there is challenging enough. Elk are particularly difficult to survey in Kentucky because they occur in relatively remote places in an area over twice the size of Yellowstone National Park. Areas occupied by elk are difficult to access from the ground, thus making population estimates based on these types of surveys unreliable. Further, elk in Kentucky typically bed in forests during the day which makes expensive aerial surveys unreliable given the gregarious nature and clustered distribution of elk.

As of 2009, six reintroduced elk populations had been successfully established east of the Mississippi River (PA, TN, KY, WI, MI, Great Smoky Mountains National Park). Kentucky's elk population remains 10 times the size of others and individuals have already moved into the neighboring states of Virginia, Tennessee, North Carolina, and West Virginia. Thus, it's likely that elk will continue to influence the region's ecosystems and land management for the foreseeable future. In particular, it appears that elk have much potential to influence policymakers and land stewards in how and why we manage surface mined lands. To better understand the role of elk in this highly altered landscape, it is recommended that long-term ecological studies be established to monitor the changes this megaherbivore will continue to bring to this biologically diverse, yet ecologically threatened area.

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ABSTRACTS

A SIMPLIFIED HARDWOOD LOG-SAWING PROGRAM FOR THREE-DIMENSIONAL PROFILE DATA

R. Edward Thomas¹

ABSTRACT

Current laser scanning systems in sawmills collect low-resolution three-dimensional (3D) profiles of logs. However, these scanners are capable of much more. As a demonstration, the U.S. Forest Service, Forestry Sciences Laboratory in Princeton, WV, constructed a 3D laser log scanner using off-the-shelf industrial scanning components. The scanner is composed of three laser scan heads positioned at 120° intervals around the log. Each log is held on steel supports as the scanner mechanism traverses the length of the log. The scanner collects a high-resolution 3D dot cloud map of exterior log surfaces. Each log scan is composed of a series of scan lines around the circumference of the log. The distance between scan lines is 1/16 (0.0625) inch and each line is composed of approximately 300 to 600 points, depending on log size.

Recent advances in image processing have resulted in methods capable of detecting external degrade log defects in the 3D laser data cloud (Thomas and others 2006). With the automated defect detection results or manually recorded log defect information, internal defect shape and position can be inferred using internal defect prediction models (Thomas 2008, Thomas 2009). Combining these three data formats yields a complete 3D view of the log, complete with external and internal defect data.

A computer program was developed using ray-tracing methods to process the 3D log data. Ray-tracing is a computer graphic and mathematical method that simulates the path of a light ray emanating from a source and records the points at which rays intersect and reflect off of objects. If you imagine the saw blade as a ray of light, you can use this method to detect where the saw intersects with the log and and surface or internal defect. The virtual saw blade travels along the log in a line and we record the points where the blade intersects the log or defects. By default, the sawing line travels parallel to the geometric center of the log. If taper sawing is desired, simply adjust the line such that it is parallel to the log surface. Similarly, it is possible to also enable curve sawing by having the saw follow the log profile.

The sawing program takes user-defined sawing patterns, applies it to the set of log data, and generates a set of lumber for each log processed. The sawing program does not perform any optimization. That is, it will saw only the sawing pattern specified. The virtual boards can be graded and a maximum value or yield calculated using the Ultimate Grading and Remanufacturing System (Moody and others 1998) or other lumber grading or processing programs. Repeating the process for different sawing patterns and rotations allows different sawing scenarios to be examined quickly and easily in the office. For example, sawing an 18-inch diameter log into 19 boards takes approximately 45 seconds on an average desktop computer.

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DO TREES FALL DOWNHILL? RELATIONSHIP BETWEEN TREEFALL DIRECTION AND SLOPE- ASPECT AND WIND IN EIGHT OLD-GROWTH OAK STANDS IN THE CENTRAL HARDWOOD FOREST

James S. Rentch¹

Abstract.—This study examined the relationship between direction of treefall and slope-aspect, and prevailing wind in eight old-growth stands where single-tree canopy gaps characterize the dominant disturbance regime. All live and downed trees were inventoried in 0.45-ha sample plots. To determine crown asymmetry, crown sizes of live trees were measured along two perpendicular axes. Directions of fall and slope-aspect of downed trees also were recorded. Regional prevailing winds and wind gusts were obtained from two nearby airports. I used circular statistics to determine whether directions of treefall for each study stand had a mean direction or whether the directions of fall were uniformly distributed. If directions of treefall had a true mean, they were then compared to mean slope-aspect and mean wind directions. Mean crown asymmetry (ratio of long and short diameters) was 1.26, and there were no significant differences in asymmetry values when current gap border trees and non-border trees were compared. At two stands, treefall directions were uniformly distributed (i.e., no mean direction). However, only one of eight stands showed a significant similarity between mean slope-aspect and mean direction of fall, and only two of the eight stands showed significant similarities between mean treefall and prevailing wind directions. While trees may fall downhill and downwind, the high variation in treefall and wind directions precluded establishing a consistent statistical relationship between these datasets. I suggest that crown asymmetry, resulting from differential crown growth on sloped hillsides as well as within canopy gaps, exerts a larger influence on direction of fall than either slope-aspect or wind direction.

INTRODUCTION

The goal of this study was to investigate the relative influence of wind and topography on directions of treefall in eight oak-dominated, old-growth stands in the central hardwood region. I asked three questions: 1) How asymmetric are the crowns of canopy trees in our old-growth study stands, and does asymmetry vary among gap border trees and non-border trees? 2) Do trees fall downslope, or are slope-aspect and direction of treefall statistically independent of each other? 3) Do trees fall downwind, or are prevailing winds and direction of treefall statistically independent?

METHODS

I inventoried all downed trees in eight 0.45-ha study plots in Ohio, West Virginia, and Pennsylvania. Direction of fall and slope-aspect also were recorded. I measured crowns of live trees along two perpendicular axes, and calculated crown asymmetry as the ratio of long and short axes. I then compared crown asymmetry

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among crown classes, and among current gap-border and non-border trees. Prevailing wind and wind gust data for 1996-2009 were obtained from Pittsburgh, PA, and Clarksburg, WV, airports. I used circular statistics to determine whether treefall directions had a true mean or whether they were uniformly distributed (Rayleigh Test). If treefall directions had a true mean, they were then compared to mean slope-aspect and mean wind directions (Watson's U^2 two-sample test) by study stand.

RESULTS

1. Mean crown asymmetry = 1.26, and there were no significant differences in means among current gap border trees and non-border trees.
2. At two stands, treefall directions were uniformly distributed (i.e., no mean direction).
3. Only one of eight stands showed a significant similarity between mean treefall and mean slope-aspect directions.
4. Only two of eight stands showed significant similarities between treefall and wind directions.

DISCUSSION

The absence of differences in mean asymmetry between current gap border trees and non-border trees suggests that all canopy trees are now, or have been, gap border trees once or several times in these stands. This result is consistent with canopy disturbance rates: between 1700 and 2000, four partial releases were detected from the tree-ring record (Rentch and others 2003). Although trees often fall downhill and downwind, there was no consistent relationship between treefall direction and aspect and wind datasets. I suggest that crown asymmetry, rather than slope-aspect or wind, is responsible for variability in treefall direction. Asymmetry may occur in three forms:

1. Due to differential tree heights and crown shading on sloped sites, trees tend to develop asymmetrical crowns that are larger and heavier on down-slope sides.
2. Trees that form the border of canopy gaps develop asymmetrical crown shapes away from the nearest large neighbor as they laterally close gaps.
3. Trees that lean may exhibit imbalance and asymmetry even if the crown itself is symmetrical around the bole.

CONCLUSION

In this study, asymmetrical crown growth of gap border trees was an important response to the change in resource availability in the gap environment. While directions of treefall often follow slope-aspect and wind direction, the high variability of treefall direction and the absence of consistent relationships between direction of treefall and aspect and wind suggest that crown asymmetry of gap border trees exerts a greater influence than either of these factors. While border trees may not be more likely to fall, when they do, the most common direction of fall is into pre-existing gaps, regardless of slope-aspect or wind direction.

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GLOBAL POSITIONING SYSTEM (GPS) AND GEOGRAPHIC INFORMATION SYSTEM (GIS) ANALYSIS OF MOBILE HARVESTING EQUIPMENT AND SEDIMENT DELIVERY TO STREAMS DURING FOREST HARVEST OPERATIONS ON STEEP TERRAIN: EXPERIMENTAL DESIGN

Daniel Bowker, Jeff Stringer, Chris Barton, and Songlin Fei¹

Abstract.—Sediment mobilized by forest harvest machine traffic contributes substantially to the degradation of headwater stream systems. This study monitored forest harvest machine traffic to analyze how it affects sediment delivery to stream channels. Harvest machines were outfitted with global positioning system (GPS) dataloggers, recording machine movements and working status. Sediment delivery to streams was monitored by scouting for overland sediment delivery paths after the completion of harvest operations. GPS positional data from the mobile harvesting equipment, along with the sediment delivery information, were analyzed in a geographic information system (GIS) to draw conclusions about the influences of buffer width; intended and actual canopy retention level; harvest equipment traffic intensity; slope; aspect; and trail type, length, and distance from stream on the delivery of sediment to headwater streams. Results from this study will have important implications for designing harvest roads and trails and effective streamside management zones (SMZs).

INTRODUCTION

The possible negative effects of timber harvesting on streams and riparian habitat are well documented. Forest operations in and around riparian areas can cause increased stream sediment levels, as well as increases in water temperature and nutrient delivery to streams (LeDoux and Wilkerson 2006). In 1972, the Federal Water Pollution Control Act and its related amendments directed the states to develop best management practices (BMPs) to address these nonpoint source pollution impacts of forest operations, of which sediment is commonly seen as the most important type in forested areas (Miller and Everett 1975). Most states' BMPs include SMZ recommendations or regulations in which a buffer strip is left undisturbed or minimally disturbed. This buffer strip is intended to filter sediment and nutrients as they travel through the SMZ, as well as mitigate the effect of canopy removal on stream temperature (Blinn and Kilgore 2001). Most sediment delivered to streams during forest operations involves road, trail, and landing construction and use (Croke and Hairsine 2006). There is thus a need for research into the mechanisms delivering sediment to small streams (Aust and Blinn 2004), and specifically how forest harvesting machinery and its associated transportation network deliver this sediment (Kreutzweiser and Capell 2001).

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METHODS

During a timber harvest on the University of Kentucky's Robinson Forest, MultiDAT GPS receivers (Castonguay Electronique, Longueuil, Quebec, Canada) were installed on all mobile harvesting equipment. At the completion of the harvest, GPS data were entered into ArcGIS software (ESRI, Redlands, CA). Three harvest boundaries will be focused on for this study, with three SMZ configurations. Boundary 1 had a 55-foot harvesting equipment buffer on the perennial stream section with canopy retention of 50-percent, a 25-foot buffer on the intermittent stream section with no canopy retention, and no buffer on the ephemeral channels with no canopy retention. Boundary 2 had 110-foot buffers on the perennial stream sections with 100-percent canopy retention, 50-foot buffers on the intermittent stream sections with 25-percent canopy retention, and 25-foot buffers on the ephemeral channels with retention of channel bank trees. Boundary 3 had a 55-foot buffer on the perennial stream section with 100-percent canopy retention, a 25-foot buffer on the intermittent stream section with 25-percent canopy retention, and no buffers on the ephemeral channels but with retention of the channel bank trees. After completion of the harvest, all perennial, intermittent, and ephemeral stream sections were scouted for overland sediment paths. Each of these paths was characterized by width, slope distance to source, slope degree, source type, skid trail morphology at source, and the influence of water-control structures.

Data will be analyzed by multiple linear regression. Each experimental unit encompasses a section of stream, the ground slope above that stream section, and the segments of the skid trail network directly upslope from the stream section. The response variable for the regression analysis will be the total width of sediment paths included in the experimental unit. Independent variables derived from the experimental unit to be analyzed in the regression will be: traffic intensity index (the number of GPS positions recorded divided by the area); SMZ buffer width; total linear feet of skid trail; linear feet of primary, secondary, and tertiary skid trail; unit slope; unit aspect; minimum distance from stream to skid trail; average distance from stream to skid trail; intended canopy retention percent; and average basal area actually retained post-harvest.

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CONTROLLING JAPANESE BARBERRY: ALTERNATIVE METHODS AND IMPACT ON TICK POPULATIONS

Jeffrey S. Ward, Scott C. Williams, and Thomas E. Worthley¹

ABSTRACT

Japanese barberry (*Berberis thunbergii*) is classified as invasive in 20 states and four Canadian provinces. It is also established in another 11 states. In addition to forming dense thickets that can inhibit forest regeneration and native herbaceous plant populations, barberry understories can harbor greatly enhanced levels of blacklegged ticks (*Ixodes scapularis*), which transmit the causal agents of several diseases, including Lyme disease (Williams and others 2009).

Since 2006, alternative methods to control barberry have been examined on 122 plots at 22 sites in southern New England and New York. Plot sizes ranged from 0.1 to 0.6 acres. Initial treatments (prescribed burning, mechanical mowing with a rotary wood shredder or with a brush saw, and directed heating with a propane torch) killed the aboveground stems in early spring. All initial treatments were equally effective in reducing clump size and cover (Ward and others 2009). Treatment costs ranked from least to most expensive were: prescribed burning, mechanical mowing with a brush saw, directed heating with a propane torch, and mechanical mowing with a rotary wood shredder. Without a follow-up treatment, individual clumps recovered to 50 percent of original size within 16 months of initial treatment (Ward and others 2010).

Follow-up treatments in July to kill the small sprouts that developed from the root crown included foliar application of triclopyr or glyphosate, and directed heating. Directed heating treatments were repeated in October. Effectiveness of follow-up treatments varied by initial size of barberry clumps. Mortality was greater than 86 percent for clumps smaller than 4 feet with directed heating or herbicide treatments. However, directed heating was less effective than herbicides for larger clumps (≥ 4 feet); mortality was 65 and 90 percent mortality, respectively. All follow-up treatment options reduced barberry cover by 90 percent or more. Follow-up herbicide treatment costs were one-sixth that of directed heating. Regardless of follow-up treatment, some clumps will survive and additional treatments the following summer will be needed for eradication.

Control of Japanese barberry had an effect on earthworm biomass ($F = 6.86$, d.f. = 2, $P < 0.003$). Earthworm biomass averaged 197 pounds per acre where barberry was controlled and 535 pounds per acre in dense barberry infestations. By comparison, earthworm biomass was 304 pounds per acre in adjacent forests without barberry. The high earthworm biomass may have contributed to the low litter weight on plots with barberry infestations, 2.3 tons per acre, compared with 5.0 tons per acre observed on our prescribed fire plots without barberry.

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A surprise benefit of controlling Japanese barberry is the reduction of blacklegged (or deer) tick populations. Relative to adjacent plots where barberry infestations were left untreated, white-footed mice (*Peromyscus leucopus*) on plots where barberry was controlled had 27 percent fewer larval ticks. Controlling barberry also reduced the density of adult ticks by 58 percent. In addition, tick infection prevalence with the spirochete *Borrelia burgdorferi*, the causal agent of Lyme disease, was much higher in barberry infestations. As a result, 166 ticks per acre were infected with *B. burgdorferi* in untreated barberry infestations compared with 50 infected ticks per acre where barberry was controlled and only 20 infected ticks per acre in adjacent forests without barberry. Thus, controlling Japanese barberry can improve public health by reducing both the number of blacklegged ticks and *B. burgdorferi*-infected ticks that commonly feed on humans.

Japanese barberry can be effectively controlled in a single growing season by integrating an initial early-season treatment (prescribed fire or mechanical) that kills the aboveground tissues, followed by a mid-summer follow-up treatment such as directed heating or targeted herbicide application. Although time consuming, propane torches provide a nonchemical alternative in forests, nature preserves, parks, or other ecologically sensitive areas where herbicide use is restricted and where barberry infestations are still light.

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DO MATURE FORESTS PRESENT BARRIERS TO NONNATIVE PLANT INVASION? *LONICERA MAACKII* INVASION IN DECIDUOUS FORESTS OF CENTRAL KENTUCKY

Heather N. Wilson, Mary A. Arthur,¹ Ryan W. McEwan, Brian D. Lee, and Robert D. Paratley

ABSTRACT

Nonnative invasive plants can negatively affect the abundance and survival of native plant species and alter ecosystem function. Amur honeysuckle (*Lonicera maackii*) is an invasive shrub that is an increasingly onerous problem for forest management in eastern North America. While much is known about the effects of Amur honeysuckle on native plant communities, less is known about the invasion process and the factors that influence the encroachment of Amur honeysuckle into forested sites. Factors that have been implicated in the spread of invasive shrubs include canopy openness, proximity to urban/suburban areas and roads, and historical disturbance. We examined site-specific characteristics that correlated with the establishment of Amur honeysuckle and other nonnative invasive plants in mature forests in central Kentucky. Data were gathered from 15 mature forest remnants throughout the Inner Bluegrass region (IBG) of Kentucky. Forest age was determined from tree core samples, which showed that tree age averaged 70 years. To characterize individual sites, we measured frequency and cover of nonnative invasive shrubs, and co-occurring native shrubs, as well as the basal area of overstory trees. Trees commonly found throughout forest stands included sugar maple (*Acer saccharum*), white oak (*Quercus alba*), chinkapin oak (*Q. muhlenbergii*), shagbark hickory (*Carya ovata*), and white ash (*Fraxinus americana*), with a few other species interspersed. Total basal area of trees > 10 cm diameter at breast height ranged from 23.9 to 41.9 m² ha⁻¹, with a total stem density of 339.4 to 841.7 stems ha⁻¹. Forest floor litter depth, species composition of the leaf litter, and soil chemical characteristics were also measured. Litter depth was measured along transects in each study site and samples were separated into oak and 'other' species in the lab. Soil pH, percent carbon (C) and nitrogen (N), and C/N were measured. Amur honeysuckle was found in 13 of 15 study sites; percent cover ranged from 0.03 to 64 percent. The percent cover of Amur honeysuckle in a forest was significantly related to forest floor litter depth ($P = 0.03$): as the depth of the leaf litter increased, the percent cover of Amur honeysuckle decreased. Of the variables measured, only forest floor litter depth ($P=0.01$) and the percent of oak litter on the forest floor ($P=0.004$) were significantly related to Amur honeysuckle presence, and the relationship was also negative. Although previous investigations have found basal area to be a significant predictor of invasive plant presence, it was not a predictor of Amur honeysuckle presence ($P=0.36$) or cover ($P=0.60$) in this study. Co-occurring native shrubs, as a group, had no significant effect on the presence ($P=0.91$) or abundance ($P=0.46$) of Amur honeysuckle. Further, none of the individual species used in analysis, including coralberry (*Symphoricarpos orbiculatus*, $P=0.31$), pawpaw (*Asimina triloba*, $P=0.13$), and spicebush (*Lindera benzoin*, $P=0.93$), were significant in explaining Amur honeysuckle cover. Similarly, soil characteristics such as soil pH ($P=0.08$; $P=0.34$) and soil percent C ($P=0.96$; $P=0.16$) did not explain either the presence or cover, respectively, of the exotic shrub.

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Nonmetric multidimensional scaling (NMDS) ordination was used for spatial analysis of all our invasive species and environmental variables. NMDS also showed a clear separation (in ordination space) among species (Axis 1 $R^2 = 0.678$, Axis 2 $R^2 = 0.229$, final stress 7.2), where the axes show the strength of the relationship between the invasive species and the environmental variables. The strongest relationships between the two sets of variables were for soil pH, litter depth, oak basal area, and percent oak in the leaf litter; invasive species generally arrayed directly opposite of the forest floor leaf litter vector.

Our data suggest that forest floor mass creates a barrier to invasion by exotic plants and that forests dominated by oak species are more resistant to invasion and establishment by Amur honeysuckle.

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MAXIMUM ENTROPY MODELING OF INVASIVE PLANTS IN THE FORESTS OF CUMBERLAND PLATEAU AND MOUNTAIN REGION

Dawn Lemke, Philip Hulme, Jennifer Brown, and Wubishet Tadesse¹

ABSTRACT

As anthropogenic influences on the landscape change the composition of 'natural' areas, it is important that we apply spatial technology in active management to mitigate human impact. This research explores the integration of geographic information systems (GIS) and remote sensing with statistical analysis to assist in modeling the distribution of invasive species. It is applicable to both native and nonnative species and can help land managers to identify areas of conservation importance and areas of threat.

In this study, maximum entropy (MaxEnt) models were used to map the extent of Japanese honeysuckle (*Lonicera japonica*), tall fescue (*Lolium arundinaceum*), and mimosa (*Albizia julibrissin*), three nonnative plant species, in the Cumberland and Mountain Region (CPMR), which extends from northern Alabama through Tennessee, Kentucky, and Virginia. MaxEnt is a nonparametric modeling program that integrates occurrence-only data without the assumptions of some of the more traditional statistical methods. The objectives of this study were to model distribution of three invasive species and assess potential hotspots.

The CPMR covers 59,000 km² and supports one of the most diverse woody plant communities in the eastern United States. Like many of the forests in the eastern United States, the native deciduous hardwood forests of the CPMR have a long history of land-use change driven by agricultural conversion, timber extraction, and more recently, urban sprawl and large-scale conversion to intensively managed pine plantations. These human activities may influence the distribution and spread of invasives.

Information on the invasive species was extracted from the U.S. Forest Service's Forest Inventory Analysis (FIA) database as absence/presence data. Landscape-associated variables were derived from digital information and grouped in six categories: those derived from remote sensing data, elevation, land use, those related to climate, those related to anthropogenic disturbance, and water. Correlation within groups was assessed, and those with high correlation (>0.80) removed. MaxEnt models were run for each group of variables. Variables that contributed more than 5 percent were then rerun in final models. Accuracy of model output was assessed by using area under the curve (AUC) of the receiver operating characteristic, and withholding 30 percent of the data. Hotspot analysis was done by mapping models across the region.

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Ten variables were used in the three models, with elevation used in all three and minimum temperature used in two of the models. Japanese honeysuckle had an AUC of 0.87 and the model showed probable occurrence in 42 percent of the CPMR. Tall fescue had an AUC of 0.84 and the model showed probable occurrence in 11 percent of the CPMR. Mimosa had an AUC of 0.92 and the model showed probable occurrence in 13 percent of the CPMR. To assess similarities between species and possible hotspots of invasions, the 'best' MaxEnt models for each species were compared. In comparing the three species, we found that Japanese honeysuckle has the widest distribution; both Japanese honeysuckle and mimosa are concentrated in the south. By adding the model values together, we can assess areas where both invasives have a low chance of occurrence or both show a high chance of occurrence.

MaxEnt modelling successfully predicted occurrence for all three species (AUC 0.84 to 0.92), regardless of species prevalence (mimosa 2 percent, tall fescue 5 percent and Japanese honeysuckle 30 percent). All models had a mix of environmental and anthropogenic variables, suggesting species distribution is related to both environmental niche and spread through human activities. Overall elevation had the greatest impact on the models. Not only was it selected in all models in which it was entered, but it was also the dominant variable in most models. Elevation influences temperature, rainfall, and soils in the region, and these have been shown to be a controlling factor for many species worldwide. The hotspot analysis identified areas at greatest risk of invasion. This work will be continued to examine the distribution of 11 nonnative species and assess how climate change may influence their distribution.

This work has been supported by the Center for Forest Ecosystem Assessment at Alabama A&M University (AAMU) and was partially funded by the National Science Foundation (Award ID: 0420541). Samuel Lambert of the U.S. Forest Service was instrumental in extracting the FIA data. Callie Schweitzer (U.S. Forest Service) and Yong Wang (AAMU) have given valuable advice in the development of this research.

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MAKING HISTORY: FIELD TESTING OF BLIGHT-RESISTANT AMERICAN CHESTNUT (*CASTANEA DENTATA*) IN THE SOUTHERN REGION

Stacy L. Clark, Scott E. Schlarbaum, A.M. Saxton, and Fred V. Hebard¹

INTRODUCTION

The American chestnut (*Castanea dentata* Marsh. Borkh.) was decimated by an exotic fungus (chestnut blight [*Cryphonectria parasitica* Murr. Bar]) in the early part of the 20th century. The American Chestnut Foundation (TACF) uses a back-cross breeding program to produce a tree that is predicted to be American chestnut in character with blight resistance from Chinese chestnut (*Castanea mollissima* Blume). The first breeding generation produced in TACF's program predicted to have stable blight resistance is referred to as the "BC₃F₃" and nuts were first available for testing in autumn 2007. Our objective was to examine first-year response in bud break phenology and growth and to determine competitiveness of TACF's American chestnut breeding material, including the putative blight-resistant generation.

METHODS

Nuts were collected at TACF's Research Farm in Meadowview, VA, in autumn 2007, and grown for 1 year at the Georgia Forestry Commission's state nursery near Byromville, GA. Experimental material consisted of five breeding generations containing: three genetic families of American chestnut (240 seedlings), two genetic families of Chinese chestnut (182 seedlings), two genetic families of BC₁F₃ (206 seedlings), two genetic families of BC₂F₃ (194 seedlings), and five genetic families of BC₃F₃ (337 seedlings). Seedlings from each family were divided into two seedling size classes (small and large) by visually assessing size. Seedlings were planted at three national forests in Region 8 of the U.S. Forest Service in February 2009. Stands were treated with a shelterwood with reserve prescription (2-4 m² ha⁻¹ residual basal area), and stump sprouts were treated with an herbicide application. Seedlings were planted on a 2.5- by 2.5-m spacing using a single-tree plot, incomplete block design with a factorial treatment arrangement of family and seedling size. We assessed bud break in late April/early May using phenological stages ranging from 0 (bud dormant) to 5 (internodes visible and leaves enlarged). We measured planted chestnuts for total height and ground-line diameter (GLD), determined whether the tree was free-to-grow, and noted any damage. We assessed competition within a 6.2-m² plot around each planted chestnut. Four classes defined density of competition (Low = 0-5; Medium = 6-15; High = 16-30; Extreme >30). All data were analyzed in SAS (SAS Institute, Cary, NC). Main effects (planting location, breeding generation, family, and seedling size class) and interactions were tested using PROC MIXED. Differences were significant at a $\alpha \leq 0.05$.

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RESULTS AND DISCUSSION

Across three locations, survival averaged 88 percent, and 93 percent of seedlings were free-to-grow. Bud break was affected by breeding generation, family, and seedling size class, but not by location. Chinese seedlings were more developed (ranking = 3.5) than any other generation. The BC_3F_3 were more developed (ranking = 2.3) than American seedlings (ranking = 1.8), possibly indicating a departure from desired characteristics of TACF's breeding program. Small seedlings were significantly but only slightly more developed (ranking = 2.6) than large seedlings (ranking = 2.5).

Across locations, seedlings were 96 cm in height and 12.3 mm in GLD at planting, and were 97 cm tall and 13.2 mm in GLD after one growing season. Locations exhibited differences in height and GLD growth, likely due to differences in deer browse pressure. Height growth was also affected by breeding generation, family, and seedling size. Chinese seedlings had the smallest height growth at each location (-25 to 9 cm), but height growth of BC_3F_3 seedlings depended on location (-3 to 18 cm). The American, BC_1F_3 , BC_2F_3 , and BC_3F_3 seedlings did not differ in height growth. Breeding generation, family, or seedling size did not affect GLD growth. Small size seedlings had larger height growth (7 cm) than large size seedlings (3 cm), but small seedlings were smaller in total height after 1 year (83 cm), compared to large size seedlings (111 cm).

Location 3 had the smallest height growth due to heavy deer browse pressure (84 percent of trees browsed). Trees without deer browse were 26 cm taller than trees with deer browse. Blight affected six American trees and one BC_3F_3 tree.

Yellow-poplar (*Liriodendron tulipifera* L.) and red maple (*Acer rubrum* L.) were the most common and the tallest competitors. Plots most commonly had a low density class (48 percent of plots) and only 1 percent of plots had an extreme competition density class. Height of the tallest competitor on average ranged from 82 cm to 147 cm. With the exception of Location 2, the majority of planted chestnut trees were taller than the competition.

Protection of seedlings will be critical for future development, and deer shelters and fencing are being erected. Treating stump sprouts prior to planting appears to be effective at controlling competition after one growing season, but competition control will be applied if planted seedlings begin to lose dominance. Seedlings will need to overcome planting shock and deer browse pressure before we can make inferences on growth performance and blight resistance. We predict that larger size seedlings will be more competitive over time, despite their slower growth in the first year. We predict the putative blight-resistant chestnut seedlings (BC_3F_3) will eventually thrive and be competitive with natural competition.

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EXAMINING POSSIBLE CAUSES OF MORTALITY IN WHITE PINE SEEDLINGS

Elizabeth Gilles, Ronald Reitz, Greg Hoss, and David Gwaze¹

ABSTRACT

White pine (*Pinus strobus* L.) is one of the most important timber trees in the northeastern United States and eastern Canada (Demeritt and Garrett 1996). White pine is not native to Missouri; it is commonly planted for wind breaks and erosion control and as an ornamental. Unusual mortality of bare-root seedlings of white pine purchased from the George O. White State Forest Nursery has been occurring after planting throughout the state. There are numerous possible explanations for white pine seedling mortality which may act alone or in combination. Lifting, packing, transportation, storage, and planting techniques can all influence seedling mortality after planting (Venator 1985). Also, extreme weather conditions after planting will increase seedling mortality (e.g., drought, overwatering, flooding, hot summers). The objective of this study was to determine the perspectives of private landowners toward possible causes of white pine mortality.

In early 2009, a telephone survey was conducted of individuals selected from the State Forest Nursery customer database for 2007, 2008, and 2009. One hundred and sixty people were called between April 21 and May 13, 2009. Each household was called two to five times during this period. Fifty individuals completed the telephone survey.

Forty-eight percent of respondents reported an increase in white pine seedling mortality during the past 3 years, but the increase was not limited to any particular area of the state. A majority of respondents who noticed an increase in mortality (62 percent) indicated that the mortality problem started in 2007. Most respondents (60 percent) did not have an opinion on what might be causing the mortality; however, 45 percent of the respondents who expressed an opinion about the possible cause of white pine seedling mortality thought that the mortality was caused by drought or flooding after planting. In fact, a number of people suggested that 2007 had drought-like conditions, and 2008 and 2009 were too wet, and this combination contributed to seedling mortality during the past 3 years. Twenty percent of respondents that experienced a problem believed that packaging and transportation caused the seedlings to die.

The factors responsible for seedling mortality were believed to be 1) rough handling during transportation, which caused seedlings to arrive with broken tops; 2) the large number of seedlings in each bundle, which they believe caused root mold; 3) the variation in size of seedlings in each bundle, which they believed led to machine planting problems; and 4) rough handling during transportation, which they believed caused roots of seedlings to arrive at their destination dried out. To avoid roots drying out, it was suggested that roots should be clay dipped prior to shipping.

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Ten percent of the respondents that experienced a problem believed that the seedlings were too big. They indicated that the overall size had increased over time and was highly variable within individual bundles. According to the respondents, the size of the seedlings and variability in sizes in each bundle made machine planting difficult. They also believed that size of the seedlings resulted in an unfavorable root-to-shoot ratio.

Other reasons advanced by a small minority of respondents were 1) planting on wrong sites, such as clayey soils; 2) planting on sites with little sunlight; and 3) conditions under which the seedlings were raised at the State Forest Nursery. Conditions at the nursery that may affect seedling survival after field planting include watering regime, soils, fertilization, root pruning, length of cold storage, and how seedlings are stressed prior to lifting.

There were no noticeable differences in transportation, planting, or management of the seedlings after planting between people experiencing seedling mortality and those who had not. For example, some people had been planting white pine for 20 years and had not changed the way they handle and manage white pine, but observed mortality problems recently. On the other hand, other people planted white pine over the same period and their trees were fine.

The mortality problems have caused several people to stop buying white pine seedlings from the nursery. Looking at the list of who bought white pine in 2007, 2008, and 2009, we see that although there is around the same number of names, there is a big change in who is on the list in 2007 and 2008. Very few individuals show up on the list for all 3 years. We think customers may have noticed a change in survivorship and may have started buying their white pine seedlings elsewhere.

The survey suggested that the increased white pine seedling mortality started in 2007, and weather, transportation and packaging, seedling size, and nursery conditions were the probable causes. White pine seedlings are known to be intolerant to drought, heat, and flooding (Gilman and Watson 1994). While these weather factors are beyond our control, other factors could be modified to improve survivability. For example, private landowners should avoid planting white pine in clayey soils or in areas where the seedlings do not receive enough sunlight. The State Nursery should ensure that the root-to-shoot ratio is high because seedlings with a higher ratio usually have better survival (South and others 2005). The nursery can also improve uniformity of seedlings within each bundle. Length of cold storage can be ruled out as a factor because seedlings are now rarely stored for a long time. Gwaze and Hoss (2007) found that seedlings of white pine planted immediately after lifting had better survival.

Further research into other factors believed to be affecting mortality in white pine seedlings is needed. In the future, the nursery is likely to examine the following factors: 1) seed source effects (North Carolina and Virginia), age effects (2- and 3-year-olds), effect of watering (normal watering after lifting and watering plus root gel application), and effect of rough handling from nursery to planting site (UPS and individual landowner pick up). The investigation suggested by the State Forest Nursery provides an important step in understanding factor(s) responsible for the seedling mortality in white pine. The results will be useful in enhancing the planting of white pine seedlings to meet aesthetic, economic, and restoration objectives.

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PRELIMINARY EVALUATION OF OPERATIONAL OAK REGENERATION METHODS IN CENTRAL MISSOURI

Carrie Steen, Gus Raeker, and David Gwaze¹

ABSTRACT

Oak regeneration on mesic upland sites continues to be a major challenge throughout the hardwood forests of eastern North America. Oak forests across northern and central Missouri have been historically maintained through natural and anthropogenic disturbance regimes. This cycle has been interrupted in many areas through fire suppression, high-grading, and intensive grazing so that more shade-tolerant species are out-competing oak advancement. Resource managers are seeking silvicultural methods that increase or sustain the oak component on these mesic sites. Reducing overstory and midstory densities to increase light, reduce competition, and increase growing space for oaks is one strategy of increasing oak regeneration potential.

Data were collected on permanent plots in three stands each at Daniel Boone, Little Lost Creek, and Reifsnider Conservation Areas (CA) in the St. Louis Region of the Missouri Department of Conservation in 2007. Each stand within a CA was randomly assigned one of three mid- and overstory reduction treatments: leaving 40-percent stocking using herbicide (40H), 60-percent using herbicide (60H), or 60-percent using mechanical methods (60M). One to two acres of each stand were reserved for a control. Removal focused on maple, the most competitive species in the region. Treatments were completed by spring 2008. After allowing a growing season for response, we collected data again in the winter of 2008/2009. Analysis of variance (ANOVA, $\alpha = 0.1$) was used to measure the effects of treatments on canopy cover and density.

Treatment effects were detected for mean canopy cover ($P = 0.010$). Cover was reduced by 12.3 percent in the 40H treatments, 2.3 percent in the 60H treatment, and 7.6 percent in the 60M treatment (Fig. 1). Before treatment, the most abundant overstory species were oaks (64 percent) followed by maples (27 percent). Selective removal of maple increased the oak composition to 86 percent and sharply decreased the composition of maples to 5 percent across all treatments.

Treatments substantially reduced maple composition and slightly reduced oak species composition in the midstory (Fig. 2), but treatment effects were not significant for oak species composition. Reduction in maple composition was accompanied by an increase in proportion of other species.

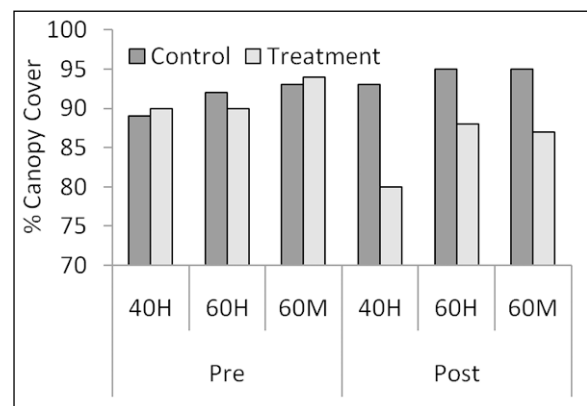


Figure 1.—Estimates of canopy cover before treatment and 1 year after treatment in the St. Louis Region of the Missouri Department of Conservation.

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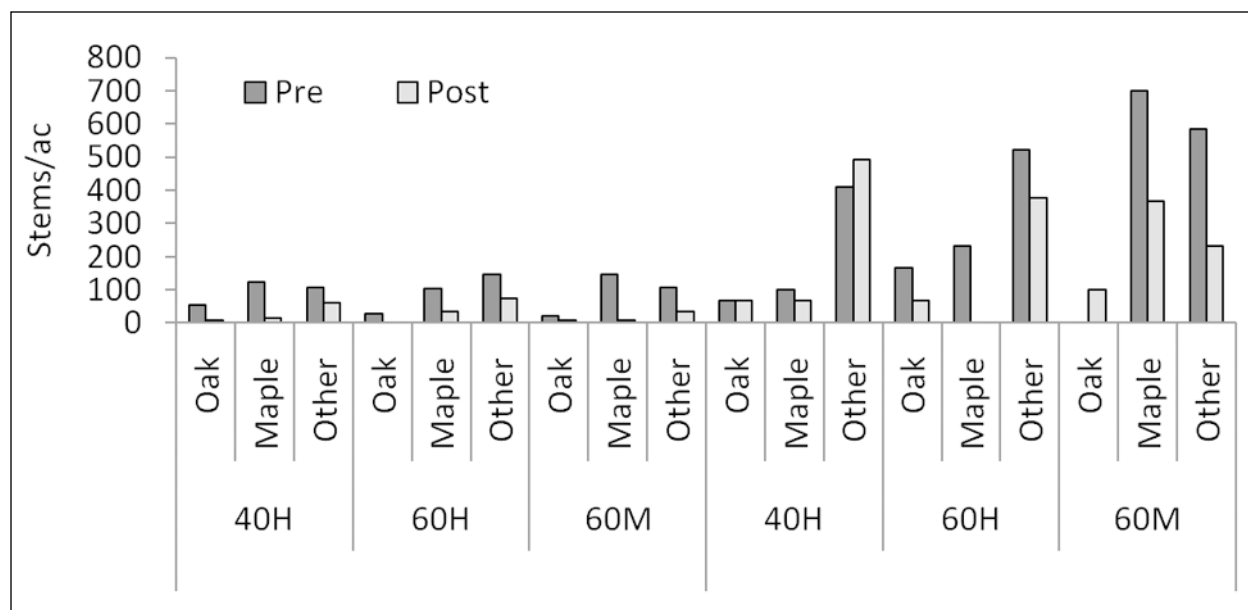


Figure 2.—Density of lower midstory (< 1.5 in. d.b.h. and > 3.3 ft.) and upper midstory (> 1.5 to < 4.5 in. d.b.h.) before treatment and 1 year after treatment in the St. Louis Region of the Missouri Department of Conservation.

Oaks made up a significant proportion (8-31 percent) of the regeneration (< 0.3 feet) prior to treatment (0 to nearly 10,200 trees/acre). Treatments resulted in slight reductions in oak and maple regeneration, but these reductions were not significant. Average seedling height increased after treatment, but the increase in height was greater for maple and other species than for oak.

No significant relationships existed between canopy cover and seedling stem density before treatment. However, a scatter plot shows that canopy cover less than 89 percent was required to consistently achieve a density of > 600 oak stems/ac without falling below that density.

In conclusion, the study demonstrates that although oaks were the dominant overstory species before and after treatments, other species, particularly maples, were the dominant species in the midstory. Because oaks were a minor component in the midstory after treatments, it is likely that removing the overstory trees in the near future will reduce the oaks to a minor component in the new stands. Although the primary competitor (maple) has been dramatically reduced, the proportion of oaks in the regeneration stage is moderate. Oak seedlings are too small to compete with other midstory trees and non-oak regeneration. These results are preliminary, but management techniques may require more aggressive removal of midstory competitors beyond the primary competitor.

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SHADE TOLERANCE OF SELECTED AFFORESTATION SPECIES ON LOESS PLATEAU, CHINA

Naijiang Wang, Yong Wang, and Callie J. Schweitzer¹

ABSTRACT

To select tree and vegetation species for afforestation on Loess Plateau of China, we tested the shade tolerance of nine tree and one vine species. We planted 3-year-old seedlings in the greenhouse of the Seedling Nursery at Northwest Agriculture and Forest University in China. A photosynthesis curve was constructed to determine the light compensation point (LCP) and light saturation point (LSP) of each species using a LI-6400 Portable Photosynthesis System (LI-COR Inc., Lincoln, NE) at the peak of the growth season in 2006. We used LCP as an indicator of shade tolerance. Honeysuckle (*Lonicera japonica*), *Sabina vulgaris*, and shiny-leaved yellowhorn (*Xanthoceras sorbifolia*) had the highest shade tolerance with the lowest LCP ($< 20 \mu \text{mol m}^{-2} \text{s}^{-1}$), followed by Liaodong oak (*Quercus wutaishanica*), oriental arborvitae (*Platycladus orientalis*), apricot (*Prunus armeniaca*), Chinese pine (*Pinus tabulaeformis*), forsythia (*Forsythia suspensa*), and lilac (*Syringa oblata*). Locust (*Robinia pseudoacacia*) had the lowest shade tolerance (LCP $< 180 \mu \text{mol m}^{-2} \text{s}^{-1}$). On the other hand, lilac, oriental arborvitae, and locust had the highest LSP ($> 1500 \mu \text{mol m}^{-2} \text{s}^{-1}$), and the remaining seven species had similar LSP values, ranging from 1,240 to 1,400 $\mu \text{mol m}^{-2} \text{s}^{-1}$. The results are consistent with our field observations that lilac, oriental arborvitae, and locust are shade-intolerant and grow well at sites with intense light, while shiny-leaved yellowhorn, *Sabina vulgaris*, and honeysuckle can grow under canopy with limited light. Lilac, oriental arborvitae, and locust also had the largest range between LCP and LSP, suggesting that these species can grow in habitats with large variation in light intensity.

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GROWING HIGH QUALITY HARDWOODS: PLANTATION TRIALS OF MIXED HARDWOOD SPECIES IN TENNESSEE

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ABSTRACT

Hardwood plantations are becoming increasingly important in the United States. To date, many foresters have relied on a conifer plantation model as the basis of establishing and managing hardwood plantations. The monospecific approach suggested by the conifer plantation model does not appear to provide for the development of quality hardwood logs similar to those found in natural hardwood stands. Thus, there is interest in creating mixed species plantations to simulate natural hardwood stand development. The key assumption is that high quality logs can be better produced through following natural stand development patterns. Conceptual and quantitative models have been proposed to validate this assumption and help guide forest land managers planning hardwood plantations. However, robust experimental plantation trials for temperate hardwood species have yet to be developed. Here, we outline the rationale, conceptualization, and implementation of an experimental mixed hardwood species plantation in eastern Tennessee. Three two-species mixtures (1. yellow-poplar/cherrybark oak, 2. sweetgum/cherrybark oak, and 3. black cherry/cherrybark oak) were planted within three spacing patterns (6 x 6 ft, 8 x 8 ft and 10 x 10 ft). Each species mixture by spacing combination was replicated three times in 0.5-acre blocks and planted in early spring 2009. Early hypothetical development patterns and silvicultural prescriptions are explored. Modeling research suggests that mixed species hardwood plantations will provide for the development of a greater volume of clear, knot-free wood and therefore higher-quality logs. This plantation trial will be an important component in validating the assumptions and results of the models that have been developed.

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

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EFFECTS OF TWO PRESCRIBED FIRES ON RED MAPLE REGENERATION ACROSS FOUR LEVELS OF CANOPY COVER

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INTRODUCTION

Oak species have maintained an important role in eastern U.S. forests for the last 8,000 years (Foster and others 2002). Intense changes in land use and disturbance regimes brought on by European settlement helped oak persist and expand its range through the 1800s. Mixed-oak forests persist as the predominant forest type in present-day eastern forests, but very little oak recruitment into the overstory has occurred since the implementation of fire suppression in the early 1900s (Abrams 1992). Advanced oak regeneration on high-quality and some intermediate-quality sites is being replaced in the midstory by more mesic, shade-tolerant species (Abrams and Nowacki 1992). Red maple (*Acer rubrum*) is often the most frequently occurring competitor on upland oak sites in much of the eastern United States (Abrams 1998, 2005). Current thinking is that reintroduction of representative fire regimes into eastern oak systems may help promote oak recruitment and impede the development of red maple competition.

Brose and Van Lear (1998) found that red maple was controlled most effectively with spring and summer burns, when carbohydrate reserves in the rootstocks are low and fire intensities tend to be higher. Burns of lower intensity present less potential hazard, but several low-intensity burns may be necessary to achieve the same results as a single, high-intensity burn (Loftis 1990, Brose and Van Lear 1998).

OBJECTIVE

The objective of this study was to determine the additive effects of two low-intensity prescribed fires on red maple regeneration across four levels of canopy cover by testing the hypothesis that red maple densities in medium and large size classes (25 cm tall – 2.54 cm diameter at breast height [d.b.h] and > 2.54 cm d.b.h., respectively) will be reduced to a greater degree following the second burn than following the first.

METHODS

SITE DESCRIPTION

Second-growth natural oak stands on state forest land were selected as study sites in Michigan: southern Crawford County (84°45'W, 44°31' N, elevation 400 m) and southern Roscommon County (84°41' W, 44°14' N, elevation 300 m). Both counties are within the Grayling Outwash Plain of the Highplains District of the northern Lower Peninsula (Albert 1995). All stands occurred on sandy, mixed, frigid, Alfic Haplorthods developed in pitted outwash, and soil physical and chemical properties among stands were comparable (Kim and others 1996). All sites were designated as moderately productive. Slopes were ≤ 5 percent. Site index for

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northern red oak in the Lakes States region was 17-18 m at 50 years (Carmean and others 1989, Kim and others 1996). Tree ring counts on stumps, following canopy treatment in 1991, indicated the oak stands were 88-100 years old.

EXPERIMENTAL DESIGN

Natural oak stands were divided into three replicate blocks measuring 1.74 ha. Each of the three blocks was subdivided into four plots measuring 66 m x 66 m (0.44 ha). Each plot randomly received one of four canopy cover treatments: clearcut, 25-percent residual canopy, 7-percent residual canopy, or uncut control. All stands were cut between fall 1990 and early spring 1991. Four understory treatment subplots measuring 15 m x 15 m (0.02 ha each) were arranged in a square at the center of each canopy treatment plot in 1991, and maintained intermittently through 1996.

MEASUREMENTS

Natural regeneration was measured in 2001, 2003, 2006, and 2009 at four sampling plots within each understory subplot at the end of the growing season in late July/early August. Genets of woody stems were tallied by species into three size classes: (small) stems < 25 cm height, (medium) stems 25 cm height – 2.54 cm diameter, and (large) 2.54 cm diameter – 10 cm diameter. Small size-class stems were measured within square meter quadrats. Medium size-class stems were measured within 2-m diameter circular plots. Large size-class stems were measured within 4-m diameter circular plots. Smaller sampling plots were nested within larger plots.

STATISTICAL ANALYSIS

Red maple densities were averaged across understory plots to achieve canopy cover treatment means for each oak block. Treatment means and standard errors were calculated using SAS 9.2 (SAS Institute, Cary, NC). Paired t-tests were used to evaluate differences between pre- and post-burn seedling densities associated with each fire using SAS 9.2. P-values were reported at a significance level of $\alpha = .05$ for two-tailed t-tests.

RESULTS

MEDIUM SIZE CLASS

In clearcut treatments, red maple densities were reduced by 54.4 percent following fire 1 and by 40.3 percent following fire 2. In 25-percent canopy cover, densities were reduced by 43.5 percent following fire 1 and by 26.77 percent following fire 2. In 75-percent canopy cover, densities were reduced by 44.0 percent following fire 1 and by 31.7 percent following fire 2. In uncut control treatments, densities were reduced by 60.1 percent following fire 1 and by 43.2 percent following fire 2.

LARGE SIZE CLASS

In clearcut treatments, red maple densities were reduced by 48.6 percent following fire 1 and by 3.0 percent following fire 2. In 25-percent canopy cover, densities were reduced by 65.5 percent following fire 1 and by 24.3 percent following fire 2. In 75-percent canopy cover, densities were reduced by 79.5 percent following fire 1 and by 58.3 percent following fire 2. In uncut control treatments, densities were reduced by 85.7 percent following fire 1 and by 75.0 percent following fire 2.

DISCUSSION

The hypothesis that red maple stems in the two larger size classes would be reduced more by the second fire than by the first was not supported. Each fire reduced red maple densities across all canopy treatments, but densities were reduced to a greater degree following the first fire. Many of the larger surviving stems were stump sprouts, which emerged from thinned midstory saplings following canopy treatments in 1991. These stems had 10 years to develop between initial thinning treatments and the first prescribed fire. Densities may have been reduced more following the first fire because of high mortality among stressed, ill-formed, or diseased stems, leaving only the most robust, well established stems, which then had six growing seasons to recover before the second fire. Reducing these robust stems may require continued repeated burning.

CONCLUSION

Low-intensity prescribed burns implemented more frequently than in this study may help to reduce large red maple regeneration that can survive and recover between longer periods. More frequent burns, however, may also have an adverse effect on oak regeneration. Overall, the two prescribed fires had a cumulative effect, reducing red maple densities by 46-74 percent in the medium size class and by 53-87 percent in the large size class, which may increase the competitive ability of desired oak regeneration.

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INFLUENCE OF ALTERNATIVE SILVICULTURAL TREATMENTS ON SPATIAL VARIABILITY IN LIGHT IN CENTRAL HARDWOOD STANDS ON THE CUMBERLAND PLATEAU

Stephen F. Grayson, David S. Buckley, Jason G. Henning, Callie J. Schweitzer, and Stacy L. Clark¹

INTRODUCTION

Effective oak silvicultural treatments allow light to reach the forest floor with sufficient intensity and duration to enable establishment, growth, and development of preferred species. Although it is intuitive that increases in light will accompany various levels of canopy removal, specific amounts and the distribution of light resulting from different silvicultural treatments have not been precisely determined. Our objectives were to document the amounts and distribution of photosynthetically active radiation (PAR) in stands receiving alternative silvicultural treatments.

METHODS AND STUDY SITES

Each treatment and controls were replicated three times in three comparable upland oak forest type units (upland sites typical of the Cumberland Plateau) on the London Ranger District of the Daniel Boone National Forest (latitude 37°05' N, longitude 84°05' W) in Kentucky. Two silvicultural treatments—(1) shelterwood with reserves ([10-15 ft²/ac] 2.3 – 3.4 m²/ha residual basal area to create a two-aged stand; and (2) thinning to the B-level of the Gingrich Stocking Chart (marking based on tree vigor and crown class)—and untreated controls were replicated thrice. Canopy cover was measured once at each of 20 permanent sample points located in each treatment unit with a digital plant canopy imager (CID Inc., Camas, WA).

PAR measurements (micromoles m⁻² s⁻¹), were also acquired with an Accupar Ceptometer (Decagon Devices, Pullman, WA) at each of these 20 points. During each measurement period, a quantum sensor and data logger (Li-Cor Inc., Lincoln, NE) was set up in a nearby opening to collect reference or “above canopy” measurements for subsequent calculation of percent full PAR within the stands.

RESULTS

PAR measurements in untreated controls were typically 10 percent of ambient PAR (total available PAR in the open). Maximum PAR levels were observed in the shelterwood with reserves treatment, typically 80 percent of ambient PAR. Mean PAR measurements in the intermediate intensity overstory removal treatment, Gingrich B-level thinning, were 35 percent of ambient PAR. Understory PAR increased significantly with each increasing level of overstory removal, as expected. Variability in PAR did not follow the same pattern. Variability in PAR was maximized by intermediate levels of cutting in the Gingrich B-level thinning.

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DISCUSSION AND CONCLUSIONS

Interpretation of PAR levels in treated stands should take into account the potentially wide ranges in PAR from point to point. The precise spatial arrangement of residual stems will determine the amount and distribution of PAR throughout the day. These factors will interact with the PAR requirements of regenerating species, and could substantially influence future stand composition.

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INFLUENCING WOODLAND MANAGEMENT USING WEB-BASED TECHNOLOGY

William R. Thomas and Jeffrey W. Stringer¹

Abstract.—The University of Kentucky, Department of Forestry Extension delivered hosted Web-based forestry educational programs (“webinars”) in 2009 to promote woodland management in Kentucky and engage county extension agents in forestry programming. These webinars were hosted by county extension agents and attended by woodland owners. This hosted webinar approach was intended to engage county agents in forestry programming, provide forestry programs for owners who manage their woodlands, reach non-managing woodland owners or those who may not attend traditional forestry field days, and use Web-based technology to deliver forestry educational programs. Participant responses indicated a high degree of acceptance of this programming approach with strong intentions to apply the knowledge gained. Increased extension agent involvement in forestry educational programming may prove the most valuable outcome of this programming effort.

INTRODUCTION

The woodlands of the central hardwood region, including Kentucky, are important ecologically and economically. Most of these woodlands are privately owned and their future depends largely on the decisions of private woodland owners, many of whom do not actively manage their property (NWOS Table Maker 2010). An important challenge for the forestry profession is to raise the awareness of forestry issues and promote woodland management among these woodland owners. Given the nature of shrinking outreach and extension budgets, coupled with the spread of high-speed internet connectivity in larger communities, Web-based technology may serve an important role in influencing private woodland owners to practice sustainable management. Raising woodland owners’ awareness of the opportunities and assistance available in managing their woodlands is an important first step.

HOSTED WEBINAR DEVELOPMENT

The University of Kentucky, Department of Forestry Extension developed 10 forestry Web-based programs for woodland owners in 2009. These webinars were organized into two forestry program series. One series focused on forest risks and risk management and offered three webinars; the other series focused on a variety of forestry topics and included seven webinars. All were hosted by extension agents at county extension offices and broadcast from the University of Kentucky, Department of Forestry. Most extension agents do not have a forestry background and have limited time to engage in forestry programs without support (Bardon 2001). This hosted webinar approach engages extension agents in forestry programming with minimal investments of their time. Some of the webinars included on-site foresters with the Kentucky Division of Forestry and prerecorded video segments while others were composed of live presentations only. Numerous practice sessions

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and frequent communications were maintained with hosting extension agents to ensure the best experience possible for agents and participating woodland owners. The presentations were sent to extension agents in advance and a conference call system was in place for contingency purposes. Considerably more planning and systems testing were required compared to traditional forestry programs.

EVALUATION RESULTS

Webinar attendees and hosting extension agents completed surveys to determine knowledge gained and acceptance of this educational delivery method. A large majority of the participants, 98.1 percent, indicated that webinars were an acceptable way to learn forestry information and 91.1 percent rated the webinars as “Good to Excellent.” More than 88 percent of participants indicated the information would help with management and work in their woodlands and 99.3 percent indicated they would use at least some of the information to manage their woodlands. Extension agents also returned evaluations and all indicated they would like to be involved in future forestry programming. The extension agents provided comments that complemented the woodland owners’ responses and feedback that can be used to adjust future webinars.

DISCUSSION

This educational programming effort allowed a better understanding of the role webinars can play in Kentucky and throughout the central hardwood region. Advantages to using hosted webinars include: alleviating some of the technology challenges that individuals may encounter, reducing the demand on personnel resources required to deliver individual programs, allowing forestry professionals to reach many audiences at once, and engaging extension agents in forestry programming. Technological advances are likely to improve forestry professionals’ ability to influence woodland management over the Internet; however, sending “hands-on” activities that can be completed before the program and having a local professional forester in attendance can be important enhancements to webinars. In addition, forestry professionals should not completely bypass nonformal educators. Extension agents and other nonformal educators can be an important part of forestry educational programming and efforts to engage them in forestry programs will likely multiply the impacts of forestry professionals in promoting woodland management.

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THE ECOLOGY AND SILVICULTURE OF OAKS, SECOND EDITION: A NEW BOOK

Paul S. Johnson, Stephen R. Shifley, and Robert Rogers¹

ABSTRACT

The second edition of *The Ecology and Silviculture of Oaks* was recently published (Johnson and others 2009). The approach of the book is fundamentally silvicultural, but the content is based on the premise that effective and environmentally sound management and protection of oak forests and associated landscapes must be grounded in ecological understanding. The central concepts are based on recognizing similarities and differences among the various kinds of oak forests. Consequently the underlying principles presented are applicable to many if not most of the oak forests of the world. Topics include the distribution of oaks and oak-dominated ecosystems; regeneration ecology; site productivity; development of natural stands and stand dynamics; even- and uneven-age silvicultural systems; silviculture for nontimber objectives; and growth and yield.

This new edition provides guidance for managing oak forests to meet a range of objectives, including timber production, oak savannas, acorn production, old-growth forests, and aesthetics. New and updated sections include information on carbon sequestration, the production of biomass, and impacts of climate change. Content on insect and disease problems of oak forests has been expanded to include sudden oak death and more material on oak decline.

This book is intended not so much as a how-to-do-it management manual as it is a source of ideas on how to think about oak forests as responsive ecosystems. The book is focused on oaks of the United States and was written for forest and wildlife managers, ecologists, silviculturists, environmentalists, students of those fields, and others interested in sustaining oak forests for their many tangible and intangible values.

Much has been written about the ecology and silviculture of oaks. The great quantity of relevant literature represents, in one sense, an informational “embarrassment of riches.” The embarrassment stems from the lack of synthesis within and across two broad fields of study: ecology and silviculture. Although the literature on North American oaks dates to the colonial period, most of it was written within the last 50 years. Much literature resides in relatively obscure scientific and technical journals, proceedings, government publications, and other sources that can be difficult to locate and retrieve. The first 16 proceedings of the Central Hardwood Forest Conference are important sources for much relevant knowledge. But even with ready access to this information, its synthesis into a holistic framework of knowledge is a daunting task.

One of the objectives of the book is to present ecological and silvicultural concepts that can be used to address an array of problems defined by various perceptions of how oak forests should be treated. The current trend

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in managing forests and forested landscapes is away from sustaining timber and other commodity outputs and toward a broader philosophy of sustaining desired ecological states. We believe that armed with a greater understanding of oak forests as responsive ecosystems, managers and conservators of oak forests will be better able to adapt to changing social values and to simultaneously build and act on co-evolving ecological and silvicultural knowledge.

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ASSESSING NATIVE SMALL MAMMALS' RESPONSES TO AN INCIPIENT INVASION OF BEECH BARK DISEASE THROUGH CHANGES IN SEED PRODUCTION OF AMERICAN BEECH

Justin N. Rosemier and Andrew J. Storer¹

INTRODUCTION

Exotic tree diseases have direct impacts on their host and may have indirect effects on native fauna that rely on host tree species. For example, American beech (*Fagus grandifolia* [Ehrh.]) is a dominant overstory component throughout its range and, like all tree species, is vulnerable to a broad array of insects and pathogens. These pests include beech bark disease (BBD), a disease complex consisting of an introduced scale insect (*Cryptococcus fagisuga* [Lind.]) and several species of native and introduced *Nectria* fungi. Due to the high levels of mortality sustained by American beech (50-85 percent in the areas of highest infection), it is likely that forests in the aftermath of BBD could differ greatly from pre-invasion beech forests. This abstract reports on small mammals' relative preference for European beech (*Fagus sylvatica* [L.]) and sugar maple (*Acer saccharum* [Marsh.]) seed, the direct impacts of BBD on seed production of American beech, and indirect impacts of this disease complex on native small mammals in the Upper Peninsula (UP) of Michigan.

METHODS

This project considered heavily infected forest stands (BBD and beech present; hereafter "Infected Beech"), forest stands containing healthy American beech (no BBD; hereafter "Healthy Beech"), and forest stands just outside the range of American beech (no beech or BBD; hereafter "No Beech"). Five study plots were established within each of the above stand types.

In 2005, a giving-up density (GUD) study was used to assess small mammal preferences for sugar maple and European beech seed. American beech seed was not used because this species was not available naturally or commercially at the time. However, we assumed that American and European beech are similar enough for the latter to be a valid substitute for the former. Subsequent lab-based analysis of seed characteristics and small mammal preference for these two species' seeds confirm the validity of this substitution.

Trays containing European beech and sugar maple seed were placed randomly within each study plot. Seeds were mixed with sand to provide foragers with diminishing returns the longer they remained at a food patch. Optimal foraging theory suggests that the longer a forager remains at a patch, the higher the costs it incurs. It follows that foragers should be willing to spend significant amounts of time only at high-quality food patches. Higher-quality food items (sugar maple or beech seed) should therefore be depleted to a lower GUD than the alternative. The small mammal portion of this study took place from 2003-2005. One 105-m x 105-m (8 traps x 8 traps with 15 m between each station) trapping grid was established on each of the study plots

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and was trapped repeatedly during each field season. Lincoln Peterson population estimates were determined based on the data collected. Comparisons of population estimates were made for the most common species among stand types: mice (*Peromyscus* spp.), eastern chipmunk (*Tamias striatus*), and southern red-backed vole (*Clethrionomys gapperi*). In addition, seed traps were placed randomly in each of the Healthy and Infected Beech study plots to provide annual estimates of seed production in these two stand types between 2003 and 2005.

RESULTS AND DISCUSSION

Results showed that seed predators in our study plots preferred European beech to sugar maple seed. While this result was not always statistically significant, it was consistent. Year-to-year effects were observed in small mammal population sizes, but we found relatively few significant differences among the stand types. The only exception was for southern red-backed voles, which were found in relatively high abundances in the No Beech stand type, occasionally in the Infected Beech stand type, and never in the Healthy Beech stand type. The mechanism behind this observation is unknown. Otherwise, it does not appear that mice species and eastern chipmunks have been negatively or positively affected by the incursion of beech bark disease into the UP. We were unable to obtain population estimates for other less common species.

Results of seed collections in the three stand types were inconsistent throughout this study. For example, in 2003, trees in the Infected Beech stands produced significantly more seed on average than trees in the Healthy Beech stands. One possible explanation for this observation is that trees in the Infected Beech stand type might have been producing a stress crop in response to the initial wave of infection by BBD. This pattern was reversed in 2004, when trees in the Healthy Beech stands produced significantly more seed than those in the Infected Beech stands. There are several possible explanations for this observation. First, it simply would have been difficult for trees to produce two large seed crops in sequential years. Second, trees in the Infected Beech stands might have begun to succumb to BBD, as infection often proceeds quickly. Although seed production was at a minimum in 2005, trees in the Infected Beech stands again produced significantly more seed than trees in the Healthy Beech stands.

It does not appear that small mammals have been strongly affected by the introduction of BBD into the UP, at least through changes in seed production by American beech. In part, this response may relate to these species' generalist tendencies; as one food source is diminished, they probably will find another. That said, there are other potential effects that could be significant (e.g., changes in light intensity in the understory as large overstory trees are lost, or increases in soft mast production), and a longer-term study is needed to get a more complete picture of changes that may be occurring in these forests.

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

OPTIMIZING WILDLIFE HABITAT QUALITY AND OAK REGENERATION IN BOTTOMLAND HARDWOODS USING MIDSTORY CONTROL AND PARTIAL HARVESTS

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ABSTRACT

Bottomland hardwoods can provide both wildlife habitat and timber. However, past high-grading practices limit future income potential and have resulted in undesirable species composition in many areas. Thus, prevalence of desirable oak species should be increased. Our study will attempt to determine the proper harvest level for bottomland hardwoods which will optimize wildlife habitat quality and conditions for oak regeneration. We will conduct partial harvests on six hardwood stands in minor stream bottoms in Mississippi. Each of the six study areas will consist of four 5-acre experimental units, in which three of the experimental units will receive a partial harvest with residual basal areas of 30, 50, or 70 square feet per acre with one unharvested control experimental unit. To determine the correct amount of trees to be harvested, overstory stems were inventoried using five inventory plots, 1/5th acre in size, within each experimental unit. Inventory of overstory stems consisted of species identification, total tree height, number of logs, and diameter at breast height (d.b.h.) measurements for each tree. To ensure that the correct amount of basal area was marked for removal from each experimental unit, a post-marking cruise was completed. Each experimental unit receiving a partial harvest received a midstory injection. Each midstory stem, except for oak species, received one hack per three inches d.b.h. and 0.03 fl. oz. of a 20 percent volume to volume Arsenal AC aqueous solution in each hack.

Utilizing deer exclosures and transects, we will inventory plant species during years one and two post-harvest and determine the crude protein levels of each species. Hundred-foot line transects will be used pre- and post-harvest to provide the following percentages: woody stems, herbaceous stems, and vines present. In addition, herbaceous and woody plants will be identified and recorded by species, size, and height class. Eight circular deer exclosures, 10.8 square feet in area, will be placed in the field after harvest, but prior to the growing season, within each experimental unit. Clippings will be taken inside the exclosures in July. The clippings will be dried in a forced-air oven to eliminate moisture. Forages will be identified by species and categorized accordingly into forbs, grasses, legumes, vines, woody species, and total forage. The amount of crude protein present in each species will be determined using a Kjeldahl procedure. Vegetation structure will be measured using a Nudds Density Board, by taking a reading in each cardinal direction from a randomized plot center. The reading will be recorded at 33 feet from an established plot center.

We will measure the amount of light that reaches the forest floor to determine the relationship between light availability and oak regeneration or plant species on our study areas valuable to white-tailed deer. Oak regeneration will be measured on five plots inside of each experimental unit. These plots will be centrally

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located inside an 80-foot buffer from the edge of each experimental unit to eliminate edge effect. The abundance of oak regeneration will be determined by counting all the stems inside 120, 1/100th-acre sample plots. The number of stems of non-oak woody regeneration will be counted and categorized based on species. Light availability will be measured at plot centers before and after harvest using a LI-COR light spectrophotometer quantum sensor (LI-COR Environmental, Lincoln, NE), which measures photosynthetically active radiation in the 400- to 700-nm waveband.

The experimental design for this project is a randomized complete block design blocked by study area. The experimental units are the treatment plots, and treatments are randomly assigned within each study site. We will test the null hypotheses that species richness, diversity, abundance of preferred forages, oak regeneration, crop tree growth, sunlight penetration, and nutritional carrying capacity do not differ among treatments using a repeated-measures mixed model analysis of variance. Nutritional carrying capacity for white-tailed deer will be compared among treatments using estimates of nutrient quality and biomass with a nutritional carrying capacity model. Overall habitat quality for white-tailed deer will be compared qualitatively among treatments.

ANTICIPATED OUTCOMES

Upon conclusion of the study, we will have established a preferred level of partial harvest for optimizing white-tailed deer forage, oak regeneration, and light availability. Our findings will allow us to develop and disseminate educational materials for optimizing oak regeneration and wildlife habitat. The materials will be used to improve sustainable forestry land management decisions in the Southeast.

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Includes 64 papers and 17 abstracts pertaining to research conducted on forest regeneration and propagation, forest products, ecology and forest dynamics, human dimensions and economics, forest biometrics and modeling, silviculture genetics, forest health and protection, and soil and mineral nutrition.

KEY WORDS: silviculture, regeneration, forest health, forest biometrics, forest products, forest ecology

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