



Forest Service

**Northeastern
Research Station**

General Technical
Report NE-316



Proceedings

14th Central Hardwood Forest Conference

March 16-19, 2004
Wooster, Ohio



14th

CENTRAL HARDWOOD FOREST CONFERENCE

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Published by:
USDA FOREST SERVICE
11 CAMPUS BLVD SUITE 200
NEWTOWN SQUARE PA 19073-3294

March 2004

For additional copies:
USDA Forest Service
Publications Distribution
359 Main Road
Delaware, OH 43015-8640
Fax: (740)368-0152

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14th

CENTRAL HARDWOOD FOREST CONFERENCE

Proceedings of a Conference held at the Ohio Agriculture Research and Development Center
(OARDC), The Ohio State University, Wooster, Ohio

March 16-19, 2004

Edited by:

Daniel Yaussy, U.S. Department of Agriculture (USDA) Forest Service
Dr. David M. Hix, The Ohio State University
Dr. P. Charles Goebel, The Ohio State University
Dr. Robert P. Long, USDA Forest Service

Sponsored by:

School of Natural Resources and the OARDC of The Ohio State University
and
USDA Forest Service, Northeastern Research Station

Proceedings published by:

USDA Forest Service, Northeastern Research Station

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Matthew Bumgardner, USDA Forest Service

FOREWORD

The Central Hardwoods Forest Conference is a biennial meeting previously hosted by universities and USDA Forest Service research stations in the Central Hardwood forest region in the eastern United States. 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. This, the 14th Conference, included presentations pertaining to forest health and protection, silviculture, forest biometrics and mensuration, harvesting and utilization, forests and wildlife management, fire in central hardwood forests, soils, hydrology and nutrient cycling, forest management, forest ecology, tree physiology and genetics, forest products and economics, forest stand dynamics, and field tours of forest ecosystems and state-of-the-art secondary manufacturing facilities in northeastern Ohio. The Conference consisted of 71 oral and 31 poster presentations resulting in the 66 peer-reviewed papers, 36 research notes and abstracts published here.

REVIEW PROCEDURES

Manuscripts from oral presentations were assigned to one of the editors and peer-reviewed by at least two professionals unless otherwise indicated. Reviews were returned to authors to revise their manuscripts and resubmitted electronically to the Northeastern Research Station, USDA Forest Service, for final editing and publishing. A proceedings editor or an associate editor reviewed the research notes and abstracts. Authors are responsible for the accuracy and content of their papers.

LIST OF TECHNICAL REVIEWERS

The editors sincerely appreciate the efforts of the following colleagues who reviewed one or more manuscripts. Their efforts greatly improved the content of these proceedings.

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Gary Graham	Gary Miller	Roger Williams
Thomas Hall	Arthur Morris	Thomas Wyse
	Rose-Marie Muzika	Dan Yaussy

CONTINUING FORESTRY EDUCATION

For attending the conference, each registrant was eligible for 15 hours of Continuing Forestry Education (CFE) credit through the Society of American Foresters.

ACKNOWLEDGMENTS

The Program Committee wishes to acknowledge the insightful plenary session messages from Bob Romig, Ohio Forestry Association; Mark Ford, USDA Forest Service; Brian McCarthy, Ohio University; and Dan Herms, OARDC. We would also like to express our appreciation to Dan Balsler, Mike Bowden, Matt Bumgardner, Charles Goebel, Gary Graham, David Hix, Bob Long, Bob Romig, Tom Schuler, Scott Stoleson, Jack Vimmerstedt, and Roger Williams for serving as session moderators. A special thank you goes to Julie Vujevich for providing the artwork for the notices and web site. Web site design, manuscript submission and retrieval, and registration were made possible by Dave Lohnes and Kurt Knebusch. Cheryl Capek and Bev Winer provided registration and logistic support. The field tours were led by Charles Goebel, David Hix, Kathryn Holmes, Lisa Petit, and Marie Semko-Duncan. Finally, we need to acknowledge the exceptional work by Rhonda Cobourn and Mary Boda who formatted the manuscripts and assembled the proceedings and electronic media.

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APPENDIX I: METRIC TO ENGLISH CONVERSIONS

Convert from:	To:	Multiply by:
bar (106 dynes/cm ²)	pound/inch ² (psi)	14.5
bar (bar)	atmosphere (atm)	0.9869
calorie	British thermal units (Btu)	0.00397
Centigrade (C)	Fahrenheit (F)	1.80C + 32
centimeter (cm)	inch (in)	0.394
centimeters, square (cm ²)	inches, square (in ²)	0.155
centimeter, cubic (cm ³)	inches, cubic (in ³)	0.061
gram (g)	pound (lb)	0.002205
gram (g)	ounces (oz)	0.0353 (for water density)
grams/liter (g/L)	percent (%)	0.100 (in water)
hectare (ha)	acre	2.471
kilogram (kg)	pound (lb)	2.205
kilogram/hectare (kg/ha)	pound/acre	0.893
kilometer (km)	mile (mi)	0.621
kilometer, square (km ²)	miles, square (mi ²)	0.386
kilometer, square (km ²)	acre	247.1
liter (L)	ounce (fluid oz)	33.8
liter (L)	quarts (qt)	1.057
liter (L)	feet, cubic (ft ³)	0.0353
liter/hectare (L/ha)	quarts/acre (qt/acre)	0.428
lux (Ix)	candle-foot ² (ft-c)	0.0929
meter (m)	yard (yd)	1.094
meter (m)	feet (ft)	3.281
meter ² (m ²)	feet ² (ft ²)	10.76
meter ² /hectare (m ² /ha)	feet ² /acre (ft ² /acre)	4.356 (basal area)
meter ³ (m ³)	feet ³ (ft ³)	35.31
meter ³ (m ³)	board feet, International 1/4"	287.4
meter ³ /second (m ³ /s)	gallons/second (gal/s)	264 (flow)
meter ³ /hectare (m ³ /ha)	feet ³ /acre (ft ³ /acre)	14.29
milligrams/liter (mg/L)	parts per million (ppm)	1.000 (in water)
millimeter (mm)	inch (in)	0.0394
ton (t) or megagram (Mg)	ton (U.S.)	1.102
ton/hectare (t/ha)	ton/acre (t/acre)	0.446

Yaussy, Daniel A.; Hix, David M.; Long, Robert P.; Goebel, P. Charles, eds.
2004. **Proceedings. 14th Central Hardwood Forest Conference**; 2004
March 16-19; Wooster, OH. Gen. Tech. Rep. NE-316. Newtown Square, PA:
U.S. Department of Agriculture. Forest Service. Northeastern Research
Station. 539 p. [CD-ROM].

Proceedings of the 14th Central Hardwood Forest conference held 16-19
March in Wooster Ohio. Includes 102 papers and abstracts dealing with
silviculture, wildlife, human dimensions, harvesting and utilization, physiology,
genetics, soils, nutrient cycling, and biometrics.

KEY WORDS: Central Hardwood, silviculture, oak-hickory, upland hardwoods,
bottomland hardwoods

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FIRST-YEAR EFFECTS OF *MICROSTEGIUM VIMINEUM* AND EARLY GROWING SEASON HERBIVORY ON PLANTED HIGH-QUALITY OAK (*QUERCUS* spp.) SEEDLINGS IN TENNESSEE

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ABSTRACT.—Continuing difficulty in the successful natural regeneration of economically important oak species (*Quercus* spp.) on highly productive sites has led to considerable research regarding the use of artificial oak regeneration to ensure recruitment of oak as an important component of future stands. Two obstacles to the success of some oak plantings in Tennessee are herbivory by white-tailed deer (*Odocoileus virginianus* (Boddaert)) and competition from an invasive non-native grass, Nepalese browntop (*Microstegium vimineum* (Trin.) A. Camus). We examined the effects of both deer herbivory and the Nepalese browntop on the first-year growth of outplanted, high quality, locally adapted, 1-0 northern red oak (*Q. rubra* L.) seedlings on the Ames Plantation in Tennessee. Northern red oak seedlings were outplanted under four overstory treatments (no cut, high grade, commercial clearcut, and two age). Seedling growth, deer browse pressure (defined as browse on terminal or lateral shoots), and herbaceous biomass were monitored monthly through the 2002 growing season. Browse pressure accounted for approximately 67 percent of the variation in total seedling height growth, and exhibited a strong negative relationship with total growth ($r = -0.82$; slope = -0.48). Additionally, Tukey-Kramer multiple comparison tests detected seedling height differences between the no-harvest treatment as compared with the three harvest treatments. Herbivory was prevalent early in the growing season when the seedlings first flushed, then declined during the remainder of the growing season. Analysis indicated that seedlings with a height of 148 cm or greater escaped browse. Thus, with larger seedlings planted, it may be expected that many seedlings will surpass this “browse line” in the first or second growing seasons. Linear regression revealed a strong negative relationship between Nepalese browntop biomass production and mean seedling height growth ($r = -0.74$; slope = -0.0046). No differences in Nepalese browntop biomass production were found between treatments ($P = 0.29$) with first year results, but means ranged from 45 percent in the two-age treatment to 23 percent in the no cut treatment. This range may broaden with additional time and significant differences arise. These results still suggest that canopy disturbance may encourage a growth flush of this competitive species.

Introduction

The problem of successful oak (*Quercus* spp.) regeneration on higher quality sites (S.I. ≥ 65 , base age 50) has attracted considerable attention and generated numerous research papers over past decades. However, the successional replacement of oak as a major forest component persists as a major concern in eastern deciduous forests, particularly on mesic or highly productive hardwood sites of the Central Hardwood Region (CHR). Studies indicate that promoting abundant large oak advance reproduction prior to overstory removal will aid in successfully regenerating oak on these sites (Sander 1971, Loftis 1982, 1990). However, the altered disturbance regimes of the CHR combined with short-term economic factors that constrain management options do not often produce conditions favorable to the development of such advanced reproduction. Therefore, some researchers have shifted focus to a post-harvest approach utilizing artificial means to maintain oaks on these sites.

Over the past ten years, a number of research projects have attempted to use nursery-grown seedlings to enhance post-harvest oak composition with varying results (Lorimer 1994, Demchik and Sharpe 1999, Buckley 2001, Spetch et al. 2002). Deer herbivory (Buckley et al. 1998), herbaceous competition including non-native species (Dubois et al. 2000) and competition from other woody species has hindered success. Through genetic selection, use of proper seed source, and optimal nursery techniques, high-quality

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artificial oak reproduction has been developed (Kormanik et al. 1994a, 1994b, Schlarbaum et al. 1998) to address these matters. Although initial growth results appear positive (Oswalt et al. 2003), deer herbivory and competition by nonnative vegetation still present problems. Large deer populations throughout the eastern deciduous forest continue to affect planted oak seedlings (Demchik and Sharpe 1999, Ward et al. 2000, Buckley 2001, Romagosa and Robison 2003). In addition, an invasive Asian C4 grass, Nepalese browntop (*Microstegium vimineum* (Trin.) A. Camus.), has been identified as a potentially severe problem on productive sites in the southeast (Barden 1987, Simberloff 2000, Romagosa and Robison 2003).

This study investigates the potential impacts of early season deer herbivory on the first-year growth of high-quality northern red oak (*Q. rubra* L.) seedlings developed by the University of Tennessee Tree Improvement Program, outplanted within four overstory treatments. In addition, we explore the relationships among nonnative invasive Nepalese browntop biomass, other herbaceous biomass, and first-year planted seedling height growth.

Materials and Methods

Study Site

The study site is located along an intermittent stream in the headwaters region of the North Fork of the Wolf River (NFWR) in southwest Tennessee (35°09'N, 89°13'W) on the Ames Plantation. The site encompasses approximately 100 acres of mixed bottomland and riparian hardwood forest dominated by various oak species. Two distinct landforms were identified within the study site: a minor bottom, infrequently impacted by flooding, near the confluence of the stream with the NFWR and ancestral terraces of the minor stream (Hodges 1997).

The headwaters region of the NFWR is located within the Mississippi Embayment of the Gulf Coastal Plain. The geology is dominated by the highly erodible Wilcox and Claiborne formations overlain by loess deposits common in western Tennessee (Fenneman 1938). The principal soil groups are Grenada-Loring-Memphis on the terraces and Falaya-Waverly-Collins within the minor bottom (USDA 1964). Average site index, base age 50 yr, was estimated to be 75 for oaks, 85 for yellow-poplar (*Liriodendron tulipifera* L.) and 70 for sweetgum (*Liquidambar styraciflua* L.) on both sites (Carmean et al. 1989). Average age for the dominant and co-dominant stems across the study site was 70 yr.

Study Design

In the fall of 2001, three experimental blocks were identified based on differences in average basal area ($p < 0.05$), landform and position. Twelve 2-ac treatment units were designated within the experimental blocks, four units located within the minor bottom (bottom block) and eight units located within the terrace sites upstream from the minor bottom (four each within the east and west blocks). Species composition was dominated by cherrybark oak (*Q. falcata* var. *pagodifolia* Ell.), yellow-poplar and sweetgum in the bottom block and several species of red and white oaks on the ancestral terraces.

Four overstory treatments, including a control (no harvest), with 3 replications were randomly assigned to the 12 units using a randomized complete block (RCB) design. Harvesting for all treatments was completed in the winter of 2001-2002. Overstory treatments are described below.

- Commercial clearcut: defined by the removal of all stems greater than 6 in. diameter breast height (DBH). This treatment is designed to represent a common practice on industrial forestland.
- High-grade: standard diameter limit harvest where all stems greater than 14 in. DBH are removed. This treatment is designed to represent a common and persistent practice on NIPF lands.
- Two-age: residual stand basal area of 15-20 ft²/ac targeted. Residual stems were chosen based on spacing criteria and the desire to leave stems of desirable species with an opportunity to increase in value. Desirable species included oaks, hickories (*Carya* spp.) and yellow-poplar.
- No-cut: no harvesting; is designed to act as the study control.

Seedlings

Northern red oak seedlings originating from acorns collected from a seed orchard on the Ames Plantation (Schlarbaum et al. 1998) were chosen for planting following harvest. The seedlings were grown at the Georgia Forestry Commission's Flint River Nursery under fertilization and irrigation protocols developed by Kormanik et al. (1994a). The seedlings were lifted in February 2002 and were graded using procedures developed by Kormanik et al. (1994a, 1994b), as modified by Clark et al. (2000). The seedlings were measured for height and root collar diameter (rcd) growth and the number of first-order lateral roots (folr) (Ruehle and Kormanik 1986) were counted. Seedlings were visually graded based on initial height, rcd and number of folr. Approximately 50 percent of the seedlings were culled. Mean initial rcd, initial shoot height and number of folr for planted seedlings (n=720) were 1.14 cm (0.45 in.), 114 cm (3.74 ft) and 19 respectively. Sixty seedlings were planted by shovel at a 6.1 by 6.1 m (20 by 20 ft.) spacing in March 2002 within each of the 12 units for a total of 720 seedlings.

The planted seedlings were monitored monthly (35-45 days) throughout the growing season for four periods (beginning = 5/02, early = 6/02, mid = 7/02 and late = 8/02). Seedling mortality, defined as lack of green tissue along the primary stem, and herbivory were monitored for all seedlings in all four periods. End-of-season mortality and shoot growth data were obtained in January 2003 for all seedlings after the onset of dormancy.

Herbivory

In order to investigate the effects of deer herbivory, a "browse pressure classification" was used (Buckley 2001). Browse pressure was classified into one of 4 categories; no browse, terminal browse, lateral browse and complete browse. No browse was defined by no visible signs of herbivory, lateral browse was defined as herbivory limited to lateral shoots only, terminal browse was herbivory limited to only the terminal shoot and complete browse was defined as observed herbivory on both lateral and terminal shoots.

Nepalese browntop

During pre-harvest sampling, Nepalese browntop was observed in all units and identified as a possible future problem. Therefore, the response, after overstory disturbance, of understory biomass production was also measured, focusing on Nepalese browntop. Five randomly placed 0.46 m² samples were collected in the four sample periods during the growing season and during end-of-season data collection, within each unit for a total of 60 samples per sample period. All material was clipped at ground level, divided into 12 categories (Table 1), dried and weighed according to Mueller-Dombois and Ellenberg (1974). Woody species were removed from all analyses.

Table 1.—Mean biomass (kg/ac) by category for the 12 treatment units during the 2002 growing season for the oak regeneration study on the Ames Plantation, Fayette County, Tennessee.

Biomass Category	Mean Biomass (kg/ac)
Poison ivy	19.78
Japanese honeysuckle	73.46
Forbs (18+ spp.)	634.73
Smartweeds	52.95
Semi-woody plants	5.42
Ferns and fern allies	7.44
Sedges	42.08
Nepalese browntop	670.13
Woody shrubs and seedlings	43.64
Vines	45.09
Grasses	67.42
Rushes	14.85

Analyses

Statistical analyses were conducted using both SAS (SAS Institute Inc. 1989) and NCSS software (Hintze 2001). Mixed-model analysis of variance and Tukey-Kramer multiple comparison tests were used to discern differences in height growth, herbaceous biomass production and browse pressure among treatments. Simple linear regression was used to determine possible relationships between early growing season herbivory (calculated as percent of seedlings incurring any browse in each unit) and mean end-of-season seedling height growth with and without the control data. The category “any browse” includes pooled data from the complete, terminal and lateral browse categories. Logistic regression techniques, using pooled data of terminal browse and complete browse, referred to as terminal shoot removal, and chi-square were used to explore the possible effect of initial seedling height on observed patterns of herbivory. Stepwise variable selection and multiple regression analyses were used to test associations between mean seedling height growth and end-of-season herbaceous biomass production of the twelve different biomass categories and to select the most influential variables. Independent simple linear regression analyses were used to further explore the strongest associations with and without the control data. Statistical significance for all analyses were selected at the $\alpha = 0.05$ level.

Results

Seedling Growth and Survival

First-year mortality was greatest for the no-cut treatment with mean percent mortality of 33 percent or 59 seedlings ($n = 180$, $p = 0.004$) followed by the high-grade and two-age treatments with 5 percent each. The commercial clearcut experienced no seedling mortality.

Eighty-one percent of all seedlings experienced only one growth flush. First-year end-of-season height growth was greater for the no-cut treatment ($P = 0.0005$). Mean total growth was observed as 12.43 cm, 13.48 cm, 9.18 cm and 22.33 cm for the commercial clearcut, high-grade, two-age and no-cut treatments, respectively (Table 2). Tukey-Kramer multiple comparison tests resulted in no differences between the three harvest treatments. No significant interactions were indicated.

Table 2.—Mean initial height, average height growth for each sampling period and mean total height growth (cm) of 1-0 high-quality northern red oak (*Quercus rubra* L.) seedlings for each sampling period during the 2002 growing season for each treatment and block for the oak regeneration study on the Ames Plantation, Fayette County, Tennessee.

Treatment	Growing Season						
	Initial ¹	Early	Mid	Late	Total		
Commercial clearcut	110.49 (180) ²	12.40 A ⁴ (89)	3.17 A (41)	0.92 (43)	12.43 A (180)		
High-grade	108.78 (180)	11.46 A (91)	3.42 A (30)	6.14 (33)	13.48 A (171)		
Two-age	109.61 (180)	8.73 A (92)	0.00 B (27)	-2.79 (28)	9.18 A (170)		
Control (no-cut)	110.70 (180)	26.21 B (90)	-0.18 B (20)	-0.20 (30)	22.33 B (121)		
	$P = 0.89$ ³	$P < 0.001$	$P < 0.01$	$P = 0.52$	$P < 0.001$		
Block							
East	109.69 (240)	16.68 (119)	2.87 (38)	1.00 (40)	17.72 (222)		
West	110.26 (240)	15.60 (121)	1.64 (57)	0.58 (65)	16.35 (213)		
Bottom	109.73 (240)	11.82 (122)	0.29 (23)	-7.74 (29)	9.00 (207)		
	$P = 0.92$	$P < 0.04$	$P = 0.85$	$P = 0.98$	$P < 0.001$		

¹ Sample Dates – Initial = 2/08/02, Early = 6/10/02, Mid = 7/14/02, Late = 8/25/02, End = 1/04/03

² Sample size (n), 50% of seedlings were randomly measured at each period during the growing season with n representing the number of actual seedlings in the sample cross-referenced directly from previous sample. Total height growth was measured at the end of the growing season for all seedlings.

³ Mixed model ANOVA results

⁴ Mean separation by Tukey-Kramer multiple comparison tests. Means followed by the same letter are not significantly different at the alpha 0.05 level.

Herbivory

Herbivory was concentrated in the early part of the growing season and was not recurrent. No additional browse was observed after the May observation date. Observed browse pressure was concentrated within the harvest treatments. The three harvest treatments experienced similar pressure with 32 percent, 33 percent and 27 percent of seedlings being completely browsed for the two-age, high-grade and commercial clearcut treatments, respectively. Browse pressure was least severe for the no-cut treatments with 77 percent of seedlings experiencing no browse and only 13 percent being completely browsed. Browse levels for lateral bud browse only and terminal bud browse only were similar among all treatments. Analysis of the pooled dataset (any browse) suggested differences in observed seedling browse pressure among treatments ($P < 0.02$) primarily due to the lower amount of browse pressure experienced by the no cut treatment. No differences in browse pressure were observed between the three harvest treatments. When examined independently of all other variables and across all treatments, browse pressure (any browse) accounted for approximately 67 percent of the variation in total seedling height growth ($R^2 = 0.6720$, $P = 0.011$). Browse pressure exhibited a strong negative relationship ($r = -0.82$) with total growth. Because seedling growth differences occurred between the no-cut treatment and the three harvest treatments, and no differences were found among the harvest treatments, the data was reanalyzed without the controls. The reduced data model resulted in a slightly weaker fit ($R^2 = 0.64$), although the new model was still significant ($P < 0.01$) and a strong association was still observed ($r = -0.80$). A logistic regression model using initial seedling height indicated a relationship with terminal shoot removal ($P < 0.001$). In an attempt to identify “browse height”, chi-square was used to test 4 categories of initial height and terminal shoot removal. No seedling with an initial height greater than 148 cm experienced terminal shoot removal ($P < 0.001$).

Nepalese browntop

Nepalese browntop was the dominant herbaceous vegetation in most plots at the end of the growing season. Analysis of the 12 biomass categories, through stepwise variable selection and multiple regression across all treatments, revealed that Nepalese browntop appeared to be of greatest importance (Table 3). The overall model was significant ($R^2 = 0.97$, $P < 0.001$), but model fit was reduced by 83 percent with the removal of the Nepalese browntop term (Table 3). Therefore, simple linear regression techniques were used to explore the potential relationship between Nepalese browntop production and mean seedling height growth within individual units. Although a strongly negative relationship was found, Nepalese browntop biomass appeared to be a moderate to weak predictor of end-of-season seedling height growth ($R^2 = 0.63$, $P = 0.003$, $r = -0.80$).

Interestingly, with the removal of Nepalese browntop from the analysis, the relationship between herbaceous biomass production and mean seedling height growth did not exist. Simple linear regression of mean end-of-season height growth and mean herbaceous biomass production without Nepalese browntop

Table 3.—Reduction of R^2 from multiple regression of mean biomass (selected significant biomass terms from stepwise regression - kg/ac) and mean seedling height growth (cm) for the 12 individual treatment units for the oak regeneration study on the Ames Plantation, Fayette County, Tennessee.

Term	R^2	Sum of Squares	P
Full model	0.97	2473.08	0.0009
Error	0.03	12.94	
Smartweeds	0.19 ¹	90.04	0.002
Sedges	0.24	113.01	0.0012
Nepalese browntop	0.83	394.28	0.0001
Woody shrubs and seedlings	0.18	86.85	0.0022
Grasses	0.05	22.97	0.0308
Poison ivy	0.11	54.29	0.0059

¹ The reduction in R^2 when the term is removed from the model.

Example: when the Nepalese browntop term is removed, the remaining terms explain approximately 14 percent of the variation in seedling height growth.

resulted in no association ($R^2 = 0.008$, $r = -0.09$, $P = 0.78$). Although no differences in Nepalese browntop production occurred among treatments ($P = 0.10$), mean biomass production of Nepalese browntop ranged from 1422 kg/ac in the two-age treatment followed by the commercial clearcut, high-grade and no-cut treatments with 888 kg/ac, 324 kg/ac and 47 kg/ac (Figure 1), respectively. When the data were reanalyzed without controls, the reduced data model resulted in a stronger fit ($R^2 = 0.75$, $P < 0.003$), and an overall stronger association ($r = -0.87$). Analysis of all herbaceous material resulted in similar associations as the Nepalese browntop- only analysis. Total end-of-season herbaceous biomass production and end-of-season (total) height growth were significantly negatively related among treatments ($r = -0.74$, $P = 0.006$, $R^2 = 0.55$). Mean end-of-season herbaceous biomass production differed across treatments ($P = 0.003$).

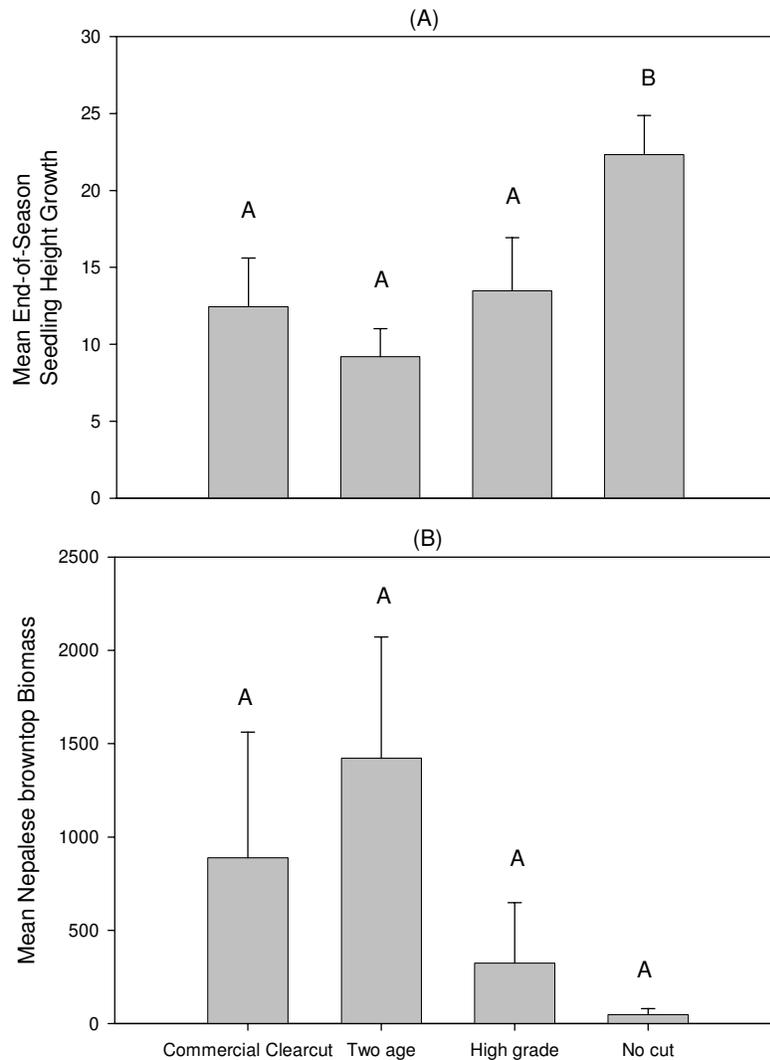


Figure 1.—Mean end-of-season height growth (cm) of 1-0 high-quality northern red oak (*Quercus rubra* L.) seedlings (a) and mean end-of-season Nepalese browntop biomass (kg/ac) (b) for each treatment for the oak regeneration study on the Ames Plantation, Fayette County, TN. Mean separation by Tukey-Kramer multiple comparison tests. Means followed by the same letter are not significantly different at the alpha 0.05 level.

Discussion and Conclusions

Browse Pressure

In this study, browse pressure, when examined independently, significantly impacted first-year seedling shoot growth, accounting for 67 percent of variation. However, one of the key benefits in using large, high-quality seedlings with a greater initial height at the time of outplanting is the capability of these seedlings to exceed browse height rapidly, escaping damage from herbivory. Conventionally, “browse lines” have been loosely defined at approximately 137 cm (4.5 ft). Results from this study indicate the browse line for this particular site is at 148 cm (4.9 ft), slightly greater. Additionally, results from logistic regression analysis suggest that a strong relationship exists between initial planting height and browse pressure, further indicating that larger seedlings may be more likely to escape browse.

After the first growing season, heights of many seedlings have already approached or surpassed the “local browse line”. If second year growth is similar to the first year, it is reasonable to suggest that a majority of seedlings will have surpassed this limiting height. Therefore, high-quality seedlings like those used in this study have a greater initial advantage over commonly available seedling stock due to the larger initial planting height.

For this study, seedlings were planted on 6.1 by 6.1 m (20 by 20 ft) spacing. Planting on such a wide spacing was consistent with enrichment plantings and was not an attempt to establish an oak plantation. Such a wide spacing was hypothesized to reduce seedling apparency and minimize herbivory. Although browse levels were greater early in the growing season, herbivory on all seedlings decreased with the onset of herbaceous vegetation growth. The measured herbaceous biomass for the mid growing season increased by approximately 300 percent from the early growing season. This marked increase in herbaceous material occurred concomitantly with a cessation of browse pressure. No detectable shoot browse was observed following the first observation period (beginning) while clear signs of herbivory were observed on surrounding herbaceous material. Therefore, the flush of herbaceous material either provided protection through concealment or offered an alternative food source for herbivores. Further empirical investigations are needed in addition to this correlative study to address the question of spacing influences on seedling apparency.

Herbaceous Biomass Production

Although herbaceous growth appeared to offer seedling protection from herbivory, herbaceous biomass production appeared to have a significant competitive effect on the development of the oak seedlings used in this study. When examined independently, herbaceous biomass production accounted for approximately 55 percent of the variation in end-of-season height growth. Herbaceous biomass production and seedling growth were also negatively correlated.

Both the stepwise variable selection and multiple regressions suggested that one particular herbaceous species had a greater influence than all other herbaceous material. Nepalese browntop, an introduced invasive, appeared to influence seedling development and at times comprised 50 percent or more of the herbaceous material sampled. Nepalese browntop, when analyzed independently, accounted for the same amount of variation as total herbaceous biomass. When Nepalese browntop was removed from the analysis, all other herbaceous material accounted for less than 1 percent of the variation in mean seedling height growth. Therefore, the results suggest that Nepalese browntop was the overwhelming competitive influence, in terms of herbaceous biomass, in this study during the first growing season. The post-harvest release of this non-native grass appeared explosive due to the species' ability to completely overwhelm the invaded site. In this study, although no significant relationship between Nepalese browntop biomass production and overstory treatment was observed, Nepalese browntop appeared to be affecting seedling growth. In treatment units where the greatest Nepalese browntop response, in terms of biomass accumulation, were observed, the lowest seedling growth was also observed. For example, the two age treatment units experienced the largest Nepalese browntop explosion and the least amount of first-year seedling growth. This suggests that Nepalese browntop could be having a significant impact and due to its growth strategies could continue to impact both artificial and natural regeneration within the study.

Trends and associations observed in the first year of this study have potential implications. It appears that seedlings herbivory by white-tail deer may be overcome by the planting of taller seedlings. Additionally, the observed relationship between seedling growth and Nepalese browntop may persist, further retarding seedling development. However, it is important to note that this is first-year data, yet the first year after planting can be viewed as very critical in establishing artificial reproduction.

Low first-year mortality and the ability to surpass browse levels were observed with seedlings in this study. As a result, these data can be seen as positive. Yet, further empirical investigations are needed to clarify the impacts of both non-native Nepalese browntop and deer herbivory on planted oak seedlings in the CHR.

Acknowledgment

The authors thank the USDA Initiative for Future Agriculture and Food Systems and the Hobart Ames Foundation for their financial support of this research. The authors also acknowledge the assistance of two anonymous reviewers whose suggestions greatly improved the manuscript.

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OAK SEED PRODUCTION, WEEVIL (COLEOPTERA: CURCULIONIDAE) POPULATIONS, AND PREDATION RATES IN MIXED-OAK FORESTS OF SOUTHEAST OHIO

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ABSTRACT.—Oaks have dominated much of the central hardwood forest region for thousands of years. However, there have been geographically widespread reports of its failure to regenerate on many types of sites for many decades. Most studies have approached the oak regeneration problem through the examination of direct effects. There are many indirect ecological effects that have not been well studied. Anecdotal evidence suggests that Native Americans may have used fire for many reasons including control of insect predators and/or pathogens. We chose to examine how fire and/or thinning may *directly* influence oak seed production and how those treatments may *indirectly* impact seed quality by influencing the major species of pre-dispersal oak seed predators (curculionid weevils). Weevils have been known to destroy > 90% of a typical acorn crop. Two mixed-oak forests were selected for study in southeastern Ohio. Within each forest, four stands were selected and randomly assigned a control and a treatment: prescribed fire, thinned, thinned & followed by fire. Thinning treatments were executed during the fall-winter of 2000-2001, followed by controlled burns in the spring of 2001. After the treatments were executed, we selected two species for study: black oak (*Quercus velutina*) and chestnut oak (*Q. prinus*). We selected nine trees of each species (144 trees total) within each unit and erected two 0.25 m² seed traps beneath each tree. Traps were sampled monthly throughout the growing season and for a period after leaf drop (August-December) during 2001 and 2002 to collect seeds. Acorns were destructively sampled and scored as sound, aborted, weevil-depredated, or other. Treatments did not appear to strongly influence seed production or predation rates in the first growing season (floral buds had already initiated and treatment effect was likely minimal). However, during the second season, treatment influence was dramatic. Both oak species responded by producing considerably more seeds in the combined thin-burn units. Any unit receiving a burn also produced a significantly greater number of sound acorns (weevil depredation was lowest in burn units). Long-term studies are continuing to help work out the details of how silvicultural treatments may influence periodic seed-producing (masting) species such as oaks.

Based on stand composition and structure, numerous studies have concluded that there is the potential for large-scale conversion of mixed oak forests in the central Appalachians to domination by more mesic species. In stands where the overstory is dominated by oaks and hickories, the understory is often dominated by more shade tolerant maples and beech (Lorimer 1984, McCarthy and others 1987, Crow 1988). In addition, recent studies have begun to document the decline of oak in the overstory. The cause of decline in the overstory appears to be multi-causal and is likely the result of combined factors such as drought, acid deposition, atmospheric pollution, two-lined chestnut borer, and *Armillaria* fungi (Millers and others 1989, LeBlanc 1998). The seriousness of the overall problem has been recognized for almost two decades and examined in considerable detail (Loftis and McGee 1993); however, few immediate solutions have been forthcoming and many details are still not well understood.

The failure of oak to regenerate naturally has prompted an interest in the role of prescribed fire as a silvicultural tool. The suppression of fire for the past 60-80 years is correlated to the period of oak regeneration failure (Loftis and McGee 1993). The role of fire is still not well understood and there may be a variety of direct and indirect effects. Moreover, during this same time frame, there have been dramatic

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shifts in wildlife populations (e.g., deer, turkey, and squirrel). The direct role of wildlife in the hardwood regeneration has been fairly well studied. Deer can have an enormous negative impact when their densities are sufficiently high (Harlow and others 1975, Marquis and others 1976, Tilghman 1989, Gill 1992, Healy and others 1997, Horsley and others 2003). Rodents may have a remarkable effect on pre- and post-dispersal seed mortality, often > 90% (Barnett 1977, Sork 1983, McCarthy 1994). And likewise, insects may negatively influence oak regeneration via pre- and post-dispersal seed depredation (Gibson 1972, Galford et al. 1991).

The relationship between fire, wildlife, and oak regeneration has received considerably less attention. With the exception of one study by Wright (1986) we are unaware of any other work that has experimentally evaluated the role of prescribed fire in controlling insect populations in mixed oak forests with the goal of increasing oak regeneration. While Galford et al. (1991) found that there were a number of acorn predators including acorn weevils (*Conotrachelus posticactus*), nitidulids (*Stelidota octomaculata*), and acorn moths (*Valentia glandulella*) that destroy great numbers of acorns in Pennsylvania oak stands, we concentrate here on the role of acorn weevils as they have been observed (BC McCarthy, *personal observation*) as being particularly problematic in southeastern Ohio mixed oak stands.

Several studies have examined the masting nature of seed production in oaks and hickories (Collins 1961, Christisen and Kearby 1984, Sork and Bramble 1993, Healy and others 1999, Greenberg 2000, Kelly and Sork 2002). Factors that influence mast production in oaks include weather during anthesis (Sharp and Chisman 1961, Cecich and Sullivan 1999), tree ontogeny (Greenberg 2000), genetics and site quality (Wolgast 1978, McCarthy and Quinn 1989), weather during maturation (Sharp and Sprague 1967), and predators (Gibson 1972, Short 1976, Barnett 1977, Lewis 1980, Gibson 1982, Weckerly and others 1989, Galford and others 1991). The effects of management strategies, e.g., mechanical thinning and prescribed fire, on seed production are not as clear. Healy and others (1999) suggest seed production is influenced more by individual tree and annual variation than by thinning. Additionally, the increased acorn production in thinned areas was a short term response and production returned to unthinned levels in ca. 6 years.

Historically, fire has been an integral component of the disturbance regime of many central Appalachian mixed-oak forests (Rouse 1986, Abrams 1992, Sutherland 1997, Brose and Van Lear 1998). Forest fire suppression since the turn of the century has changed the natural disturbance regime and may very well turn out to play a pivotal role in oak regeneration and forest conversion. What is less clear is whether fire plays a direct role or an indirect role in oak regeneration. Certainly, wildland fires open up the midstory, allow more light penetration, produce a flush of nitrogen in the system, and reduce herbaceous competition. Several large studies have been launched in Missouri (MOFEP) and Ohio (USFS) to better understand the role of fire in oak regeneration. The role of fire may involve a variety of direct and indirect effects.

We hypothesize that the role of fire in oak regeneration may be strongly tied to acorn weevil population dynamics and a concomitant increase in the availability of viable seed. Weevils have been found to kill up to 90% of oak (Barnett 1977, Galford et al. 1991, Gribko 1995) and hickory (McCarthy and Quinn 1989, McCarthy 1994) seed crops and have been implicated as a primary cause of oak regeneration failure (Marquis and others 1976, Weckerly and others 1989) in certain forests. The *Curculio* weevil life-cycle begins while the acorn is developing on the tree. The adult weevils emerge from late spring to early summer, after pollination has occurred. Females drill a hole into the developing nutmeat and eggs are deposited into a channel (Gibson 1969). The larvae consume the remaining nutmeat generally before the acorn detaches from the tree (Marquis and others 1976). Full-grown larvae hatch in 5 to 14 days (Brooks 1910). The *Curculio* larvae emerge from the acorn after it has fallen and will burrow up to 20 cm into the soil to over-winter. The adults emerge the following year and the process repeats. Adult *Conotrachelus* weevils tend to emerge later and will not oviposit through an acorn shell (Gibson 1964). Oviposition occurs in damaged, cracked, sprouted or previously infested acorns. Interestingly, the prime time for forest fires in southeastern Ohio is the autumn and spring—the same time that weevil larvae are emerging from the nuts (autumn) or from the soil as adults (spring). Fires during this time would result in major population declines for the weevil and decreased nut predation during fruit ripening in the summer and fall. Thus, the role of fire may be in

controlling the major seed predator of oak. If we are to manage oak forests for maximum wildlife abundance, we must begin to understand the relationship between fire, weevils (and other arthropods), and oak mast.

The objectives of this research are to examine the role of fire and thinning with respect to acorn production and to *Curculio* and *Conotrachelus* populations and survival. Specifically, we wish to test several hypotheses. To test the effects of silvicultural treatments on acorn production the null and several alternative hypotheses are:

H₁₀: Silvicultural treatments do not affect oak seed production.

H_{1A}: Thinning treatment will increase production vs. the control due to increased light.

H_{1B}: Burning treatment will increase production vs. the control due to increased nutrients.

H_{1C}: Thinning and burning combined will produce a greater effect than either thinning or burning alone.

To examine factors affecting the variability of acorn production and the silvicultural treatments, the null and alternative hypotheses are:

H₂₀: Reproduction among trees will be more or less uniform due to population masting habit.

H_{2A}: Considerable variability among trees in seed production because of genetics or differences in moisture levels (Integrated Moisture Index: IMI (Iverson and others 1997)).

H_{2B}: Variability will be greater in treatments vs. the control due to the interaction of genetics and patch effect.

To examine the effect of treatments on weevil acorn depredation we tested the hypotheses that:

H₃₀: Weevil predator abundance will be similar among silvicultural treatments.

H_{3A}: Weevil abundance will be lower in burn treatments due to mortality incurred by weevils.

Finally, we wanted to test the effect of treatments on viable acorn production and the hypotheses that:

H₄₀: Proportion of sound seed produced will be similar among treatment applications.

H_{4A}: Proportion of sound seed will be greater in treatment units because of increased resources.

H_{4B}: Proportion of sound seed will be greater in burn treatments because of fewer weevils.

Study Areas

This study was conducted in Vinton County, OH in the mixed-oak forests of the Raccoon Ecological Management Area (REMA: Vinton Furnace Experimental Forest) and Zaleski State Forest (ZAL). The region lies within the unglaciated Allegheny Plateau physiographic province (Fenneman 1938) of Ohio and has been designated as part of the Mixed Mesophytic Forest Region by Braun (1950). Southeast Ohio exhibits a humid continental climate with mean yearly temperatures of 12.2°C (NOAA 2003). July is the warmest month (mean 23.7°C), and January is the coldest (mean -0.5°C). Mean annual precipitation is 106.7 cm with October being the driest month and July the wettest month.

Methods

The study areas are part of a landscape-scale ecosystem restoration project, Fire and Fire Surrogate Study (FFS), using mechanical thinning and prescribed fire as management tools to restore ecosystem structure, function and composition. Within each forest, four stands ca. 20 ha each were selected and were randomly assigned into three treatments and a control (control unit). The treatments consisted of a shelterwood thin aimed at reducing basal area by ca. 30% (thin), prescribed fire (burn) and a combination thin and prescribed fire (thin & burn).

Seed Traps

To assess the level of seed predation and to evaluate treatment effect on acorn production, 288 seed traps (36 traps in each unit) were placed under mast trees of black (*Quercus velutina* Lam.) oak (BO) and

Table 1.—Number of chestnut oak and black oak acorns collected during 2001 and 2002 at two forests in southeast Ohio. Category 1: sound & filled; 2: unfilled/aborted (immature & small); 3: depredated (weevil or exit hole present); and 4: other (e.g., fungi).

Year	Species	Category				Total
		1	2	3	4	
2001	Chestnut oak	1	3	10	3	17
	Black oak	1029	704	2160	64	3957
2002	Chestnut oak	332	680	267	47	1326
	Black oak	57	958	553	68	1636

chestnut (*Q. prinus* L.) oak (CO). Two 0.25 m² conical seed traps were placed ca. 1.5 m aboveground under each oak tree and every four weeks, the seeds were collected and returned to the laboratory. Seed traps were installed in July and mid-month sampling occurred from August–December during 2001 and 2002. Seeds were identified and scored as (1) filled and sound, (2) unfilled/aborted (immature and small), (3) depredated (weevil or exit hole present), and (4) other (e.g., fungi).

Emergence Traps

To elucidate the phenology of weevil populations, several methods were employed during 2001 and 2002 to capture weevils as they emerged from the soil. First, 10 white oak (*Q. alba*) and 10 black oak (*Q. velutina*) in each unit (160 trees total) were selected and a 15-20 cm ring of Tanglefoot™ was applied to each bole. Trees were monitored at weekly intervals to see if weevils were trapped in the Tanglefoot™. Second, emergence traps were built (Raney and Eikenbary 1969) and placed under 3 white oak and 3 black oak trees in each unit for a total of 48 traps. To avoid confounding effects, different trees were used for each study. Emergence traps were placed in the field in late February 2002 and checked on a weekly basis through June 2002. Captured weevils were returned to the laboratory and identified.

Statistical Analysis

Mixed model analysis of variance (ANOVA) with orthogonal contrasts (Hintze 1997) was used to test for treatment effects on acorn production. Orthogonal contrasts included the control vs. thin, control vs. burn, and thin and burn combination vs. thin and burn only, respectively. Log-linear analysis (Hintze 1997) was utilized to test seed viability among sites, treatments and oak species. Data collected during 2001 was not included in the analysis because treatments were applied after bud formation so any effect of treatment would have been missed. ANOVA (Hintze 1997) was used to test for treatment effects on weevil populations and to examine differences between collection methods. The study areas were treated as random while treatments were fixed effects. All data conformed to normality and equal variance assumptions.

Results

In 2001, a total of 3974 chestnut oak and black oak acorns were collected of which only 17 were CO acorns (table 1). One sound and 10 weevil depredated CO acorns were recorded. Of the 3957 BO acorns produced, 26 percent (1028) were sound and 54.6 percent (2160) were weevil depredated. The remaining 19.6 percent (769) were inviable for other reasons (aborted, fungi, etc.).

In 2002, a total of 2962 CO and BO acorns were collected, 1326 were CO and 1636 were BO acorns (table 1). More CO acorns were sound than weevil depredated (25.0 vs. 21.0 percent, respectively) but more BO acorns were weevil depredated than sound (33.8 vs. 3.5 percent, respectively).

Chestnut oak acorn production responded significantly to the silvicultural treatments (table 2). CO acorn production increased in response to thinning and burning (fig 1A). Greater variability was recorded in CO

Table 2.—Analysis of variance testing for differences in *Q. prinus* and *Q. velutina* acorn production among site and treatment units in 2002. C: control, T: thin; TB: thin and burn; and B: burn. Significance ($P < 0.05$) is indicated by an asterisk.

Source	<i>Quercus prinus</i>				<i>Quercus velutina</i>			
	df	MS	F	P	df	MS	F	P
Unit	3	0.55	4.24	0.007*	3	0.197	2.91	0.037*
C vs. T	1		0.65	0.516	1		0.59	0.553
C vs. B	1		0.37	0.711	1		2.83	0.005*
TB vs. T+B	1		3.23	0.001*	1		0.92	0.358
Site	1	0.05	0.40	0.528	1	0.586	8.62	0.004*
Total	114				138			

Table 3.—Log-linear analysis of chestnut and black oak acorn seed viability in 2002 at two forests in southeast Ohio subjected to three treatments. S:site; T: treatment; and O: oak species.

Effect	df	Partial Chi-Square	Probability
Site	1	51.48	0.0001
Treatment	3	127.55	0.0001
Oak species	1	46.45	0.0001
S × T	3	11.26	0.0104
S × O	1	1.16	0.2818
T × O	3	144.72	0.0001
S × T × O	3	62.16	0.0001

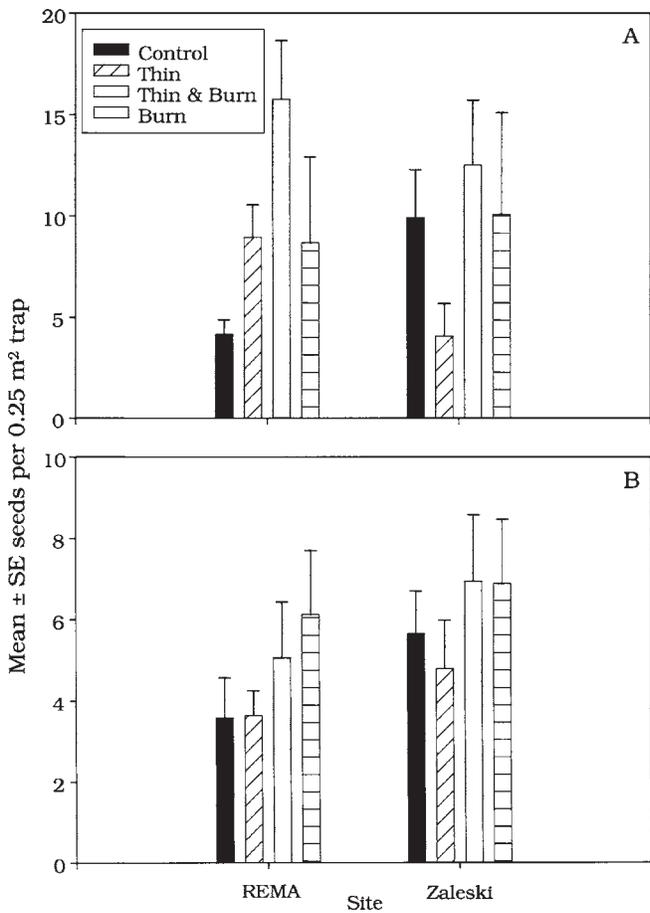


Figure 1.—Mean ± SE of chestnut (A) and black (B) oak acorns produced in different treatment units in two forests in southeast Ohio. Note that panels are on different scales.

production in the burn vs. the thin & burn unit. CO acorn production responded differently at each site in the control and thin units (fig. 1A).

Black oak acorn production responded significantly different at each site and to the silvicultural treatments (fig. 1B). BO acorn production increased in response to the thinning and burning and was greater than CO production.

Log-linear analysis indicated that seed viability at each site, for each treatment and between oak species was significantly different (table 3). More sound seeds were produced in the thin and burn and burn units than in the control and thin units (287 vs. 102 acorns, respectively).

A total of 35 *Curculio* weevils were trapped representing eight species and one unknown (table 4). Fourteen of the 25 were caught in the emergence traps (ET). The remaining *Curculio* weevils serendipitously appeared in the seed mast traps (ST). The dominant species was *Curculio strictus* with 15 individuals. Two rare species, *C. iowensis* and *C. orthorhynchus* (L.P. Gibson, personal communication), were collected only from Zaleski.

A total of 16 *Conotrachelus* weevils were caught representing 3 species (table 4). The most abundant was *Conotrachelus naso* with 11 individuals while the least abundant was *C. carinifer* represented by only one individual. *C. naso* weevils found in seed traps were generally associated with chestnut oak trees.

There was no difference in the number of weevils caught between the forests ($F = 0.90$, $P = 0.379$) or among the units ($F = 0.86$, $P = 0.529$). There was no difference in the number of weevils caught in emergence traps or weevils found in mast traps ($F = 0.61$, $P = 0.465$). No weevils were caught using Tanglefoot™ on tree boles.

Discussion

Many factors influence seed production. Increased seed production results from increased crown area (Greenberg 2000) and potentially from fire (Abrahamson and Layne 2002). We found that seed production increased in response to prescribed fire and to thinning combined with prescribed fire. Thinning reduced standing basal area, creating more space for existing oak trees; however, we did not record increased acorn production in thinned units.

Studies indicate that there is a high level of variability in acorn seed production among individual trees and even populations (Healy and others 1999, Greenberg 2000). McCarthy and Quinn (1989) suggest that most variability results from individual and annual variation in trees, rather than as a response to thinning (Healy and others 1999). While Healy and others examined the effects of thinning, our study combines the results of increased light produced by thinning with increased nutrients produced by fire. We found a great deal of variability among trees and forest sites but our data indicate that thinning and prescribed fire combined increased variability of acorn production greater than thinning alone.

Acorns are an important food source for wildlife. Native Americans also consumed acorns as part of their diet. McCarthy (1993) suggests that Native Americans in California burned in order to control invertebrate acorn infestation rates. Wright (1986) found evidence to support the role of prescribed fire in controlling acorn weevils in southeastern Ohio. We found that weevil populations were distributed evenly throughout the units but that sound seeds increased in the burn & thin relative to the control and thin units. Indeed, prescribed fire may decrease *Curculio* populations as a result of mortality. Since most weevil emergence occurred after the spring prescribed fire season in Ohio, weevil mortality is more likely to occur as a result of fall burning. During the fall fire season, generally October to November, there is an increased likelihood that weevils would be killed while exiting from fallen acorns or while in the forest floor layers while attempting to reach the soil. However, it may be possible for spring fire to have an effect on weevil survival as well. Some weevils may overwinter in the acorn and emerge the following spring (Gibson 1969). Likewise, absence of a litter layer due to burning in any season may influence predation rates.

Table 4.—*Curculio* and *Conotrachelus* species emergence dates, unit locations and collection method in two forests in southeastern Ohio. C: control, T: thin; TB: thin and burn; B: burn; ET: emergence trap; and ST: seed trap.

Species	Date	Site	Unit	Qty	Collection Method	Associated Oak mast tree
<i>Curculio strictus</i> Casey.	05/15/02	REMA	T	2	ET	
	05/20/02	ZAL	TB	4	ET	
	05/20/02	ZAL	B	3	ET	
	05/24/02	ZAL	TB	1	ET	
	05/26/02	REMA	B	1	ET	
	05/29/02	ZAL	T	1	ET	
	08/15/02	ZAL	TB	1	ST	Black
	08/19/02	REMA	T	1	ST	Chestnut
	10/12/02	ZAL	C	1	ST	Black
Total				15		
<i>Curculio confusor</i> Horn.	05/24/02	ZAL	B	1	ET	
	05/26/02	REMA	TB	1	ET	
	Total				2	
<i>Curculio proboscideus</i> Fab.	05/26/02	REMA	T	1	ET	
	10/05/02	ZAL	C	1	ET	
	10/09/02	REMA	TB	1	ST	Chestnut
	10/12/02	ZAL	C	1	ST	Black
	Total				4	
<i>Curculio iowensis</i> Casey.	08/15/02	ZAL	B	1	ST	Chestnut
	Total				1	
<i>Curculio pardalis</i> Chtrtn.	09/09/02	REMA	C	1	ST	Black
	09/09/02	REMA	B	1	ST	Chestnut
	10/09/02	REMA	T	1	ST	Black
	Total				3	
<i>Curculio sulcatulus</i> Casey.	09/09/02	REMA	C	1	ST	Black
	09/10/02	ZAL	C	1	ST	Black
	10/05/02	ZAL	TB	1	ET	
	0/09/02	REMA	T	1	ST	Black
	Total				4	
<i>Curculio orthorhynchus</i> Casey.	09/10/02	ZAL	T	1	ST	Chestnut
	09/25/02	ZAL	B	1	ET	
	10/12/02	ZAL	C	1	ST	Black
	Total				3	
<i>Curculio nasicus</i> Say.	10/05/02	ZAL	C	1	ET	
	10/05/02	ZAL	TB	1	ET	
	Total				2	
<i>Curculio</i> sp.	10/12/02	ZAL	B	1	ST	Chestnut
<i>Conotrachelus naso</i> Lec.	04/17/02	REMA	B	1	ET	
	05/26/02	REMA	TB	2	ET	
	06/04/02	ZAL	C	2	ET	
	06/04/02	ZAL	T	2	ET	
	08/15/02	ZAL	B	1	ST	Chestnut
	08/19/02	REMA	TB	1	ST	Chestnut
	09/09/02	REMA	TB	2	ST	Chestnut & black
	Total				11	
<i>Conotrachelus posticatus</i> Boh.	04/17/02	REMA	B	1	ET	
	05/26/02	REMA	C	1	ET	
	05/26/02	REMA	TB	1	ET	
	08/19/02	REMA	C	1	ST	Chestnut
	Total				4	
<i>Conotrachelus carinifer</i> Casey	09/09/02	REMA	C	1	ST	Black
Total				1		
Total <i>Curculio</i>				35		
Total <i>Conotrachelus</i>				16		

The position of the weevil larvae in the soil/litter horizons during a fire may be the key to their survival. Weevils that emerge from acorns earlier in the fall and descend into the soil will most likely escape the effects of an autumn fire. Prescribed fire temperatures in mixed-oak forests of southeast Ohio rarely exceed 30°C 2 cm below the soil surface (Riccardi and McCarthy, unpublished data) and will not cause weevil mortality. Weevils present in the forest floor litter layer during a fire will most likely be killed since forest floor fuels are the major fuel sources of prescribed fires in mixed-oak forests of southeast Ohio. Consumption of the forest floor removes potential hiding areas for weevils and could lead to greater predation by small forest mammals, also resulting in a reduction of weevil populations.

Three species of *Conotrachelus* are known to infest acorns in the United States (Gibson 1964) and we recorded all three species in our study. *Conotrachelus* weevils began emerging in April, before *Curculio* weevils. *Conotrachelus* weevil populations potentially would be affected by spring fires; however, because they infest damaged or otherwise open acorns, their effects on acorn viability are probably less crucial than *Curculio* weevil damage.

Our data suggest that the combination of thinning, to increase light levels and to increase spatial availability for tree crowns, coupled with a flush of nutrients resulting from prescribed fires can increase acorn oak production. Furthermore, prescribed fire, particularly during the autumn, may result in weevil mortality leading to an increase in the production and survival of sound acorns.

Acknowledgments

This is contribution 39 of the National Fire and Fire Surrogate Research (FFS) Project. This research was funded by the USDA Forest Service through the National Fire Plan and the U.S. Joint Fire Science Program (JFSP) to BCM. CLR is grateful to the many field assistants and especially to L.P. Gibson for assistance with weevil identification.

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FOREST DYNAMICS AT THE EPICENTER OF AN OAK DECLINE EVENT IN THE BOSTON MOUNTAINS, ARKANSAS – YEAR ONE

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ABSTRACT.—Oak decline is a significant problem in the Boston Mountains and Ozark Highland region of Arkansas and Missouri which recently has resulted in mortality of oaks across thousands of hectares of forest. However, research plots at oak decline sites usually are established after the decline event is visually evident. Results of a case study in which we serendipitously established research plots at what became the epicenter of an oak decline event are presented. In August of 2000, standing trees greater than 14 cm in diameter were inventoried on twenty-four 0.3025 ha plots. By late summer of 2001 oak decline symptoms became visually evident at the site. In November 2001, overstory trees on six of those plots were remeasured. I compare survival and structural changes on those six plots. For instance, an average of 38 trees per ha died by the fall of 2001. In addition, basal area of standing dead trees (hereafter referred to as snags) increased from 1.8 m²/ha in 2000 to 4.4 m²/ha in 2001 ($p = 0.019$). Correspondingly, basal area of live trees decreased from 24.1 m²/ha in 2000 to 21.2 m²/ha in 2001 ($p = 0.022$). Mean values for diameter of snags increased from 20.4 cm DBH in 2000 to 24.3 cm DBH in 2001 ($p = 0.015$). All 24 plots are scheduled for remeasurement in 2004. The serendipitous establishment of permanent study plots on this site will provide a unique opportunity to examine an oak decline event before, during and long after visual evidence of the event has occurred, providing potential insights that are rarely available for such a disturbance event.

From 1856 to 1986, there had been 57 oak mortality events recorded in the eastern United States (Millers and others 1989). This included one in 1959 in the Ozark Mountains of Arkansas (Toole 1960), one in 1980-1981 in Northwestern Arkansas (Bassett and others 1982, Mistretta and others 1984), and an event in Missouri from 1980 to 1986 (Law and Gott 1987). The current oak decline event in Arkansas and Missouri is severely affecting up to 120,000 ha in the Ozark National Forest of Arkansas alone (Starkey and others in press).

Stresses that lead to oak decline in the eastern US are complex interactions of many factors (Wargo and others 1983). Manion (1991) describes oak decline as the outcome of the interaction of three major groups of factors: predisposing factors, inciting factors and contributing factors. Predisposing factors include physiologic age and oak density; inciting factors include a long drought; and contributing factors include opportunistic insects such as oak borers and diseases like Hypoxylon canker.

In 1998-2000, a three-year drought occurred across the region. Drought is defined as an “inciting factor” of oak decline by Manion (1991) and by Starkey and others (in press). This, coupled with a forest of high tree density and old mature trees, has made Arkansas upland hardwood forests more vulnerable to oak decline (Oak and others in press). Drought, density and maturity likely were important factors in the current oak decline event in Arkansas and Missouri.

An oak decline event, such as we have now, has the potential to significantly alter forest structure and species composition. Based on previous oak decline events (Tainter and others 1984, Oak and others 1988, Starkey and others 1989), it is likely that oaks will remain an important component of these forests at the regional scale. Within many stands, however, oaks will no longer be the dominant tree without active management to encourage oak regeneration and recruitment. On sites where oak reproduction exists but competing species have the advantage, active management will be necessary to successfully grow a new cohort of oak into the tree canopy. An understanding of how species and forest structure are changing will aid our understanding of how to address these issues in the future.

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One year after measuring vegetation on permanent plots at a site in the Boston Mountains of Arkansas, oak decline symptoms became visibly evident. Although this meant the temporary loss of one replication of another study, it provided a serendipitous and unprecedented opportunity to establish a case study examining the dynamics of an oak decline event in which detailed early data are available for comparison.

The objective of this paper is to evaluate mortality by species and changes in forest structure one year after the first woody vegetation measurements were taken. The long-term objective is to compare stand dynamics among areas treated with a growing season prescribed fire, a dormant season prescribed fire and a control area.

Study Area

The study site is a 32 ha area in an upland oak-hickory stand which is approximately 80 years old. It is located in the Boston Mountains of Arkansas, part of the southern lobe of the Central Hardwood Region (Merritt 1980). More specifically the study is located in the northwestern corner of Pope County, approximately 3 kilometers southeast of Sand Gap, Arkansas. The forest stand is dominated by oak (*Quercus* spp.) and hickory (*Carya* spp.) and is now the epicenter of an oak decline event. In 2000, prior to identification as an oak decline site, basal area for all standing trees was 25.9 m²/ha and there were 417 standing trees/ha, of which 1.8 m²/ha of basal area and 53 trees/ha were standing dead trees (hereafter referred to as snags). Stocking for this stand was 88 percent in August of 2000.

The Boston Mountains are the highest and most southern member of the Ozark Plateau Physiographic Province. They form a band 48- to 64-km wide and 320-km long from northcentral Arkansas westward into eastern Oklahoma. Elevations range from about 275 m in the valley bottoms to 760 m at the highest point. The plateau is sharply dissected. Most ridges are flat to gently rolling and generally are less than 0.8 km wide. Mountainsides consist of alternating steep simple slopes and gently sloping benches. Vegetation across the landscape is a forest matrix with inclusions of non-forest.

Methods

In late winter of 2000, twenty-four 55 x 55 m overstory plots (0.3025 ha) were established across the study area. During the summer of 2000, all standing live trees and snags > 14 cm DBH (1.37 m above ground level) within each plot were measured, and azimuth and distances from plot center to each tree were recorded. Diameter was measured to the nearest 0.1 cm. Species, log grade, crown class, and multifactor damage codes were also recorded for each tree. A Hewlett Packard 200LX palmtop computer was used to record all data in the field.

In each of the twenty-four 55 x 55 m plots, four circular 0.01 ha plots were established to inventory midstory trees. Within each 0.01 ha plot, all trees from 5 cm to < 25 cm dbh (1.97 to < 9.84 in dbh) were measured in July of 2000. Diameter at breast height (1.37 m above ground level) was measured to the nearest 0.1 cm and recorded.

In each of the 0.01 ha plots, five circular 0.00054 ha plots were established to inventory regeneration. Within each 0.00054 ha regeneration plot, all trees < 5 cm dbh were measured in spring of 2000.

These plots were intended to be part of one replication of a periodic fire study, but by the late summer of 2001, oak decline symptoms were visibly evident at the site. At that point, this site became the start of a long-term case study of oak decline. In November and December of 2001, the overstory on six of the 24 plots was remeasured. Three plots are located at the northwest end and three at the southeast end of this southwest facing slope. These plots were chosen because they are located in areas where a growing season prescribed fire (southeast end) and dormant season prescribed fire (northwest end) will be applied in the near future. Only six plots were remeasured due to time constraints. This site is now the epicenter of a severe oak decline event covering hundreds of hectares.

The data were analyzed using the following methods. First a t-test was used to compare 2000 plot values with 2001 plot values. For one measure (mean DBH of snags) the equal variance test (Levene Median test) failed. In this case the Mann-Whitney Rank Sum Test was used. To model survival, logistic regression was used to estimate tree survival probability. A model building approach was used as described in Hosmer and

Lemeshow's (1989, p. 82) text on applied logistic regression. Logistic regression has been used in studies to predict growth and success of regenerated oak stems (Lowell and others 1987), to estimate the contribution of planted hardwoods to future stocking (Johnson and Rogers 1985), to model regeneration of oak stands (Dey and others 1996), and to model the competitive capacity of trees underplanted in shelterwoods (Spetich and others 2002).

Results

Changes in forest structure occurred within one year of the first measurements. For instance, an average of 38 trees per ha died by the fall of 2001. In 2000, the percent of all standing trees that were snags was 13 percent. This increased to 20 percent by 2001. Basal area of snags increased from 1.8 m²/ha in 2000 to 4.4 m²/ha in 2001 ($p = 0.019$). Correspondingly, basal area of live trees decreased from 24.1 m²/ha in 2000 to 21.2 m²/ha in 2001 ($p = 0.022$). Mean values for diameter of all snags also were statistically different, increasing from 20.4 cm DBH in 2000 to 24.3 cm DBH in 2001 ($p = 0.015$).

The number of snags increased across all diameter classes from 2000 to 2001, with a corresponding decrease in standing live trees (figs. 1A and 1B.). The percent of snags generally decreased with increasing diameter class in 2000 (fig. 2). By 2001 this percent increased over most diameter classes with a peak at the diameter class of 60 cm.

Northern red oak (*Quercus rubra* L.) had the highest mortality rate. In only one year (from 2000 to 2001), the number of northern red oak snags per ha more than doubled, increasing from 23 trees per ha to 51 trees per ha respectively ($p = 0.029$) (fig. 3). Other species (white oak, *Quercus alba* L.; hickory, *Carya spp.*; black gum, *Nyssa sylvatica* Marsh.; post oak, *Quercus stellata* Wangenh.; red maple, *Acer rubrum* L.; and black maple, *Acer nigrum* Michx. f.) remained near their 2000 levels. The increase in northern red oak mortality

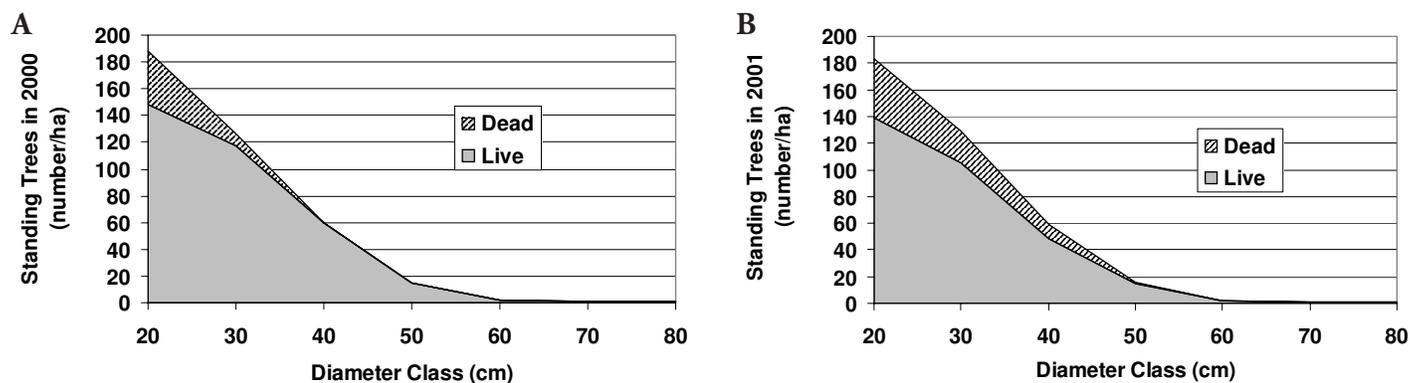


Figure 1.—A) Live and dead (snags) standing trees per ha in 2000 – all species. B) Live trees and snags per ha in 2001 – all species.

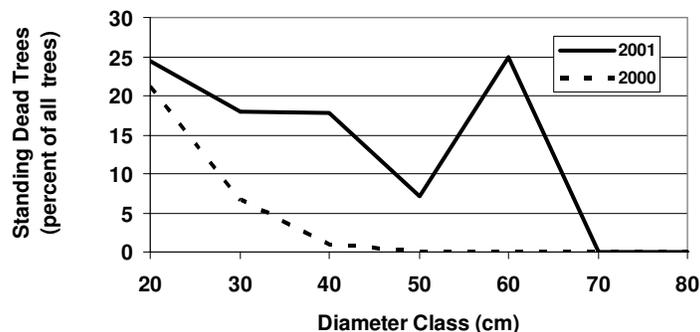


Figure 2.— Percent of all standing trees that were snags.

occurred over all diameters from 2000 to 2001 with the greatest number of dead trees increasing in smaller diameters (figs. 4A and 4B). Based on these results, a first-year survival probability model was developed for northern red oak using logistic regression (fig. 5). This model indicates that by the end of the first year (2001) of visible oak decline symptoms, survival was highest for large diameter trees while smaller trees were more likely to succumb to mortality.

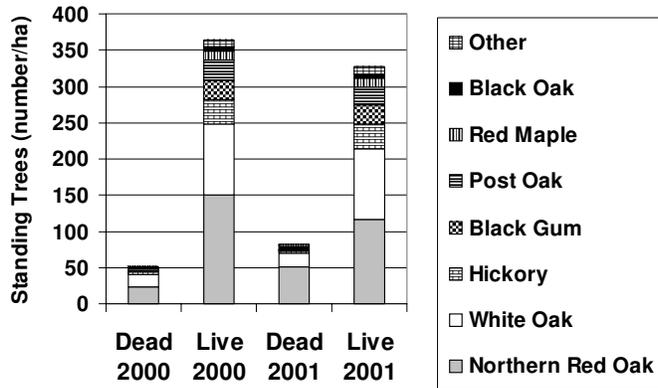


Figure 3.—Trees per ha by species and status. Other species include black cherry, *Prunus serotina* Ehrh.; black walnut, *Juglans nigra* L.; American elm, *Ulmus americana* L.; black locust, *Robinia pseudoacacia* L.; serviceberry, *Amelanchier* spp.; sassafras, *Sassafras albidum* Nutt.; and White ash, *Fraxinus americana* L.

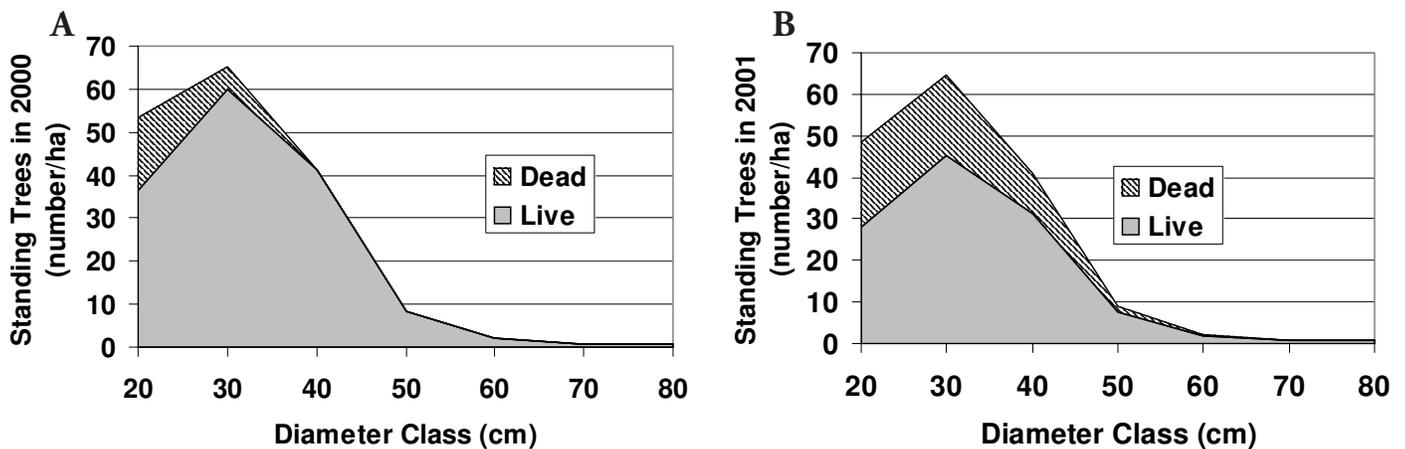


Figure 4.—A) Number of standing northern red oak trees per ha in 2000 by diameter class and status. B) Number of standing northern red oak per ha in 2001 by diameter class and status.

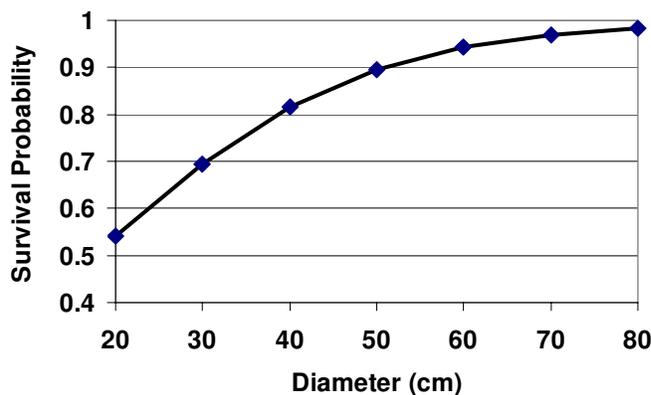


Figure 5.—Survival probability of northern red oak one year after oak decline symptoms were visually evident.

Discussion

Although the decline event wasn't visually identified in this stand until 2001, stresses on the stand began much earlier. These stresses included the three-year drought from 1998 to 2000 and stand density (364 trees/ha and basal area of 24.07 m²/ha in 2000). Forest structure had been significantly altered by 2001. Increases in the number of snags along with corresponding changes in density will likely allow a greater amount of light through the canopy. Without intervention, many regenerating red oaks will be outcompeted by faster growing co-occurring species. However, because this is a xero-mesic site, some of the more drought tolerant advance regeneration white oak should survive on dryer microsites. On moister microsites, white oak will be more likely to succumb to competitors that exhibit more rapid height growth.

The percent of standing trees that were dead (snags) decreased from the 20 to 50 cm diameter classes in 2001. However, there was a dramatic increase around the 60 cm diameter class. This is likely due to the much smaller number of trees at larger diameters (fig. 1) and the corresponding increase in variability with a smaller sample size. For instance at the 60 cm class in 2000 there was an average of 2 standing live trees per ha and zero snags. The loss of only one tree per ha would result in a 50 percent increase in mortality in that diameter class.

It isn't surprising that northern red oak represented the majority of the mortality since, of the oak species at this site, it is the species most sensitive to drought. Figures 4A and 4B indicate that most of the mortality in red oak occurred in trees of relatively small diameter classes. These small trees are typically under more competitive stress than larger diameter trees and mortality rates for small trees are typically higher than for larger diameter trees (Harcombe 1987). At this site, this could lead to displacement of northern red oak by other co-occurring species.

Northern red oak in larger diameter classes generally survived through the first year of the oak decline event. However, overall mortality will likely increase for larger trees by the 2004 inventory since many overstory trees were already showing signs of stress by the fall of 2001.

In future inventories of this stand, we hope to have the resources to take more detailed measurements, as we did in 2000. These included litter weight, tagged regeneration, midstory trees, and down deadwood. We also plan to examine the effectiveness of summer season prescribed fire, dormant season prescribed fire and no fire options. Following forest dynamics of this site starting at a time before it was identified as an oak decline site should increase our understanding of oak decline dynamics.

Acknowledgments

The author extends his appreciation to the field technicians who installed and measured this study: Richard Chaney, Jim Whiteside, Arvie Heydenrich, and Brenda C. Swboni. Thanks to Ozark National Forest personnel for assistance with stand selection. Thanks to Shibu Jose, Daniel A. Marion and two anonymous reviewers for reviewing this manuscript and to Betsy L. Spetich for editorial guidance.

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FACTORS ASSOCIATED WITH OAK MORTALITY IN MISSOURI OZARK FORESTS

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ABSTRACT.—Forest management to improve stand vigor and growth requires foresters to distinguish between trees that will respond favorably to treatment and trees that are likely to grow poorly or die. The Missouri Ozark Forest Ecosystem Project provided an opportunity to quantify factors associated with oak mortality in mature, fully-stocked, second-growth forests. We monitored more than 24,000 permanently-tagged oak trees from 1992 to 2002 and identified key factors associated with their survival or mortality. Factors considered included tree characteristics such as species, crown class, and diameter as well as site characteristics such as slope, slope position, aspect, and ecological land type. Classification and Regression Tree (CART) analyses showed that species and crown class were more important than site factors in identifying cohorts with high mortality. For a given diameter and species, tree mortality decreased with improved crown position, but mortality rates increased sharply with increasing diameter for dominant and codominant scarlet and black oaks. Mortality rates for dominant and codominant white and post oaks were stable or increased only slightly with increasing tree diameter. Mortality of black and scarlet oaks exceeded 20 percent per decade for dominant and codominant trees > 8 inches dbh. Managers can use these results to identify groups of trees with a high probability of mortality.

Forest management designed to improve stand vigor and growth requires distinguishing between trees that will respond favorably to treatment and trees that are likely to grow poorly or die. In doing so foresters must try to identify the tree and site characteristics that best indicate whether or not an individual tree will remain healthy or will die before the next treatment opportunity. Of course tree mortality is a normal part of forest dynamics. Most of the mortality in stands arises from competition for growing space and is largely the result of self thinning (Johnson and others, 2002). As trees get larger and require more growing space to survive, some trees are periodically crowded out and die. Events such as wildfire, drought, windstorms, and insect and disease epidemics also cause trees to die and often at rates exceeding those caused by competition alone (Johnson and others 2002).

Although mortality is fairly predictable at stand scales, it is more difficult to predict which individual trees are likely to die. However, this is precisely what foresters attempt to do when marking trees for harvest. While marking, they consciously (or perhaps subconsciously) evaluate the form and quality of each tree and look for characteristics that indicate whether or not an individual tree will survive and grow vigorously. They also consider many other factors such as stand density and site quality that influence tree growth and mortality. In Missouri, there is considerable concern about oak decline. Red oak group species (*Quercus spp* L.; subgenus *Erythrobalanus*) appear to be particularly susceptible, especially those that are large or physiologically mature, and growing on droughty sites (Law and Gott 1987; Starkey and Oak 1989). Because of concern about oak decline, foresters in Missouri are more likely to remove red oak group species during stand improvement operations, especially those that are large (>18 inches DBH) or on poor sites even if the trees have no visible symptoms of decline (Kabrick and others 2002). However, these marking decisions are based upon the forester's intuition or experience rather than upon documented evidence.

Quantifying factors associated with the mortality of individual trees is challenging because it requires monitoring the survival of many individual trees for long periods. The Missouri Ozark Forest Ecosystem Project (MOFEP) afforded a rare opportunity to better quantify mortality rates of oaks. MOFEP is a long-term study to quantify the effects of forest management on upland oak systems. As such, a large number of

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individual trees were marked and are closely monitored. Here we report the results of our analysis of more than 24,000 oak trees monitored for ten years to determine if tree, stand, or site variables routinely collected during forest inventories could be used for predicting the survival of individual trees in second-growth oak forests.

Study Areas

The MOFEP study and experimental design is described elsewhere in detail (Brookshire and Shifley 1997, Shifley and Brookshire 2000, Shifley and Kabrick 2002). The study consists of nine sites ranging in size from 772 to 1,271 acres, primarily within the Current River Oak Forest Breaks and the Current River Oak-Pine Woodland Hills landtype associations of the Ozark Highlands. The Current River Oak Forest Breaks has narrow ridges and steep sideslopes with relief of 300-450 ft, which exposes the Roubidoux, Gasconade, and Eminence bedrock formations. The Current River Oak-Pine Hills has broad ridges with relief <300 feet and exposes only the Roubidoux and Gasconade bedrock formations. Upland soils of these landtype associations are primarily Ultisols and Alfisols formed in hillslope sediments or residuum; soils in upland waterways and bottomlands are primarily Ultisols and Alfisols formed in gravelly alluvium (Kabrick and others 2000, Meinert and others 1997).

Methods

Information about individual tree characteristics (e.g., species, crown class, diameter, diameter increment) were derived from 648 permanent vegetation plots distributed roughly equally among the nine MOFEP sites. Since the first inventory completed in 1992, these permanent plots have been re-inventoried approximately every three years to document the condition of woody vegetation. Characteristics recorded for each tree include species, DBH, or size class for trees < 1.5 inches DBH, and status (e.g., live, dead, den, cut, blow-down), and crown class (e.g., dominant, codominant, intermediate, suppressed) (Jensen 2000). Site characteristics (e.g., slope, land form, aspect, soil type, ecological land type, and land type association) for each plot were derived from a detailed landscape-scale soil mapping project conducted on MOFEP in 1994-1995 (Kabrick and others 2000, Meinert and others 1997).

Oaks are the dominant trees on the MOFEP sites and four oak species, white oak (*Quercus alba* L.), black oak (*Q. velutina* Lam.), scarlet oak (*Q. coccinea* Muenchh.), and post oak (*Q. stellata* Wangenh.) comprise 71 percent of the basal area (Kabrick and others, in press). Other oaks found at MOFEP include chinkapin oak (*Q. muehlenbergii* Engelm.), blackjack oak (*Q. marilandica* Muench.), Shumard oak (*Q. shumardii* Buckl.), and northern red oak (*Q. rubra* L.), but in combination they comprise only 1 percent of the basal area. Shortleaf pine (*Pinus echinata* Mill.) (8 percent), pignut hickory [*Carya glabra* (Mill.) Sweet] (4 percent), black hickory (*C. texana* Buckl.) (4 percent), mockernut hickory (*C. tomentosa* Poir. Nutt.) (4 percent), flowering dogwood (*Cornus florida* L.) (3 percent), and blackgum (*Nyssa sylvatica* Marsh.) (2 percent) also are in the study area.

Data Analysis

We monitored 24,000 permanently-tagged oak trees from the initial inventory completed in 1992 to the inventory completed in 2002 and identified factors correlated to mortality. We limited our analysis to the four most abundant oaks (white oak, black oak, scarlet oak, and post oak) and trees that were ≥ 4.5 inches DBH during the initial inventory. We analyzed data from all nine MOFEP sites but only from stands that were not harvested during the past ten years. Data from plots were converted to an acre basis for analysis. We evaluated a suite of individual tree characteristics (tree DBH, species, crown class), stand condition characteristics (stand density and basal area), and site variables (aspect, slope, soil type, ecological land type, land form, land type association) for their ability to predict individual tree mortality.

Because there is a hierarchical structure and interdependence among certain variables, we used the nonparametric, hierarchical model—classification and regression tree (CART)—(Breiman and others 1984) to study the relationship of these variables to individual tree mortality. The binary response variable is whether a tree is dead (0) or is alive (1). We used CART to recursively partition the data into paired nodes (subsets) such that one node contained as many dead trees as possible, and the other node contained as few dead trees as possible based on a cutoff value of one of the explanatory variables. Therefore, the

distribution patterns of dead trees in the data and factors (or factor interactions) and factor values associated with these patterns were revealed and explicitly represented by the regression “tree” or diagram showing the hierarchy among variables.

We used the 10-fold cross-validation to construct the best CART model which minimized the overall misclassification rates (Steinberg and Colla 2000). With the best CART model for each data set, we bootstrapped each node 1000 times and calculated the mean and 95 percent confidence interval (CI) of the probability of dead trees as a measure of mortality rate and variation within that node (risk group).

For cohorts identified by the CART model, we calculated the mean mortality per decade and ranked cohorts by mortality rate from highest to lowest. Once ranked, we calculated the cumulative mortality (i.e., the mortality rate for the remaining trees if all cohorts with a higher mortality rate are harvested) so that we could estimate the mortality rate for the residual stand. We also calculated the mortality change as the difference between successive cumulative mortality rates. Large differences in mortality change indicated where large reductions in mortality in the residual stand if the indicated cohort is removed. For simplification when summarizing results to provide guidance to managers, we selected a single threshold diameter (i.e., 12 inches DBH), the approximate midpoint between the 8-inch and 15-inch DBH thresholds identified by the CART model. By doing so, we were better able to use our findings to make practical recommendations to managers.

We also graphed mortality rates by species, crown class, and 2-inch dbh classes to illustrate trends and to compare results from the MOFEP site to those previously observed in Missouri (Shifley and Smith 1982).

Results

Of the 24,000 white oaks, post oaks, black oaks, and scarlet oaks that we monitored, more than 3,300 died during the 10-year sampling period for an overall mortality rate of 13.9 percent per decade (or 1.5 percent per year¹). The variables most correlated to oak mortality, in order of importance, were: species, crown class, environmental variables (i.e., site location and ecological land type), and DBH (fig. 1; table 1). The order of importance varied slightly by species group.

Figure 1 also shows that approximately 21 percent of the red oak group species (scarlet oak and black oak) died and their mortality rate was four times greater than for white oak group (white oak and post oak) species. Red oak group species in the suppressed and intermediate crown classes had mortality rates 2.7 times greater than for those in the co-dominant or dominant classes and suppressed red oak group species had mortality rates that were 1.5 times greater than those in intermediate crown classes. Overall, mortality was greater for suppressed and intermediate red oaks located in sites within or near the Current River Oak-Pine Woodland Hills land type association. This particular land type association is dominated by droughty and nutrient-poor soils and is considered particularly susceptible to oak decline.

Within the white oak group, suppressed trees had mortality rates that were 3.4 times greater than for the other crown classes. Suppressed post oaks were twice as likely to die and intermediate post oaks were 3 times more likely to die as white oaks in these respective crown classes. Suppressed white oaks were more likely to die in ecological land types having higher site quality such as in bottomlands or on north-facing slopes.

For all species within each crown class, mortality increased with increasing diameter. Our analysis suggested that there were threshold diameters beyond which mortality increased significantly. These threshold diameters differed by species and site conditions (table 1). For example, mortality increased significantly for dominant and codominant white oak group species > 13 inches DBH regardless of environment. However, the threshold diameter for red oak group species was only 8 inches DBH for trees in sites within or adjacent to the Current River Oak-Pine Woodland Hills land type association, but was 15 inches DBH in the other sites.

¹In general the periodic mortality rate for a period of n years is equal to 1 minus the periodic survival rate. Annual survival rates are computed from periodic rates as one would compute compound interest: annual survival = (periodic survival)^(1/n). Likewise periodic survival over n years = (annual survival)ⁿ. Similar conversions for mortality rates must be computed in terms of corresponding survival rates and then determined by subtraction.

Table 1. —Nodes for the CART partition of MOFEP mortality data (see fig. 1) with node population size, description of the partition threshold, and the 95 percent bootstrap confidence interval (CI) of the mortality rate (percent per decade) within each node. Terminal nodes are indicated by bold type.

Node#	Node size (%)	Description	95 Percent CI
1	24000(100)	All white, post, black and scarlet oak	13.5-14.4
2	13188 (55)	Black and scarlet oak	20.1-21.5
3	10812(45)	White and post oak	5.0-5.9
4	3608 (15)	Intermediate or suppressed black and scarlet oak	36.7-39.8
5	9580 (40)	Dominant or codominant black and scarlet oak	13.6-14.9
6	1961 (8)	Suppressed white and post oak	11.5-14.7
7	8851 (37)	Dominant, codominant and intermediate white and post oak	3.4-4.2
8	2016 (8)	Intermediate or suppressed black and scarlet oak on sites ¹ 2-6, 9	29.1-33.1
9	1592 (7)	Intermediate or suppressed black and scarlet oak on sites ¹ 1,7, 8	44.9-50.1
10	4658 (19)	Dominant or codominant black and scarlet oak on sites ¹ 6-9	16.9-19.1
11	4922 (21)	Dominant or codominant black and scarlet oak on sites ¹ 1-5	9.9-11.6
12	266 (1)	Suppressed post oak	20.3-31.2
13	1695 (7)	Suppressed white oak	9.7-12.6
14	4424 (18)	Intermediate white and post oak	4.6-6.0
15	4427 (18)	Dominant and codominant white and post oak	1.9-2.8
16	298 (1)	Suppressed black and scarlet oak on sites ¹ 2-6,9	41.6-53.0
17	1718 (7)	Intermediate black and scarlet oak on sites ¹ 2-6,9	26.3-30.2
18	249 (1)	Suppressed black and scarlet oak on sites ¹ 1,7,8	60.2-71.9
19	1343 (6)	Intermediate black and scarlet oak on sites ¹ 1,7,8	41.4-46.7
20	689 (3)	Dominant and codominant black and scarlet oak ≤ 8 inch DBH on sites ¹ 6-9	7.0-11.1
21	3969 (15)	Dominant and codominant black and scarlet oak > 8 inch on sites ¹ 6-9	18.4-20.7
22	4263 (18)	Dominant and codominant black and scarlet oak ≤ 15 inch DBH on sites ¹ 1-5	8.2-10.1
23	659 (3)	Dominant and codominant black and scarlet oak > 15 inch DBH on sites ¹ 1-5	17.5-23.7
24	877 (4)	Suppressed white oak on ELT ² 11,17,19-23	5.6-9.1
25	818 (3)	Suppressed white oak on ELT ² 5,6,7,15,18	12.8-17.7
26	780 (3)	Intermediate post oak	10.0-14.7
27	3644 (15)	Intermediate white oak	3.1-4.4
28	3096 (13)	Dominant and codominant white and post oak ≤ 13 inch DBH	0.9-1.8
29	1331 (6)	Dominant and codominant white and post oak > 13 inch DBH	3.5-5.7
30	2120 (9)	Intermediate white oak on ELT ² 7,11,15,17,19-23	1.3-2.4
31	1524 (6)	Intermediate white oak on ELT ² 5,6,18	5.3-7.7

¹Site refers to Missouri Ozark Forest Ecosystem Project study sites 1-9.

²Ecological land types (Miller, 1981): Dry bottomlands (5); dry-mesic bottomlands (6); mesic toeslopes (7); dry, narrow ridges (11); dry, broad ridges (15); dry south slopes (17); dry-mesic north slopes (18); glade savanna (19); dry-mesic limestone forest (20); glade (21); xeric limestone forest (22); dry limestone forest (23).

Across all diameter and crown classes mortality rates per decade for white oak (5 percent) were < post oak (9 percent) < scarlet oak (19 percent) < black oak (22 percent). Mortality rates decreased with increasing crown class. Within a crown class mortality rates increased with increasing diameter; the increase was small for white and post oaks, but mortality rates for dominant and codominant scarlet and black oaks increased dramatically (to approximately 40 percent) with increasing tree diameter (fig. 2). Mortality rates for white and post oaks on the MOFEP sites were at or below historical rates for all Missouri; mortality rates for black and scarlet oaks exceed historical state-wide rates, most notably for large diameter trees (fig. 3).

The CART analysis provides direct guidance to forest managers. This is most easily seen when the terminal nodes in the CART Diagram (fig. 1, table 1) are rearranged in order of decreasing mortality probability (table 2). Table 2 clearly shows which subpopulations are most likely to die (e.g., suppressed black and

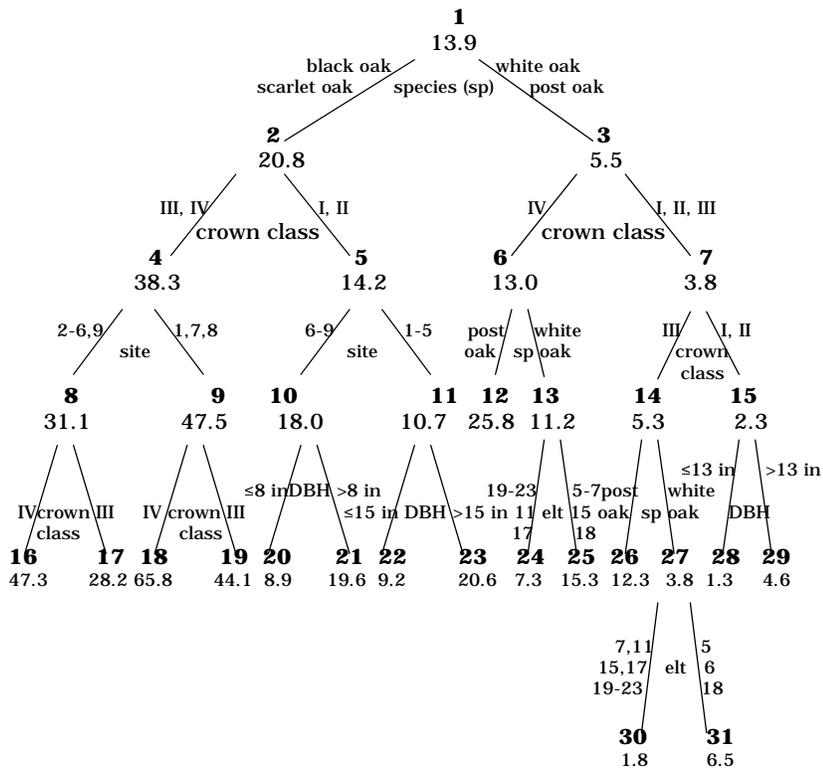


Figure 1.—CART partition of mortality for white, post, black and scarlet oaks in MOFEP plots, 1992 -2002. Nodes are numbered in bold type. Each node shows the mortality rate (percent) for the subpopulation. Variables that partition nodes are shown between node partitions. The threshold value for each variable is interpreted in Table 1.

scarlet oaks with mortality rates of 56 percent per decade) and which are most likely to survive another decade (e.g., dominant white and post oaks < 12 inches DBH with 1 percent mortality per decade).

Discussion

Tree species and crown class were the two most important predictors of tree mortality. Species in the red oak group had higher mortality rates than those in the white oak group as has been shown elsewhere in Missouri and the Central Hardwood Region (Shifley and Smith 1982, Smith and Shifley 1984). For a given diameter, trees in dominant or codominant canopy classes had substantially lower mortality rates than those in lower canopy classes: this was especially true for species that are shade intolerant such as scarlet oaks and post oaks (fig. 2). However, the population of trees larger than 12 inches DBH is comprised almost exclusively of dominant and codominant trees. The mortality rate of dominant and codominant black and scarlet oaks increased dramatically as tree diameter increased. A 20-inch dominant or codominant scarlet oak was no more likely to survive for 10 years than an intermediate scarlet oak in the 4- to 12-inch DBH range.

Environmental variables indicative of site quality played a secondary role and their effects differed by species. For example, on the more-productive ecological landtypes, there was higher mortality of intermediate canopy class white oaks. This suggests that on more productive ecological land types, competition-induced mortality occurred at higher rates than on less productive ecological land types. We anticipated greater oak mortality, particularly of red oak group species, in poorer, less-productive stands. As indicated by the CART regression tree (fig. 1, table 1) site location (in this case MOFEP experimental sites or compartments spanning about 900 acres each) did have a statistically significant relationship with mortality rates for black and scarlet oak (fig. 1 nodes 8,9,10,11). Thus, site location played a role in mortality rates for these species, and we know there are differences among sites in the types of soil that occur, topography, and the mix of ecological landtypes. However our attempts to further identify specific site characteristics by including variables such as slope, aspect, soil map unit, or ecological landtype provided little additional explanatory power to the analysis.

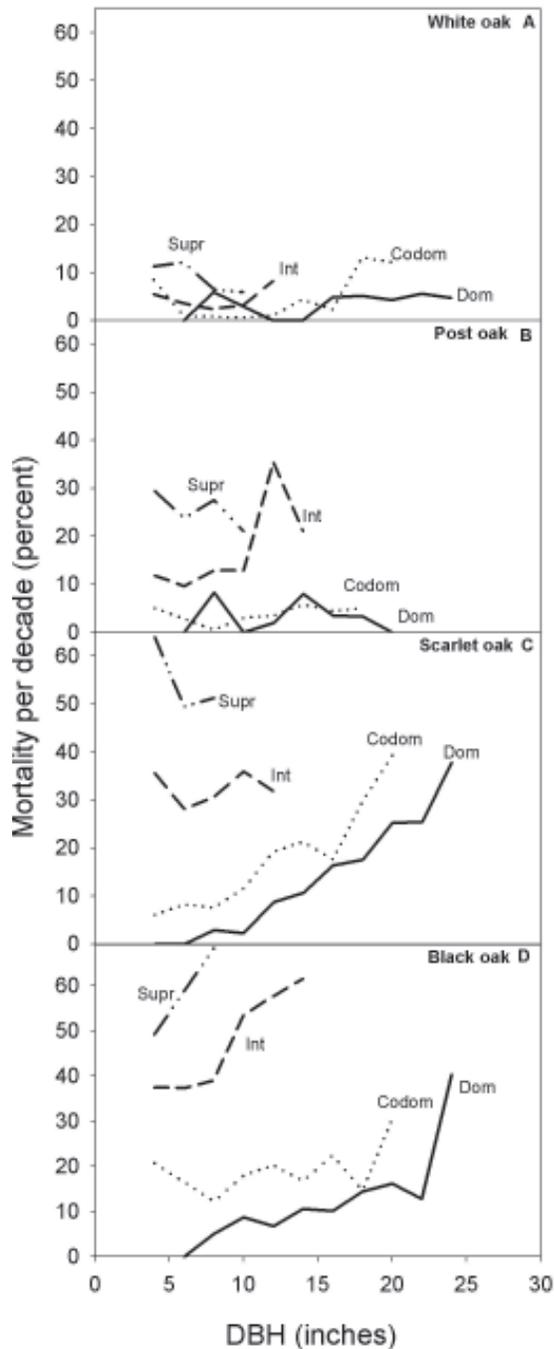


Figure 2.—Mortality rates per decade by species and crown class for MOFEP sites. As expected, mortality rates for a tree of given dbh increase markedly as canopy position decreases from dominant (Dom) to codominant (Codom), intermediate (Int), and suppressed (Supr) crown classes. Mortality rates for scarlet and black oaks increase with increasing diameter for dominant and codominant trees, a pattern that contrasts sharply with that for white and post oaks. Large black and scarlet oaks in the dominant and codominant canopy layers proved highly susceptible to mortality. As a rule of thumb, the bigger they are the sooner they fall.

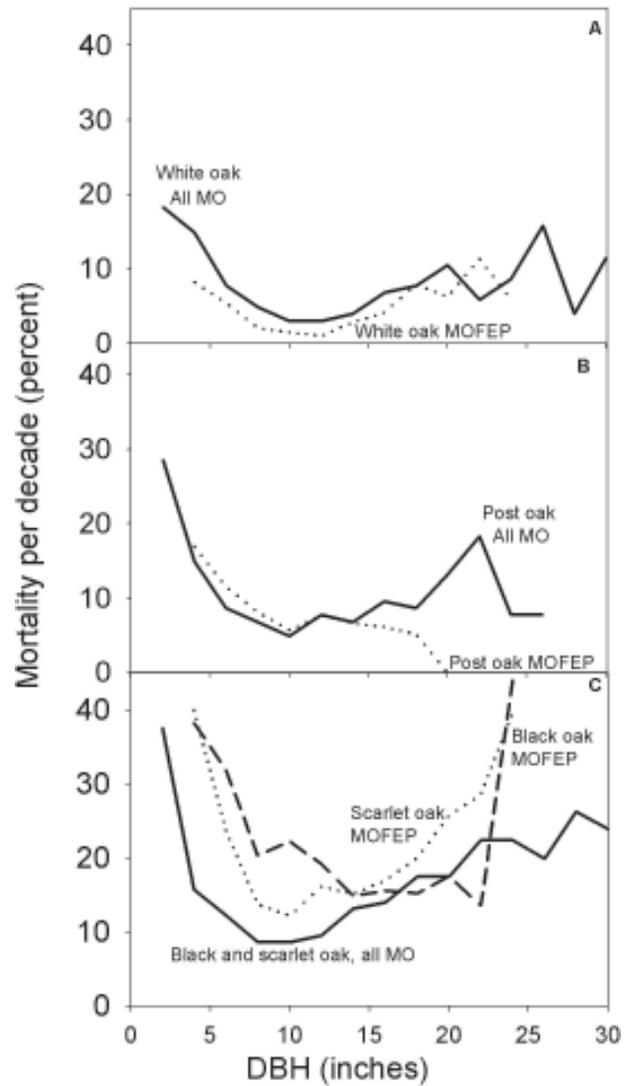


Figure 3.—Mortality rates per decade by species and dbh. Mortality rates for black and scarlet oak were roughly twice those of white and post oak. For white oak and post oak, mortality rates for MOFEP sites from 1992 to 2002 were generally less than reported for all Missouri from 1959-1972 as reported in Shifley and Smith (1982). However, mortality rates for black and scarlet oak on MOFEP sites were generally higher than the average for Missouri, 1959-1972.

Table 2.—Categories (subpopulations) of trees and related mortality rates in rank order from highest to lowest mortality probability. In general, removing cohorts in order starting at the top of the list will minimize the overall mortality rates for the remaining stand. However, cohorts such as suppressed trees are rarely cut and simply left to die through self-thinning. Mean mortality per decade indicates the mean mortality rate for that cohort. Cumulative mortality is the combined mortality rate for that cohort in combination with all those listed below it, and can be used to estimate the mortality rate for the residual stand. The column labeled mortality change is the difference between successive cumulative mortality rates; large values indicate the potential for large reductions in mortality in the residual stand if the indicated cohort is removed.

Categories	Mean mortality per decade (%)	Cumulative mortality (%)	Mortality change (%)
Suppressed black and scarlet oaks	56	14	1
Intermediate black and scarlet oaks	35	13	3
Suppressed post oaks	26	10	1
Dominant and codominant black and scarlet oaks > about 12 inches dbh	20	9	3
Intermediate post oaks and suppressed white oaks	9	6	1
Dominant and codominant black and scarlet oaks < about 12 inches dbh	9	5	2
Intermediate white oaks	4	3	1
Dominant and codominant white and post oaks > about 12 inches dbh	2	1	
Dominant and codominant white and post oaks < about 12 inches dbh	1	1	1

Our analysis suggested that variables representing stand density such as stems or basal area per acre were not correlated to increased mortality. Our findings are consistent with those of Starkey and Oak (1989) who also found no correlation between stand basal area and oak decline and oak mortality throughout the Central Hardwood Region. Although we cannot conclude whether reducing stand density will reduce oak mortality, we conclude that crown class alone is a better measure of an individual tree's ability to compete for light, water, and nutrients than are measures of stand density, thus explaining the high correlation between oak mortality and crown class in our analysis. Gottschalk and others (1998) found that crown vigor was a better predictor of oak mortality than crown class in stands that were defoliated by gypsy moth. However, crown vigor was not measured in our study so we were not able to evaluate its importance for predicting mortality in our stands that were not defoliated and where gypsy moth does not occur. We do note that Dwyer and others (1995) stressed the importance of crown vigor and measures of crown dieback for predicting oak mortality in the Missouri Ozarks. While measures of crown vigor undoubtedly are useful for predicting mortality of individual trees, foresters often conduct inventories during the dormant season and are unable to judge crown health or vigor. Our analysis identified the variables from a list of those routinely measured by foresters that are most useful for predicting mortality.

No attempt was made to determine the cause of death for the 3,300 trees that died. Such determinations are complex when multiple agents such as drought, competition, insects, and *Armillaria* root disease are involved. We can only speculate that the mortality that we observed was attributable to a combination of mortality induced by competition and mortality induced by other mechanisms including disease and oak decline. Trees in smaller diameter classes or lower crown classes most likely died from competition (self thinning of the stand) because they were shaded by dominant and codominants. This is particularly evident in the high mortality rates for intermediate and suppressed shade-intolerant species such as scarlet oaks, post oaks, and black oaks. However, we suspect that the disproportionately large number of scarlet oaks and black oaks in dominant or codominant crown classes that died most likely did so because of oak decline. This is evident for dominant and codominant black and scarlet oak on the MOFEP sites in the high mortality rates that increase with increasing tree DBH (fig. 2). Mortality rates for these two species are uncharacteristically high relative to rates for all Missouri from 1959 to 1972 (fig. 3).

For thinning and stand improvement operations table 2 provides a priority list for trees to remove. Operationally there is little reason to harvest suppressed (or in some cases even intermediate) trees. They will be eliminated naturally through competition and the associated mortality rates in table 2 simply indicate how many trees are likely to be gone at the end of the next decade. Table 2 clearly indicates the value in removing dominant and codominant black and scarlet oaks larger than 12 inches DBH. Those subpopulations of trees have mortality rates of about 20 percent per decade. Dominant black and scarlet oaks that are smaller in diameter have a 9 percent mortality rate. That is in contrast to dominant or codominant post oaks or white oaks that have mortality rates in the range of 1 to 2 percent per decade. The cumulative mortality column in table 2 indicates the expected mean mortality rate for that line and all groups listed below it (i.e. the rate of all classes above are harvested). The cumulative mortality values are based on averages for all trees and all sites. In application the species composition and tree size structure will vary among individual stands as will the associated mortality. The column of table 2 labeled mortality change is the difference in the cumulative mortality for successive entries. Large values indicate, based on the MOFEP data, which categories listed in the table have the greatest impact on the overall mortality rate.

Our findings substantiate the actions of many foresters in Missouri who have begun the practice of marking for removal most sawlog-sized scarlet oaks and black oaks during stand improvement cuttings in even-aged stands or during selection harvesting in uneven-aged stands. Recent experience has taught them that large individuals of these species are not likely to survive until the next re-entry. This is a reasonable management strategy even if the individual tree is not showing dieback symptoms or other indications of decline because the onset of symptoms and mortality can occur rapidly and affect dominant and codominant scarlet and black oaks. Healthy white and post oaks with good crown structure and competitive position are good candidates for retention in a thinning or stand improvement operation.

Acknowledgements

We thank the Forestry Division and the Resource Science Division of the Missouri Department of Conservation for their cooperation and for partial support for this study.

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DECLINE OF RED OAKS IN THE MISSOURI OZARKS: THE STORY CONTINUES

Steven L. Voelker and Rose-Marie Muzika[†]

Recent studies and anecdotal evidence indicate an increase in the occurrence and persistence of oak decline in many of the regions formerly dominated by shortleaf pine in the Missouri Ozarks. It is yet uncertain whether growth declines and tree dieback are most influenced by stand density, age, drought stress or biotic factors such as *Armillaria* root disease or insects. A survey of tree condition and growth rates was conducted in southeastern Missouri to ascertain which conditions may interact to predispose the current and future forests to oak decline. In order to compare individual tree growth declines, four stand-level considerations were held relatively constant: (1) species composition (2) stand density (3) cohort age and (4) site index. The upland oak-pine forests of the Missouri Ozarks have undergone dramatic and continual change since the exploitative logging period of a century ago. Within oak-dominated stands in this study, evidence such as shortleaf pine remnants or shortleaf pine stumps indicates a more prominent historic role for pine in stands occurring on upper slope positions and exposed sites. Increment cores were analyzed to establish age-structural relationships for the upland oak-pine stands. As expected, most stands are either even aged or two-aged. Dendrochronological modeling was used to establish growth declines (basal area increment) of individual trees. Tree condition and dendrochronological analysis indicate that although aspect and slope position affect site index or tree growth rate, symptoms of decline at the stand level are not exclusively associated with exposed sites. Symptoms of decline vary widely within most stands due to competitive position and tree age.

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EFFECT OF CROWN RELEASE ON TREE GRADE AND DBH GROWTH OF WHITE OAK SAWTIMBER IN EASTERN KENTUCKY

Gary W. Miller and Jeffrey W. Stringer[†]

ABSTRACT.—Forest managers need information on the effect of stand density on residual tree growth and quality to prescribe effective thinning treatments in hardwood stands. Crop tree release treatments were applied in 70- to 75-year-old upland oak stands located in eastern Kentucky. Two treatment levels, applied as a complete crown release of 20 and 34 sawtimber-sized crop trees per acre, were compared to unreleased controls. Data were collected on 12 permanent 0.5-acre plots, with 4 plots randomly assigned to each of the two treatment levels and untreated controls. Tree quality and DBH growth were monitored for 17 years on 148 white oak crop trees. Average DBH growth of released trees in both release treatments was 27 percent greater than that observed for unreleased controls. After 17 years, 89 percent of selected crop trees were in grades 1 or 2, compared to 53 percent at the beginning of the study. No significant differences were found between tree grade distributions on the treated and control plots, indicating that a complete crown release did not adversely affect tree grade.

Numerous studies have shown that commercial thinning in hardwood stands 1) concentrates growth on the most desirable residual trees, 2) reduces the time required to yield products of a desired size, and 3) increases the total yield by harvesting and utilizing trees that would die in the absence of thinning (Miller 1997; Nowak 1996; Oliver and Larson 1996). Although thinning to an appropriate residual density clearly increases growth and yield, some observers caution that thinning too heavily or removing desirable crop trees during thinning can reduce the value yield of hardwood stands by adversely affecting the average quality of the residual trees (Dale 1972).

Current thinning guidelines for upland oaks recommend leaving a residual stand density of 60 to 70 percent of full stocking to maximize wood volume production (Roach and Gingrich 1968). The guidelines also recommend removing poorer quality trees and retaining better quality trees to allow value growth to accrue on trees with the highest potential rate of return. The traditional method of applying a thinning treatment involves removing less desirable trees throughout the stand on an area-wide basis to achieve the target level of average residual stand density. This study tested an alternative method of thinning by applying a crown-release treatment to individual crop trees with the potential for yielding high-quality, high-value sawtimber and veneer logs. The crown-release method of thinning involves removing only competing trees whose crowns are touching the crowns of selected residual crop trees.

Area-wide and crown-release treatments differ with respect to two important questions. 1) Is the focus on finding undesirable trees to cut or desirable trees to retain? 2) Is the focus on reducing competition throughout the stand or reducing competition around individual trees? In area-wide treatments, the emphasis is on searching for and removing trees of poor quality, poor form, or low vigor to achieve a recommended average residual stand density. Note that area-wide treatments distribute removals throughout the stand, thus reducing competition over a relatively large area. A disadvantage of the area-wide approach is that little direct effort is made to improve the microsite conditions around the best residual trees, thus some desirable trees may receive little or no benefit from the thinning operation. Alternatively, the emphasis in crown-release treatments is on identifying the most promising residual trees and then removing competing trees in their immediate vicinity. This approach assures that site resources (sunlight, water, and soil nutrients) freed by the thinning treatment are directly allocated to the designated

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crop trees. An advantage of the crown-release approach is that the residual crop trees, selected because they best meet management objectives, fully benefit from the thinning operation.

In practice, control of average stand density can be used in conjunction with the crown-release approach by varying the number of crop trees released per acre (Perkey and others 1994), thus providing a means for managing the allocation of site resources at both the stand and individual-tree levels.

The objective of this study was to determine the effect of crown-release treatments on the growth and quality of desirable white oak crop trees. Preliminary results on DBH (diameter at breast height) growth and the cost of treatments were reported by Stringer and others (1988a).

Stand Description

The study was conducted at the University of Kentucky's Robinson Forest in eastern Kentucky. The 70- to 75-year-old stands were dominated by upland oaks. The initial basal area averaged 111 ft²/ac (stems = 1.0 inch DBH), and white oak (*Quercus alba* L.) occupied 58 percent of the initial stand (fig. 1). Other stand components included black oak (*Q. velutina* Lam.), yellow-poplar (*Liriodendron tulipifera* L.), American beech (*Fagus grandifolia* Ehrh.), and hickories (*Carya* spp.). The overstory included an average of 67 dominant and codominant trees/ac that were either large pole or small sawtimber with an average DBH of 13 inches. White oak site index averaged 73 (base age 50 years). There was no evidence of recent disturbance and the stands were essentially 100 percent stocked before treatment (Roach and Gingrich 1968).

Study Methods

The long-term management goal for the study sites was to have a fully stocked stand in which codominant trees averaged 22 to 24 inches DBH at final harvest and occupied approximately 80 percent of full stand stocking (Gingrich 1967). Understory trees were expected to occupy the remaining 20 percent stocking. Equations developed by Gingrich (1967) indicated that the number of 24-inch trees needed to occupy 80 percent of full stand stocking varied from 20 to 35 trees per acre, depending on the degree of crowding and its effect on crown size. Two release treatments, defined as 20 and 34 crop trees per acre, and an untreated control were replicated 4 times each and randomly distributed among 12 experimental areas. Each area was two acres in size, and a square 0.5-acre measurement plot was centered in each area. All trees ≥ 1.0 -inch DBH within the 0.5-acre measurement plot were tagged for long-term monitoring. Crop trees within each area were selected based on the following criteria:

1. Dominant or codominant crown class.
2. White oak.
3. Potential grade 1 or 2 butt log.
4. Even spacing within the stand.

Crop trees were released by felling competing trees with a chain saw. Competing trees were defined as adjacent trees whose crowns touched those of the selected crop trees. After release, crop trees were free to grow on all sides. Crop trees selected in control areas were monitored but not released. DBH, crown class, tree grade (Hanks 1976), and remarks were recorded for each crop tree before treatment and again after 17 years. Additional data on epicormic branches and length of clear sections on the 16-foot butt log also were recorded for each crop tree during the final inventory.



Figure 1.—White oak crop trees were monitored for 17 years in eastern Kentucky.

A one-factor repeated measures ANOVA was used to examine the effect of crown release treatments on DBH growth. The fixed effect model has the form:

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

where Y is DBH growth; μ is the overall mean; α is the effect of release treatment; β is the effect of time; and ϵ is the random error. The general linear models procedure in SAS was used for all statistical analyses (SAS Institute Inc. 1998). The Tukey-Kramer HSD mean separation test was used for all multiple comparisons. Treatment effects were considered to be statistically significant when $P < 0.05$. For each analysis, the residuals were tested for normality using the Shapiro-Wilk test and for homogeneity of variance using the Levene test.

The counts of sawtimber trees (≥ 11.0 inches DBH) in each tree grade for released and control plots were compared before and 17 years after treatment using a Pearson χ^2 test to determine if the grade distributions were not equal due to the release treatment.

Results and Discussion

The crown-release treatments reduced residual stand stocking to an average of 76 percent (table 1), which was a relatively light thinning treatment as defined by Roach and Gingrich (1968). As competing trees were felled to release selected crop trees, a relatively small amount of basal area was damaged (Stringer and others 1988b). The 34- and 20-crop-tree treatments removed an average of 75 trees/ac and 54 trees/ac, respectively. The 34-crop-tree treatment also removed more basal area, cubic volume, and relative density compared to the 20-crop-tree treatment (table 1). Removals are usually positively related to the number of crop trees released per acre (Perkey and others 1994). In this case, however, the number of crop trees released per acre did not have a significant effect on residual stand stocking because plots in the 34-crop-tree treatment were slightly overstocked when the study began. The additional trees removed in the 34-crop-tree treatment were mostly smaller pole timber trees, thus releasing additional crop trees did not have a proportional effect on residual stand density. After treatment, all crop trees were free to grow on all sides.

The initial DBH of crop trees ranged from 8.3 to 18.9 inches. Released trees grew an average of 3.1 inches DBH, while unreleased trees grew 2.4 inches DBH over the 17-year study, an average increase of 27 percent due to crown release (table 2).

A total of 153 crop trees were selected at the beginning of this study, and 148 trees were still alive after 17 years. Of the 5 trees that died, 3 were in control plots and 2 were in treated plots. There was no indication that the release treatments affected mortality.

A summary of the changes in tree grade for the surviving 148 crop trees is presented in Table 3. DBH is a main factor that determines tree grade (Hanks 1976), so changes in tree grade were stratified into size classes that correspond to minimum DBH requirements for each tree grade: 9.6 inches DBH for grade 3, 12.6 inches DBH for grade 2, and 15.6 inches DBH for grade 1. Several strata had sample sizes large enough to extract some meaningful results. For example, 10 control trees and 26 treated trees were initially limited by DBH to grade 3 and then grew large enough to enter grade 2. Note that 7 of 10 control trees and 25 of 26 released trees improved in grade as they grew large enough to enter a higher grade. Similarly, the majority of trees in initial size class 2 that grew to ending size class 1 improved in grade for both control and released trees.

The release treatments did not reduce tree quality or prevent otherwise good trees from increasing in grade as they grew. Although the release treatments resulted in relatively small reductions in overall stand density, the full crown release resulted in maximum reductions in density in the immediate vicinity of selected crop trees. The data indicated that among desirable crop trees with excellent initial quality and vigor, a full crown release stimulated faster DBH growth but did not diminish the quality of the butt log, thus there was no adverse effect on tree grade.

Table 1.—Per acre stand data by thinning treatment for upland oaks in eastern Kentucky.

Inventory	Number of stems			Basal area			Volume		RD ³
	≥1.0 ¹	≥5.0	≥11.0	≥1.0	≥5.0	≥11.0	≥5.0	≥11.0 ²	≥1.0
	No./ac			ft ² /ac			ft ³ /ac	bf/ac	%
34 Crop Trees Per Acre (n=4 plots)									
Initial	782	180	42	109.0	93.4	48.2	1,982	6,274	107
Cut	74	57	18	35.3	34.8	20.2	776	2,489	30
Residual	708	123	24	73.7	58.6	28.0	1,206	3,785	77
17-year	753	155	39	104.6	83.5	51.0	1,790	7,287	104
20 Crop Trees Per Acre (n=4 plots)									
Initial	732	160	47	103.4	90.5	51.6	1,975	6,679	100
Cut	55	39	15	29.6	29.0	18.3	677	2,578	25
Residual	677	121	32	73.8	61.5	33.3	1,298	4,101	75
17-year	657	123	40	95.4	76.6	51.4	1,711	7,462	94
Control (n=4 plots)									
Initial	654	138	60	110.6	99.1	71.6	2,296	10,045	102
17-year	643	128	67	128.1	112.7	92.6	2,705	13,849	115

¹Minimum DBH of trees included in total.

²Sawtimber volume, International 1/4-inch rule.

³Relative density, or stand stocking as measured by Gingrich (1967).

Table 2.— Average 17-year DBH growth of white oak crop trees in eastern Kentucky.

Initial Size Class	Treatment	N	Initial DBH	17-yr ¹ DBH Growth ¹	
				inches	
DBH < 9.6	Control	3	8.6	2.51 a	
	20 Crop Trees/ac	2	8.8	2.49 a	
	34 Crop Trees/ac	21	8.3	2.84 b	
9.6 ≤ DBH < 12.6	Control	12	11.4	2.49 a	
	20 Crop Trees/ac	17	11.3	3.51 b	
	34 Crop Trees/ac	17	10.7	3.12 b	
12.6 ≤ DBH < 15.6	Control	20	13.9	2.53 a	
	20 Crop Trees/ac	11	13.9	3.26 b	
	34 Crop Trees/ac	19	13.8	2.88 b	
DBH ≥ 15.6	Control	15	17.2	2.27 a	
	20 Crop Trees/ac	5	18.9	3.10 b	
	34 Crop Trees/ac	6	17.3	3.20 b	
All Crop Trees	Control	50	14.0	2.44 a	
	20 Crop Trees/ac	35	13.1	3.32 b	
	34 Crop Trees/ac	63	11.5	2.96 b	

¹Means followed by the same letter are not significantly different at the 5 % level of significance.

Table 3.—Change in tree grade for white oak crop trees after 17 years in eastern Kentucky.

Initial Size Class	Ending Size Class	Initial Tree Grade	N	Control			N	Thinned		
				Change in Grade				Change in Grade		
				Increase	Same	Decrease		Increase	Same	Decrease
							number of trees			
4	4	4	1	0	1	0	4	0	4	0
4	3	4	1	0	1	0	16	12	4	0
4	2	4	1	1	0	0	3	3	0	0
3	3	3	2	0	2	0	1	0	1	0
		4	0	—	—	—	1	1	0	0
3	2	3	10	7	3	0	26	25	1	0
3	1	3	0	—	—	—	5	5	0	0
2	2	2	3	0	3	0	3	0	3	0
		3	3	2	1	0	1	0	1	0
2	1	2	6	6	0	0	18	16	2	0
		3	8	6	2	0	9	7	2	0
1	1	1	3	0	3	0	6	0	6	0
		2	11	9	2	0	4	3	1	0
		3	1	1	0	0	1	0	1	0

¹Size Class 4 (DBH < 9.6); Size Class 3 (9.6 ≤ DBH < 12.6); Size Class 2 (12.6 ≤ DBH < 15.6); Size Class 1 (DBH ≥ 15.6); Size classes represent DBH thresholds for tree grades 4,3,2,1, respectively. Grade 4 collectively represents trees below sawlog quality.

A total of 103 crop trees were sawtimber size (≥ 11.0 inch DBH) before treatment. The grade distributions of released versus control trees were compared before and after treatment. The percentage of crop trees in grades 1, 2, and 3 did not differ significantly before (P = 0.813) or 17 years after (P = 0.568) the crown release treatment (table 4). Due to size limitations, the percentage of trees in grade 1 did not exceed 10 percent when the study began. After 17 years of growth, the percentage of trees in grade 1 was nearly 50 percent for both control and released crop trees. The percentage of trees in grade 2 was slightly greater for released crop trees compared to controls, probably because more released trees reached the minimum size for grade 2 due to faster growth. The data suggested that faster growth following release reduced the time required for crop trees to reach the minimum DBH required to enter higher grades compared to controls.

In addition to DBH, surface characteristics of the 16-foot butt log of the tree are also important factors that affect grade. Epicormic branches that form on the butt log can diminish quality and value because they reduce the yield of defect-free wood. Logs with no branches or defects are much more valuable because they can yield high-quality lumber or even veneer, the highest value product. The formation of epicormic branches can occur when the quantity or distribution of auxins produced by the tree is altered by a change in light, moisture, nutrients, or temperature after a disturbance (Bowersox and Ward 1968). In this study, thinning did not increase the number of epicormic branches on white oak crop trees (table 5). Note that the crop trees observed in this study were the most vigorous and highest quality trees available when the release treatments were applied. Such trees usually do not lose quality after crown release treatments (Miller 1996; Ward 1966; Dale 1972).

The average number of epicormic branches on the butt log 17 years after thinning was related to the initial DBH of crop trees; small trees had more branches, while large trees had less. Trees that were ≥ 15.6 inches DBH before treatment averaged < 1 epicormic branch 17 years later (table 5). The proportion of crop trees with at least one 8-foot clear section (all four faces) on the butt log also increased as initial DBH increased. For trees ≥ 12.6 inches DBH at the beginning of the study, the percentage of trees with at least one 8-foot clear section on the butt log was 69 percent for control trees and 73 percent for released trees 17 years after release (table 5). This result supports the finding that crown release did not diminish the butt log quality of vigorous, desirable crop trees.

Table 4.—Pearson Chi-square comparison of tree grade distributions for white oak crop trees before and 17 years after thinning in eastern Kentucky.

Distribution	Treatment	Tree Grade				Total	Tree Grade				X ²	Prob
		1	2	3	Total		1	2	3	Total		
		—Number of trees—					—Percents by row—					
Original	Control	3	20	20	43	7.0	46.5	46.5	100	0.414	0.813	
	Thinned	6	25	29	60	10.0	41.7	41.7	100			
17-year	Control	21	16	6	43	48.8	37.2	14.0	100	1.130	0.568	
	Thinned	28	27	5	60	46.7	45.0	8.3	100			

Table 5.—Characteristics of the 16-foot butt logs of white oak crop trees 17 years after crown release: average number of epicormic branches and number of trees with at least one clear 8-foot section.

Initial Size Class	Treatment	N	Mean number	Crop trees with 8-foot
			of epicormic	clear section
			branches/tree	
			—no. of branches—	—no. of trees—
DBH < 9.6	Control	3	6.7	1
	20 Crop Trees/ac	2	8.5	0
	34 Crop Trees/ac	21	5.4	3
9.6 ≤ DBH < 12.6	Control	12	3.7	5
	20 Crop Trees/ac	17	3.0	8
	34 Crop Trees/ac	17	2.0	12
12.6 ≤ DBH < 15.6	Control	20	3.1	14
	20 Crop Trees/ac	11	1.1	7
	34 Crop Trees/ac	19	2.8	14
DBH ≥ 15.6	Control	15	0.9	10
	20 Crop Trees/ac	5	0.0	4
	34 Crop Trees/ac	6	1.0	5
All Crop Trees	Control	50	2.8	30
	20 Crop Trees/ac	35	2.3	19
	34 Crop Trees/ac	63	3.3	35

Conclusions

This study shed some light on the effect of thinning on residual tree quality. Results indicated that full crown release of 70- to 75-year-old white oak sawtimber crop trees did not have an adverse effect on butt log quality and tree grade 17 years after the thinning treatments. One reason is that the thinning treatments reduced stand density to an average of 76 percent, thus the removals were relatively light (table 1). Although a slightly lower residual density is recommended (Roach and Gingrich 1968), each released crop tree was free to grow on all sides after release. As a result, the environment around each crop tree was characteristic of a much lower average stand density. Note that released crop trees responded to additional free growing space, and their DBH growth increased by an average of 27 percent compared to unreleased crop trees (table 2).

Although growth increased, no crop trees decreased in grade following release (table 3). The grade distribution for released trees was similar to that of unreleased trees (table 4), and surface characteristics on the butt logs were similar to that of unreleased trees (table 5). However, the data suggested that crop trees with smaller initial DBH had more epicormic branches and fewer clear 8-foot sections after 17 years. Initial vigor, as evidenced by relative DBH or relative crown position, is an important factor that determines the risk of developing epicormic branches (Miller 1996; Brinkman 1955; Smith 1965). It is logical to conclude that the results would have been dramatically worse if the larger, more vigorous trees had been removed during the thinning and smaller, less vigorous trees had been chosen as residual crop trees.

The results support two major recommendations of Roach and Gingrich (1968): 1) The best quality trees of desired species should be retained and trees of poor quality or undesirable species should be removed. 2) The residual stand density should be reduced to approximately 60 to 70 percent stocking as measured by the upland hardwood stocking guide (Gingrich 1967). They also recommended that an effort should be made while marking the cut trees to provide desirable residual crop trees with a full crown release. A general conclusion of this study is that releasing approximately 35 crop trees per acre in sawtimber white oak stands similar to the study area would result in a residual stand density consistent with current recommendations, provide a significant increase in DBH growth, and produce no adverse effect on butt-log quality or tree grade of the released crop trees.

Acknowledgments

This study was partially funded by National Hardwood Lumber Association, 6830 Raleigh-LaGrange Road, Memphis, TN 38184.

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EFFECT OF SEASON OF CUTTING ON THE PRODUCTION AND GROWTH OF WHITE AND SHUMARD OAK STUMP SPROUTS IN TENNESSEE

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ABSTRACT.—The effect of season of cutting on the stump sprouting of white and Shumard oaks was investigated. Replicate plots were cut at 2-month intervals from January through November, 2001. The number of sprouts and the heights and basal diameters of the three tallest sprouts per stump were recorded bi-monthly after cutting. After 4 months of growth following cutting, mean heights of white and Shumard oak sprouts were substantially greater in the September, November, and January cutting treatments than in the May treatment, with nearly a two-fold difference in mean height of Shumard oak sprouts between the September and May treatments. Similarly, mean basal diameters of Shumard oak sprouts were significantly greater in the September and November cutting treatments than in the May treatment. The greatest mean numbers of white and Shumard oak sprouts were produced in the July, September, and November treatments, but differences were not statistically significant. These patterns likely resulted from seasonal differences in stored carbohydrate reserves, and suggest that late summer, fall, and winter cutting is preferable to spring and early summer cutting in order to regenerate oak by coppicing, or to maximize the growth of sprouts from stems damaged during harvesting.

Oak stump sprouts are an important means of regenerating oak stands whenever oak advance reproduction from seed is lacking and survival of oak seedlings and saplings into larger size classes is low due to interspecific competition and additional factors such as deer browsing. The rate of height growth in oak stump sprouts often exceeds that of oak seedlings and saplings originating from seed as a result of the extensive residual root systems of stumps, which contain large amounts of stored carbohydrate reserves (Johnson 1979, Reich et al. 1980, Cobb et al. 1985). The rate of height growth in oak sprouts is very important during the first growing season for establishing their competitive position in the regeneration layer, and also influences the amount of time they are near the ground and are most susceptible to deer browsing and late frosts. The relationship between stump diameter and the rate of stump sprout growth has been investigated in several studies. Sprout growth generally increases as stump diameter increases (Sander 1971, 1972, Johnson, 1979, Reich et al. 1980). The correlation between stump diameter and sprout growth can weaken, however, at diameters greater than 10-20 cm (Johnson 1979, Reich et al. 1980). Working with sprouts from large oak advance reproduction, Sander (1971) recommended ground-line diameters of oak advance reproduction between 1.27 and 2.54 cm in order to produce sprouts with sufficient growth rates to be competitive with sprouts and saplings of other species, yet limited in number. Oak species have a very large number of buds or bud bank that facilitates regrowth of stems, branches, and foliage following a variety of injuries (Ward 1964, Wilson and Kelty 1993). The ability of oak stumps to produce desirable sprouts, however, diminishes with increasing stump diameter and age (Johnson 1977, McGee 1978, Weigel and Johnson 1998). Long-term survival of hardwood stump sprouts is influenced by a number of factors, including intra- and interspecific competition, the type and location of buds producing sprouts, and the ability of the overall sprout clump to maintain the residual root system (Wilson 1968, Kramer and Kozlowski 1979, Johnson et al. 2002). A number of stump sprout thinning or grooming studies (Haney 1962, Lamson 1983, Johnson and Rogers 1984, Lowell et al. 1989) have been conducted to explore means of increasing the growth, form and survival of one or a small number of sprouts on a single stump.

Due to the apparent correlation between stump sprout growth and amounts of carbohydrate reserves stored in residual stump root systems, it can be hypothesized that additional factors affecting amounts of stored reserves could also affect sprout growth, regardless of stump diameter and the associated size of the residual

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root system. For example, cutting during months when stored carbohydrate reserves are at their peak should result in the greatest sprout growth rates. Observations of vigorous sprout growth following cutting of oak stems in late summer and early fall provided the impetus for formally testing effects of seasonal differences in cutting on the number and growth of oak stump sprouts produced by sapling-sized white oak (*Quercus alba* L.) and Shumard oak (*Quercus shumardii* Buckl.).

The general objective of this research was to compare effects of cutting in January, March, May, July, September, and November on the production and growth of oak stump sprouts. Specific objectives were to: 1) Test the hypothesis that greater growth per sprout would be obtained with cutting in the fall months (September through November) than in the remaining seasons, 2) To compare the sprout growth response of two congeneric species, white oak and Shumard oak, to cutting in different months, and 3) To investigate relationships between the growth of sprouts and stump basal diameter, and between sprout growth and the point of attachment on the stump.

Study Areas

This study utilized monocultures of 240 white and 240 Shumard oak planted in 1992 for another study in a 41.48 x 75.64 m area located within a level, mown field on the University of Tennessee, Cumberland Forestry Experiment Station (CFES) in Morgan County, Tennessee. The portion of the CFES utilized for this study lies near the base of Little Brushy Mountain, which falls within the Cumberland Mountain section of the Appalachian Plateau Province (Fenneman 1938). Annual weather records for CFES from 1957 to 1989 indicate that the study site receives, on average, 1,462 mm of precipitation annually, and that daily average temperatures in January and July are typically 2.6 and 24.4 °C, respectively. Soils belong to the Lily Loam Series, and are classified as Fine-loamy, siliceous, mesic, Typic Hapludults.

Methods

In 1992, nursery seedlings of each species were planted on a 2.44 x 2.44 m spacing within each 20.74 x 75.64 m rectangular half of the overall 41.48 x 75.64 m area planted. Stems for both species averaged 6 cm DBH prior to implementation of treatments for the present study. Eighteen plots of 10 stems (2 x 5 trees) were delineated within each monoculture (white or Shumard oak) in fall 2000. Prior to treatment implementation, there were no statistically significant differences in mean DBH or mean basal diameter at 17 cm above the ground between any of the 18 white oak or 18 Shumard oak plots. Treatments consisted of 6 cutting dates (January, March, May, July, September, and November 2001) that were assigned at random to the 18 plots for each species. All treatments were assigned to three replicate plots of 10 trees for each species, creating a completely randomized design. Cutting was carried out during the last week of each cutting month. All stems were cut with a chainsaw approximately 15 cm above the ground.

Prior to treatments in fall 2000, DBH of all white and Shumard oak stems was measured to the nearest mm with a diameter tape, and basal diameter of all stems 17 cm above the ground was measured to the nearest cm with calipers. Following implementation of a given treatment and the emergence of stump sprouts, sprout and sprout clump characteristics were quantified during the 2001 and 2002 growing seasons (May-September) on a bimonthly schedule patterned after the treatment schedule. The total number of sprouts produced by each stump were counted, and the 3 tallest sprouts per stump were tagged and measured. Sprout height was measured with a meter stick to the nearest cm between the point of sprout origin on the stump and the terminal bud of the terminal leader. Sprout basal diameter was measured to the nearest mm with vernier calipers 4 cm above the point of sprout origin on the stump. Locations of sprouts on stumps were qualitatively categorized as low, intermediate or high if sprouts originated on the lower third, middle third, or upper third of the stump, respectively.

One-way ANOVA was used to detect differences between treatments in sprout height, basal diameter, and numbers of sprouts after four months of post-cutting growth. Tukey's Honestly Significant Difference (HSD) was used to compare cutting treatment means. Analyses of the effects of cutting treatments on sprout heights, basal diameters, and numbers of sprouts were carried out using plot-level means corresponding to each replicate. Differences between species in sprout heights, basal diameters, and numbers of sprouts were also analyzed within a given cutting treatment using one-way ANOVA and plot-

level means. Simple linear regression was used to investigate relationships between stump basal diameter and height of the tallest sprout on each stump. Regression analyses were run both within cutting treatments and with data pooled over all cutting treatments. Analyses of sprout height in relation to position of sprouts on the stump were conducted using plot-level means for each type of sprout (low, intermediate, or high on the stump). Data were pooled over cutting treatments for this analysis, and only height of the tallest sprout on each stump and only plots containing all three types of sprout were included in the analysis. One-way ANOVA was used to detect differences in sprout height between low, intermediate, and high positions on stumps. Tukey's HSD was used to compare treatment means. Fourth-month measurements of sprouts originating from stumps cut in May 2001 were missed due to the use of a slightly different measurement schedule in the first year of the study. As a result, all analyses of sprout heights and basal diameters described above were conducted using estimates of height and basal diameter for white and Shumard oak sprouts for the May treatment. These estimates were calculated by averaging height and basal diameters taken 3 and 6 growing-season months after cutting. All statistical analyses were carried out with alpha equal to 0.05.

Results

After 4 months of post-cutting growth, mean heights of white oak sprouts were greatest in the September, November, and January cutting treatments (Figure 1a). Mean height of white oak sprouts was least in the May cutting treatment, and significantly less in this treatment than in the January and September cutting treatments. Mean height of Shumard oak sprouts in the September cutting treatment was nearly twice that of mean height in the May treatment (Figure 1a). Mean heights of Shumard oak sprouts were significantly greater than those of white oak within the July, September, and November cutting treatments, and the trend in heights of Shumard oak across treatments was similar to that of white oak, with the exception of the July cutting treatment.

No statistically significant differences in mean basal diameter of white oak sprouts were indicated by ANOVA and Tukey's HSD, but mean basal diameters of Shumard oak sprouts in the September and November treatments were significantly greater than in the January, March, May, and July treatments (Figure 1b). The greater basal diameter growth of Shumard oak sprouts on stumps cut during the dormant season than on stumps cut during the growing season and a similar trend in white oak were consistent with height growth in these species (Figures 1a, 1b). Mean basal diameters of Shumard oak sprouts were significantly greater than those of white oak within the July, September, and November cutting treatments.

White oak produced roughly twice the mean number of sprouts per stump produced by Shumard oak across most treatments (Figure 1c). This difference in sprout production between the two species was statistically significant within the March, May, July, and November treatments. Differences in the mean number of sprouts between cutting treatments were not statistically significant for either species, although the trend toward the least number of sprouts produced in the March and May treatments was fairly consistent with the patterns of sprout height and diameter growth in each species.

In all treatments combined, there were significant positive linear relationships between mean sprout heights of white and Shumard oak and stump basal diameter (Figures 2a, 2b). Coefficients of determination, however, were fairly low (Figures 2a, 2b). Regression analyses conducted within cutting treatments also indicated significant positive relationships between sprout heights and stump basal diameters within most cutting month treatments, and coefficients of determination ranged from 0.30 to 0.52. Overall, mean height of Shumard oak sprouts increased more rapidly with increases in stump basal diameter than mean height of white oak sprouts (Figures 2a, 2b).

In all treatments combined, mean height of white oak sprouts was significantly greater in sprouts originating high on the stump than those originating low on the stump near the root collar (Figure 3). The pattern of Shumard oak sprout heights in relation to position on the stump was very similar to that of white oak, but differences were not statistically significant.

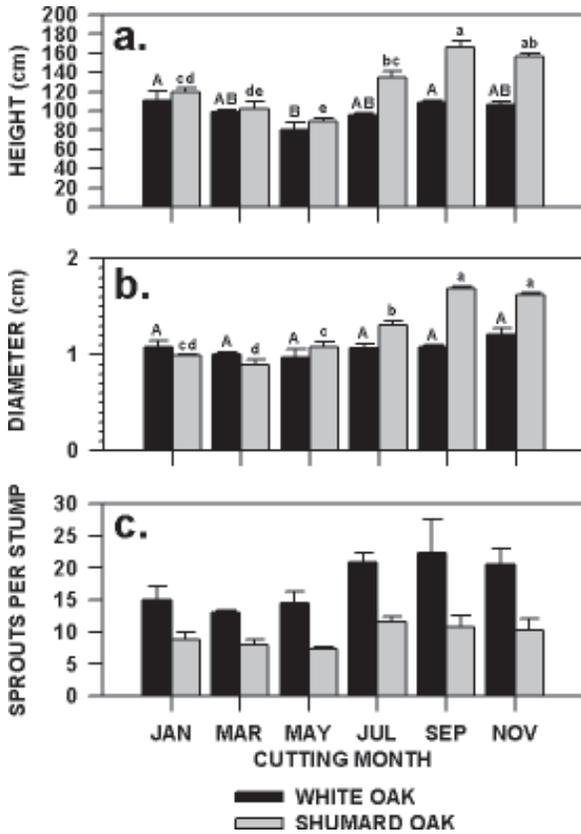


Figure 1.—Mean a.) sprout height, b.) basal diameter, and c.) number of sprouts per stump after 4 months of post-cutting growth in relation to cutting month. Means with the same letters are not significantly different at the alpha = 0.05 level of significance, and no statistically significant differences occurred in numbers of sprouts between treatments.

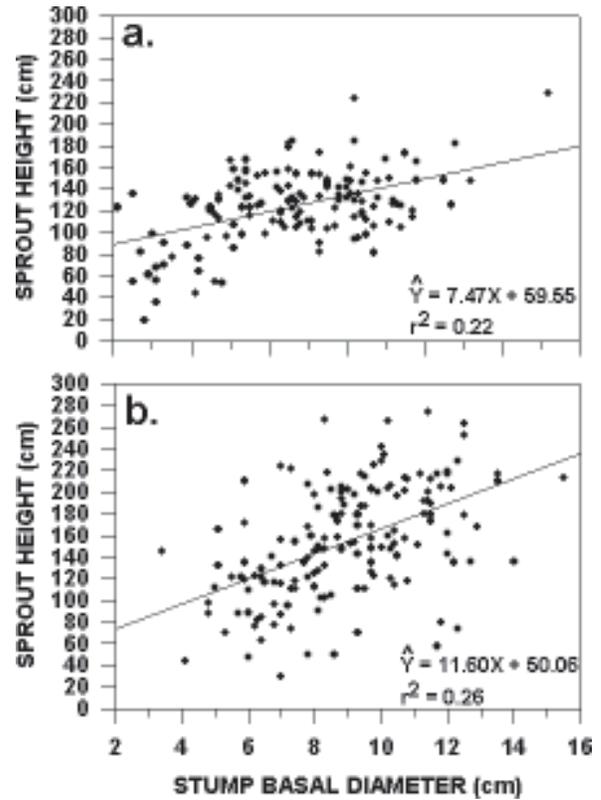


Figure 2.—Relationship between stump basal diameter at 17 cm above the ground and height of the tallest sprout per stump after 4 months of post-cutting growth in a.) white oak and b.) Shumard oak.

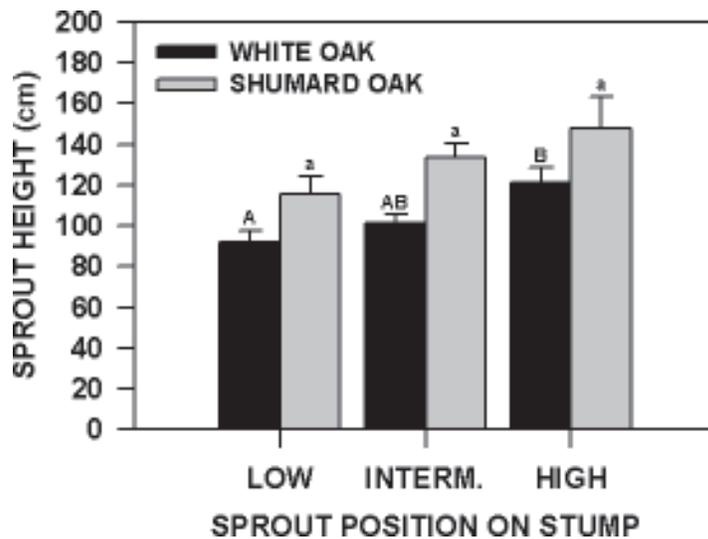


Figure 3.—Mean sprout height after 4 months of post-cutting growth by position on the stump. Means with the same letters are not significantly different at the alpha = 0.05 level of significance.

Discussion

The patterns of height growth, basal diameter growth, and numbers of sprouts observed across cutting months suggest that the overall sprouting capacity of white and Shumard oak stumps was greatest in individuals cut in late summer, fall, and winter. These results are consistent with patterns observed in the sprouting of blackjack oak (*Quercus marilandica* Muenchh.) girdled in different months of the year in Missouri, which Clark and Liming (1953) attributed to differences in the amount of stored carbohydrate reserves available for sprout growth over the course of the year. Wendel (1975) observed greater height growth of red oak (*Quercus rubra* L.) sprouts from stumps cut in the dormant season than sprouts from stumps cut during the growing season, although this difference diminished and was not statistically significant after 10 years.

White and Shumard oak responded similarly to the treatments, but not identically, as evidenced by significantly fewer and larger sprouts produced by Shumard oak than white oak in some treatments. Little (1938) noted a similar difference in which white oak and post oak (*Quercus stellata* Wangenh.) stumps produced greater numbers of smaller sprouts than members of the Red Oak Group, which includes Shumard oak. In addition, height growth of Shumard oak sprouts was significantly greater in the September than in the January cutting treatment, whereas this was not the case in white oak. The greater height and basal diameter growth of Shumard oak in the September and November cutting treatments supports the hypothesis that greater growth per sprout will be obtained if this species is cut in these fall months.

Although coefficients of determination were fairly low, results of the regressions indicate that stump basal diameter is a reasonable predictor of sprout growth in white and Shumard oak. The distribution of data points around the regression line for white oak across the range of stump diameters included in the study indicates that the relationship between stump diameter and sprout height did not weaken as white oak stump diameters approached the 10 cm range, in contrast to results reported by Reich et al. (1980).

The significantly greater height growth of white oak sprouts located high on the stump compared with those low on the stump and the similar but non-significant differences in Shumard oak in relation to stump position suggest that sprouts originating high on the stump were supplied with greater amounts of resources for growth than those originating low on the stump near the root collar. Dormant buds closer to the root collar are older than those higher up on the bole, and the chances of breaks in bud traces tend to increase with age (Smith et al. 1997). Sprouts high on the stump were much more rare than those with low and intermediate points of attachment. Many plots contained only 1-3 measured sprouts that were high on the stump, so these results are based on relatively few sprouts high on the stump per plot and should be interpreted with caution. Although the possibility of greater initial height growth of sprouts originating high on the stump exists, these sprouts are more likely to produce trees with poor form and decay than sprouts lower on the stump (Smith et al. 1997).

Whether sprouts of hardwood competitors of the oak species studied respond similarly to season of cutting remains an important question. Schier and Zasada (1973) documented a positive relationship between the mass of trembling aspen (*Populus tremuloides* Michx.) suckers and amounts of stored reserves. Wilson (1968) observed more rapid growth in red maple (*Acer rubrum* L.) sprouts from stumps cut in the winter than during the growing season, although levels of overstory competition for the two sets of red maple sprouts were described as being very different. Much heavier overstory competition may have contributed to reduced growth of sprouts from stumps cut during the growing season (Wilson 1968). Wenger (1953) observed a different pattern in sweetgum (*Liquidambar styraciflua* L.) in which sprouting was also reduced late in the growing season despite large amounts of stored reserves in the roots, and attributed this to the action of growth regulators.

An important component of the Brose and Van Lear (1998) shelterwood-burn technique for favoring oak over competitors such as yellow-poplar (*Liriodendron tulipifera* L.) and red maple is to burn in the spring in order to maximize the sprouting disadvantage of these species relative to oak species, which invest more heavily in root systems and stored reserves. If cutting alone is used to regenerate stands and yellow-poplar

and red maple behave similarly to the oak species we have studied in their response to season of cutting, it may be beneficial to cut the oak in the fall and delay cutting of these competitors until the spring. Such a technique may be difficult to implement, however, due to potential damage to the developing oak sprouts and other practical concerns. If sweetgum is a dominant competitor and behaves consistently as in Wenger's (1953) study, cutting both oak and sweetgum in the fall would maximize height growth of oak sprouts and coincide with the period of reduced sprouting in sweetgum.

Clearly, these results are short-term, and the long-term survival of sprouts and persistence of these differences in height growth over time remain to be quantified. Shifts in the dominance of red oak sprouts on the same stump have been observed, as well as diminishing differences over time between the heights of red oak sprouts originating from stumps cut during the dormant season and those originating from stumps cut during the growing season (Wendel 1975). It can be argued, however, that rapid first-year height growth of oak stems is critical for subsequent survival of sprouts and sprout clumps on many sites, particularly when competition for light, browsing, and frost damage are intense near the ground. Continued monitoring is planned.

Acknowledgment

We are grateful to Brien Ostby, Research Assistant, for spending many long hours in the field capturing data on large populations of oak sprouts, and to Martin Schubert, Manager of the Cumberland Forestry Experiment Station, for implementing the treatments and monitoring the study site. This study was made possible by McIntire-Stennis funds.

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RESPONSE OF EASTERN BLACK WALNUT TO HERBICIDE STUMP TREATMENT

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ABSTRACT.—In both managed plantations and managed forests, thinnings are often required to maintain or enhance productivity. In managed plantations especially, resulting stump sprouts may be deemed undesirable for aesthetic reasons, as well as inhibiting other management activities, such as harvesting nuts and mowing. Herbicide applications can be an effective method for eliminating sprouts that may develop following tree thinning operations. This study was established during the thinning of an eastern black walnut (*Juglans nigra* L.) plantation, with stumps treated immediately following tree felling. Four treatments, including three herbicides (Garlon 3A®, Banvel® and Roundup®) and a control (no stump treatment) were applied in a randomized complete block design to 36 stumps. Treatments were applied during April of 2001. For 2 years following stump treatment, sprouts were counted. Stumps treated with Garlon 3A® provided the best results with no sprouting over the 2-year study period.

Introduction

Of all the hardwood species, eastern black walnut (*Juglans nigra* L.) is one of the most likely to be planted in a monoculture (single species) plantation setting. As a tree with production opportunities for growing both nuts and timber of high value, it has been widely planted in agroforestry practices seeking diversified production, as well as resource stewardship. However, embodied within the goals of stewardship and productivity is the maintenance of healthy trees and forests. In both managed plantations and managed forests, thinnings are often required to maintain or enhance productivity.

When striving to promote healthy forest stands of desirable species, while at the same time maximizing individual tree growth, timely thinning operations become an invaluable tool for forest managers. In hardwood forest stands, thinnings are usually accomplished by either mechanical means that include cutting or girdling tree stems, by chemical release using injection and/or basal bark spray treatments, or by some combination of the two that will ensure removal of select stems and prevent further regrowth. However, in managed plantations thinning methods that result in stump sprouts may be especially undesirable because of their potential hindrance to other management activities, such as harvesting nuts and mowing. Herbicide applications can be an effective method for eliminating sprouts that may otherwise develop following tree thinning operations.

To best ensure the realization of forest management goals and optimize the use of personnel time, thinnings should be designed to reduce the competition between trees for limited site resources (light, moisture and nutrients). Thinnings that do not eliminate resource competition do not optimize the investment of time, or the likelihood of achieving management goals. This can occur when trees identified for removal during a thinning practice either, do not suffer loss of upper-stem growth, have upper-stem die-back but resprout from the stump, or the main stem is killed but the tree suckers from the root stock. In all cases, the competition for light may have been eliminated, yet competition for moisture and nutrients continues. This does not optimize the outcomes associated with thinning activities. Proper application of herbicides can effectively minimize the likelihood that thinned trees remain in competition for on-site resources.

While other studies, including those by Miller (1993) and Van Sambeek et al. (1995), have identified the effectiveness of injection and basal spray treatments as measured by crown reduction and/or tree mortality, they did not measure herbicide effectiveness based on cut-stump application and a count of resulting stump

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sprouts. However, these studies have identified differences in effectiveness based on species, and diameter within a species. The Miller study looked at the response of sweetgum to applications of Pathway, Garlon 3A, and Arsenal AC, while the Van Sambeek study was more generally applied to hardwoods and compared herbicides containing Glyphosate, Dicamba, and Dicamba+2,4-D. Both studies reported varied success depending on the application method, and neither reported specifically on the response of eastern black walnut. Additionally, the Walnut Council Bulletin has published at least two landowner reports that detail the use of Roundup® in thinning operations (Pannill (1997) and Merrill (2002)). These reports provided results that also seem to present limited success at completely deadening selected trees through herbicide application.

Due to the number of chemicals available to landowners and professionals, and the various methods by which they may be applied, a controlled comparison of the effectiveness of chemicals can be useful to land managers. This study was designed to evaluate the effectiveness of three herbicides to control stump sprouting after the conventional thinning of a black walnut plantation. Following tree felling with a chainsaw, cut surface application of 3 herbicide formulations was carried out according to label recommendations. It is important, for safety and to maximize the effectiveness of the herbicide on the intended target plant, that label recommendations should always be followed.

Study Site

Located in Southwest Missouri, the Sho-Neff Black Walnut Farm was established in 1975. It is currently owned and managed by the Hammons Products Company of Stockton, Missouri. The farm totals 480 acres and is divided into 25 areas. The stump treatment thinning was applied in area 16B which was planted to black walnut in 1976. With the primary goal of nut production, trees were planted at an initial spacing of 20 by 40 feet. Agroforestry was practiced on the site in years 1 through 11, with plantings of soybean, wheat and milo produced in the 40-foot alley ways. A thinning was conducted in 1998-99 to remove inferior trees, with an additional thinning occurring in 2000 to maintain growth rates on the residual trees.

Methods

Four treatments were assessed for their effectiveness in minimizing stump sprouting following thinning. These included (1) a control with no chemical stump treatment, (2) Garlon 3A® having 44.4% Triclopyr as the active compound, (3) Banvel® having 48.2% Dicamba as the active compound and (4) Roundup® having 41% Glyphosate as the active compound. Herbicide sprays were applied full strength in order to test the maximum effect that each would have on stump sprouting and the growth of adjacent trees.

Located across seven rows (40 feet between rows) within a black walnut plantation, 36 trees previously marked for removal during a thinning were cut in April of 2001, and one of the four sprouting control treatments applied. Trees were cut with a chainsaw, and application of the treatment occurred immediately following each cut. At the time of felling, the DBH (diameter at breast height, 4.5 feet above ground line) of each study tree was measured. Each stump was cut low to the ground, at an approximate height of no greater than 3 inches. Using an adjustable spray bottle that held 1-quart of chemical, herbicide was sprayed to cover the outer 2 inches of tree growth. Thinned trees ranged in size from 6.6 inches to 12.3 inches in diameter at breast height (DBH). However, the difference in mean DBH across all treatments varied by approximately 1-inch or less (Table 1).

Using a randomized complete block design, treatments were applied to the trees marked for thinning. As many replicates as possible were applied within each tree row. A replicate consisted of all herbicide stump treatments and a stump not treated (control), each randomly ordered within a row. This was done nine times and represents three blocks of treatments. Although slope is very slight (3% or less), micro-site variability and the unbiased assignment of each herbicide treatment to a stump is best ensured through use of the randomized complete block design and layout. Each cut-stump treatment was applied a total of nine times over seven rows.

Sprouts from the stump of each tree were counted twice, first in July of 2001 and again in January 2003. Statistical analysis was conducted using GLM (general linear means) as computed by SAS (1999), to

Table 1.—Comparison of the mean number of stump sprouts during years 1 and 2 for walnut trees of a given mean Diameter at Breast Height (DBH) following felling and stump treatment of either Garlon 3A®, Banvel®, Roundup®, or left untreated (Control).

Cut Stump Treatment ¹	Active Ingredient	Mean	DBH		Mean Number of Stump Sprouts ²	
			— inches —	Range	Year 1	Year 2
Control	no chemical treatment	10.47	9.3 – 12.1		5.7 A	5.7 B
Garlon 3A®	44.4% Triclopyr	10.92	9.6 – 12.3		0 B	0 C
Banvel®	48.2% Dicamba	9.83	8.7 – 11.6		2.3 B	2.8 B
Roundup®	4 % Glyphosate	10.56	8.4 – 12.3		0.2 B	2.9 B

¹n for all treatments is nine

²Mean number of sprouts followed with the same letter within a given year are not significantly different at alpha = 0.05 as compared by Duncan’s Multiple Range Test.

determine whether one stump treatment was superior to the others at reducing the number of stump sprouts. Means of the number of sprouts occurring in year 1 and year 2 were compared using Duncan’s Multiple Range Test at alpha = 0.05.

Results

The control treatment (no herbicide applied to the cut surface) had the greatest number of sprouts in both years 1 and 2, with a range of 2-18 sprouts in year 1 and 2-10 sprouts in year 2. By comparison, stumps treated with Garlon 3A® produced no sprouts during the two-year study.

Due to the fact that Garlon 3A® had zero sprouts on all stumps for years one and two of the study, the mean and variance for both years was also zero. This made statistical comparison to other treatments difficult. However, it is obvious that when compared to treatments that developed stump sprouts, the complete control of sprouting by Garlon 3A® is significantly different.

Removing Garlon 3A® (with zero mean and variance), the influence of DBH and chemical treatment on sprouting were assessed using GLM. The model output identifies only the chemical treatment as being significant at an alpha=0.05, or 95 percent confidence level. In year one, no significant differences were found between the other two treatments receiving herbicides, yet both were significantly different from the control (no herbicide) (Table 1). The same trend held for year-two, with the Banvel® and Roundup® treatments identified as similar, yet significantly different from the control. As the mean number of sprouts per stump increased for tree stumps in the Banvel® and Roundup® treatments, the difference with Garlon 3A® also increased as compared to all other treatments (Table 1).

Discussion

Properly applied, herbicides can be an effective tool for optimizing a thinning operation by eliminating, or reducing, the sprouting of stems cut during a thinning. Numerous studies have examined the use of chemicals for their effectiveness when applied as cut-surface, injection (of which “hack and squirt” is a form) and basal sprays. Thomas et al. (1988) used several cut-surface treatments on sugar maple stumps in an effort to eliminate sprouting. They identified a change over time in the number of sprouts, with stumps that initially appeared dead (without sprouts), developing sprouts in the second year. Tordon RTU® and Garlon 3A® were identified as maintaining good control of sugar maple for at least two years in the study.

However, in plantings of like-species trees, there should always be a concern with flashback when applying herbicides to thin trees in plantations. Personal communication with various forest resource professionals resulted in recommendations to avoid Tordon RTU® in black walnut plantations due to flashback potential. Flashback is the unintended negative impact of chemical application on trees adjacent to those treated during the thinning process. This occurs when a chemical translocates from the stem into the root

system and via root grafting, moves into an adjacent tree. Often flashback is first evidenced by the dieback or yellowing of the foliage of adjacent trees. We observed flashback in two separate cases. One resulted from a stump treated with Banvel® and was evidenced by approximately 20% crown dieback of the adjacent tree. The second occurred with Roundup® and caused 30-40% crown dieback. Neither tree died, but when a thinning is designed to enhance growth and development of released crop trees, their foliage dieback due to flashback is undesirable. Only the control and Garlon 3A® treatments did not result in any incidents of flashback.

Our study demonstrates significant differences between herbicides in controlling walnut stump sprouting. Furthermore, it is clear from our results that 1 year is an insufficient timeframe within which to evaluate the effects of herbicides. While all three herbicides tested were found to greatly reduce sprouting after 1 year from that observed when no herbicide was applied (control), no differences were identified between the herbicides. However, as a result of the recovery of stumps treated with Banvel® and Roundup®, both had an increase in stump sprouts occurring after 2 years, and results then indicated differences with the absence of sprouts on stumps treated with Garlon 3A®. Clearly, Garlon 3A® was the superior herbicide tested in this trial and a minimum of 2 years following the application of the chemical was required to make a valid comparison.

Acknowledgments

This work was funded through the University of Missouri Center for Agroforestry under cooperative agreement AG-02100251 with the USDA ARS Dale Bumpers, Small Farms Research Center, Boonville, AR. The results presented are the sole responsibility of the P.I. and/or MU and may not represent the policies or positions of the ARS.

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SILVICULTURAL TREATMENT EFFECTS ON HARDWOOD TREE QUALITY ON THE VINTON FURNACE EXPERIMENTAL FOREST

John Brown, Janice K. Wiedenbeck, Rado Gazo, and Daniel A. Yaussy[†]

ABSTRACT.—This study examined the effects that silvicultural treatments have on the tree grade distribution of a mixed hardwood stand located on the Vinton Furnace Experimental Forest. Four stand treatments were established: a commercial clearcut, a commercial clearcut with timber stand improvement (TSI), a selection cut, and a selection cut with TSI. Trees were found to be of better quality in the selection cuts, where the odds of a tree being classified into the same reference grade or higher grade vs. lower than the reference grade were found to be 1.5 to 12.3 times the odds of a tree in the commercial clearcut being classified into the same reference grade or higher grade vs. lower than the reference grade. As a group, stands where TSI was applied had different odds for better quality trees than did stands without application of TSI. Within the commercial clearcut treatments, TSI had a significant positive effect on tree grade odds, but TSI did not prove beneficial within the selection cuts. Past potential tree grade measurements were also examined to determine their effectiveness in predicting future grade. It was found that between one-half to two-thirds of the trees in this study had the same potential tree grade in 2000 as they did in 1989.

In addition to quantitative tree measures such as DBH and height, managers of hardwood timber have the additional concern of determining the impact of silvicultural treatment upon tree grade. Much value can be lost if the system has detrimental effects on tree grade distribution, as lower quality trees will have lesser amounts of usable volume and/or poorer quality lumber extracted per unit tree. Understanding how specific treatments affect tree grade distribution can help to at least maintain or even improve quality in treated stands.

Erickson et al. (1990) found that a light improvement cut showed the greatest gains in grade 1 and 2 trees in a Michigan northern hardwood stand as compared to several diameter limit and selection system treatments. In a 40-year study of a Wisconsin northern hardwood stand, Strong et al. (1995) found that a medium intensity selection cut had the greatest improvements in tree grade versus light and heavy selection cuts, control, crop tree, and diameter limit treatments.

The first objective of this study is to examine any possible differences in potential tree grade (PTG) distribution between four silvicultural treatments: commercial clearcut, commercial clearcut with TSI, selection cut, and selection cut with TSI. Within this broad objective, there are several specific comparisons of interest. These PTG contrasts are: selection cut stands vs. clearcut stands; TSI stands vs. non-TSI stands; within commercial clearcuts, TSI vs. non-TSI; and within selection cuts, TSI vs. non-TSI. A second objective of this study is to examine how well past PTG measurements predicted current PTG measurements.

Study Site

The four plots included in this study are located on the Vinton Furnace Experimental Forest, near McArthur, Ohio with approximate coordinates of 39° 11.4' N. lat., 82° 24' W. long. Plot elevations range between 800 and 900 ft. Soils are Germano-Gilpin complex on southerly aspects and Steinsburg-Gilpin complex on northerly aspects. The pre-treatment stand was established in 1864 after clearcutting of timber for use as fuel for iron smelting. From this stand, six adjacent rectangular five-acre plots were assigned

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treatments as part of a study of management intensity on upland forests, with each plot having a 66-ft buffer zone surrounding it on all sides. Two plots were later removed from that study. The four remaining plots consisted of the following treatments:

- Commercial Clearcut – All trees containing at least one eight-ft log with a 10-in diameter inside bark (dib) top were harvested.
- Commercial Clearcut with TSI – Same treatment as Commercial Clearcut with all remaining trees of undesirable form or species being girdled, poisoned, or cut for pulpwood.
- Selection Cut – Mature, over-mature, diseased or damaged trees with at least one eight foot log (10 inches dib top) were harvested such that basal area was not reduced below 50 square feet per acre.
- Selection Cut with TSI – Same treatment as Selection Cut with all remaining trees that were diseased, non-commercial species, attacked by insects, or considered cull were girdled, poisoned, or cut for pulpwood. Additionally, crop trees were released from grapevines.

Treatments were installed between 1955 and 1957. Both Selection Cuts were treated in 1961, 1971-72, and 1985.

Material and Methods

Field Measurements

In 2000, four half-acre subplots were selected from each plot for ease of measurement. For each tree the following measurements were taken: dbh, actual tree grade (Hanks 1976), potential tree grade (Hanks 1976 without diameter restrictions), height to 4 in. and 8 in. top diameters, total height, sweep, and crook. Previously in 1989, these same measures were taken on the same trees. A total of 401 trees were included in the analysis.

Statistical Methods

A cumulative logistic regression model (CLR) was utilized to test for differences in treatment effects, due to the ordinal structure of PTG. The CLR model is a subset of the generalized logistic regression (GLR) model. Several forestry studies have utilized GLR models. For example, they have been used to estimate tree grade distribution for Northeast forests (Yaussy 1993), to predict veneer bolt grade for loblolly pine (Lynch and Clutter 1999), and to examine the relationship of dbh and bark nitrogen to beech bark disease severity (Latty et al. 2003).

For ordinal response variables with more than 2 levels, a dichotomy is introduced such that probabilities are modeled for only two classes: a same or increasing value class and a decreasing value class. A cumulative logistic regression model is of the form:

$$g(p_j) = f_j$$

where:

$$J = \text{number of classes (ordinal)} \\ i = 1 \dots J = 4 \quad (1)$$

$$j = 1 \dots J-1 \quad (2)$$

$$p_j = \Pr(Y \leq i | x) \text{ alternatively } \sum_{i=1}^j p_i^* \quad (3)$$

p_i^* = probability of being classified into the i^{th} category

β_k = regression coefficients ($k=0$ to n , with n =number of independent variables)

f_j = an additive function of the independent variables that is linear in its parameters,

$$\text{i.e. } f_1 = \beta_{10} + \beta_1 X_1 + \dots + \beta_n X_n \quad (4)$$

$f_2 = \beta_{20} + \beta_1 X_1 + \dots + \beta_n X_n$ etc. (functions differ only by intercept)

$$g(\bullet)=\text{logit function}=\log\left(\frac{p_j}{1-p_j}\right) \quad (5)$$

$$\left(\frac{p_j}{1-p_j}\right)=\text{odds}, \quad (6)$$

$$\frac{\text{odds}_j}{\text{odds}_{j'}} = \text{odds ratio (j}\neq\text{j')} \quad (7)$$

Note:

$$p_1=p_1^* \quad (8)$$

$$\sum_{i=1}^J p_i^* = 1 \quad (9)$$

$$p_4^* = 1 - \sum_{j=1}^{J-1} p_j \quad (10)$$

so

$$\Pr(Y=1|x)=p_1^* \quad (11)$$

$$\Pr(Y=2|x)=p_2^* = \Pr(Y\leq 2|x) - \Pr(Y\leq 1|x) \quad (12)$$

$$\Pr(Y=3|x)=p_3^* = \Pr(Y\leq 3|x) - \Pr(Y\leq 2|x) \quad (13)$$

$$\Pr(Y=4|x)=p_4^* = 1 - \Pr(Y\leq 3|x) \quad (14)$$

In a CLR model, the odds ratio (OR) incorporates the ordinal nature of the response variable PTG, and is used to examine differences between categorical independent variables. The OR represents the ratio of the odds for one treatment vs. the odds for another treatment that a tree will be in a selected grade *or better*, e.g., the odds that a tree from the selection cut treatment will be categorized as grade 2 or 1 vs. the odds that a tree from the commercial clearcut treatment will be categorized as grade 2 or 1. The odds in this example are the dichotomy of the probability of being in *grades 2 and 1* vs. the probability of being in *grades 3 and 4*. Note that since the model is a CLR model, these odds are uniform regardless of which grade is selected for a reference; in the example the selected grade is 2. Thus, the grades can simply be dichotomized as same or increasing (SI) and decreasing (D). Reference to these odds will be noted as SI/D.

When the confidence limit (CL) for the odds ratio estimate includes the value one, there is essentially no statistical difference in the probabilities between two treatments. For this study, interest centered on specific contrasts, as noted in the objectives. Tests of contrasts were Bonferroni adjusted in order to maintain the family-wide experimental error rate.

Cohen's Kappa statistic is utilized to determine the level of agreement between past PTG and current PTG. This statistic, which generally ranges from 0 to 1 but may result in a slightly negative value in rare instances, describes the strength of agreement between two raters of categorically defined cases. The statistic accounts for those cases of agreement that are due to random chance. The formula for Cohen's Kappa statistic is:

$$\kappa = \frac{P_A - P_C}{1 - P_C}$$

where:

P_A =proportion of cases for which raters agree

P_C =proportion of cases for which agreement is expected by chance, computed from marginal proportions.

All statistical analyses were performed using SAS®(SAS, Cary, NC).

Results and Discussion

Establishment Conditions

Data in Table 1 indicate the pre- and post-stand conditions at establishment. Greater numbers of stems larger than 12 inches and a concomitant amount of basal area were left in the two selection cuts than in the clearcuts, as would be expected. Compared to the other treatments, the commercial clearcut with TSI had the greatest reductions, with just a few stems and a very small amount of basal area left.

Previous Stand Removals

Beyond the establishment report, detailed stand records prior to 1976 were lost; only general summaries remain. Thinnings did occur in 1961 and the winter of 1971 and 1972. A third thinning occurred in 1985 (table 2). Thinnings in the commercial clearcut removed somewhat lower amounts of basal area, 17.2 ft²/ac, as compared to the two selection cuts: 27.6 ft²/ac for non-TSI and 23.2 ft²/ac for TSI. Stems removed from the commercial clearcut stand had a mean dbh of 13.6 in, which is larger than those from either of the two selections cuts, which had dbh means of 8.9 in (non-TSI) and 9.1 in (TSI).

Table 1.—Original stand data (1953)¹.

Treatment	No. Trees ≥4" dbh (per acre)	No. Trees ≥ 12" dbh (per acre)	Basal Area Trees ≥4" (ft ² /ac)
Pre-treatment 1953			
Commercial Clearcut	151	55	93.4
Commercial Clearcut with TSI	123	60	95.8
Selection Cut	116	46	77.8
Selection Cut with TSI	101	52	87.5
Post-treatment 1956			
Commercial Clearcut	99	11	37.9
Commercial Clearcut w/TSI	7	1	3.7
Selection Cut	90	31	53.4
Selection Cut w/TSI	49	32	44.0

¹Russell Walters, unpublished establishment report, 1958

Table 2.—Harvest removals and mortality (1985).

Treatment	Status	Trees per acre	Basal Area (ft ² /acre)	Mean DBH (in)
Commercial Clearcut	Harvest	16.8	17.2	13.6 (0.13)
	Logging Damage	7.2	1.7	6.4 (0.30)
	<i>Total</i>	<i>24.0</i>	<i>18.9</i>	
Selection Cut	Harvest	53.2	27.6	8.9 (0.25)
	Logging Damage	3.2	0.8	6.5 (0.49)
	<i>Total</i>	<i>56.4</i>	<i>28.4</i>	
Selection Cut with TSI	Harvest	34.8	23.2	9.1 (0.47)
	Logging Damage	5.6	1.9	7.1 (0.64)
	TSI	0.2	0.0	4.9
	<i>Total</i>	<i>40.6</i>	<i>25.1</i>	<i>21.1</i>
Commercial Clearcut with TSI	N/A	N/A	N/A	N/A

Current Stand Conditions

Major species in each of the stands consist of oaks (northern red, white, and chestnut), red maple, yellow-poplar, and some hickories. There is no dominant species on any stand. The greatest frequency of any species is red maple, which represents 41.3 percent of the commercial clearcut stand (fig. 1). Percentage representation by basal area (fig. 2) is generally similar in value to tree count percentage. The most notable exception is for white oak, which had an approximately 10 percent greater representation by basal area than by frequency count.

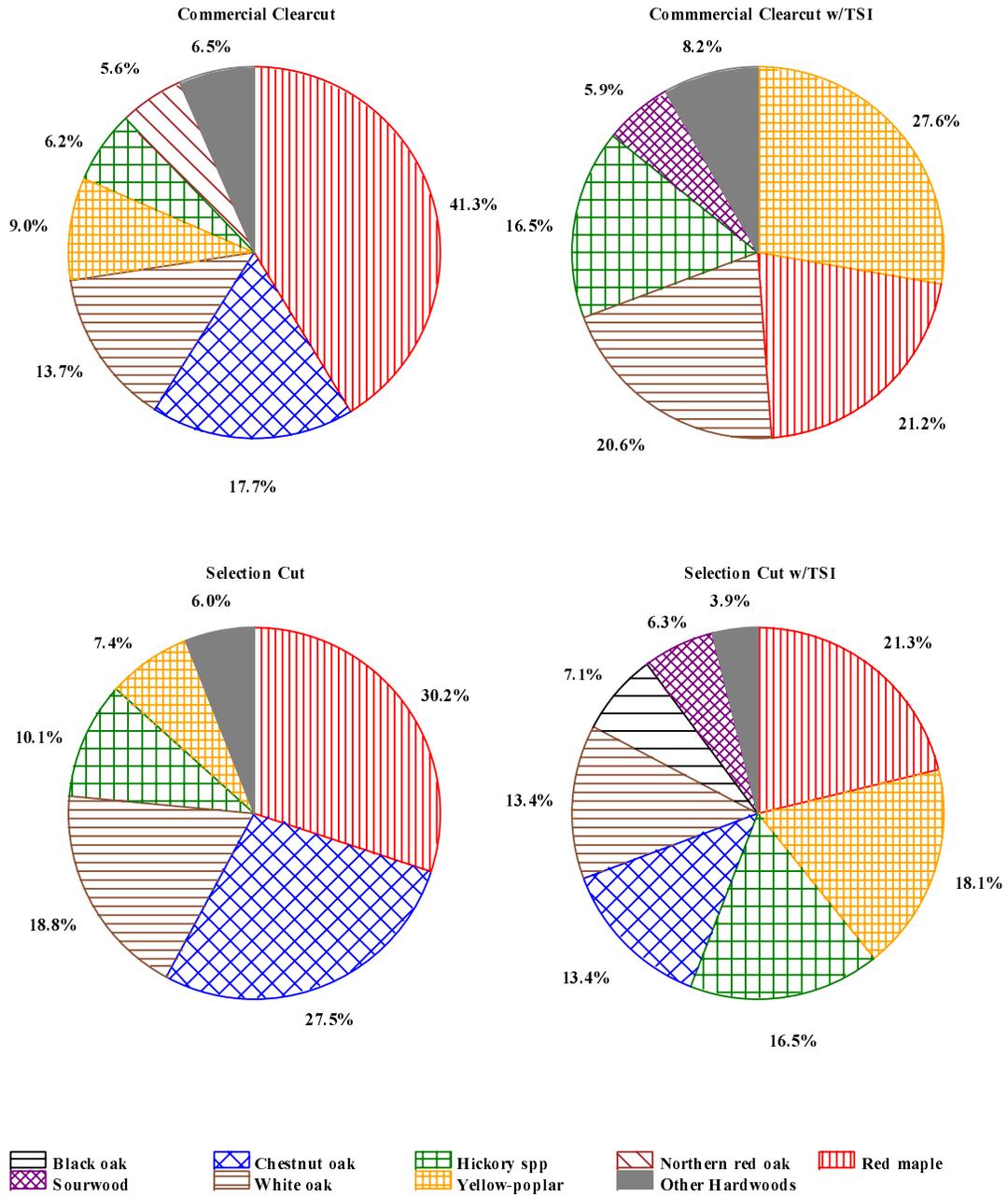


Figure 1.—Year 2000 percentages of trees (≥ 4.5 in. dbh) for each treatment summarized by counts.

Mean dbh across all species is lowest in the commercial clearcut stand, 9.6 in, and is highest in the selection cut stand, 12.9 in (table 3). The greatest density of stems (≥ 4 in dbh) is found in the commercial clearcut treatment, with 161 trees per acre (tpa). The selection cut with TSI had the least number of stems, having 64 tpa. Basal area ranged from 64.2 ft²/ac to 88.5 ft²/ac, for the selection cut with TSI and commercial clearcut treatments respectively. However, the number of stems (≥ 12 in dbh) was similar across treatments, ranging from 14-18 tpa.

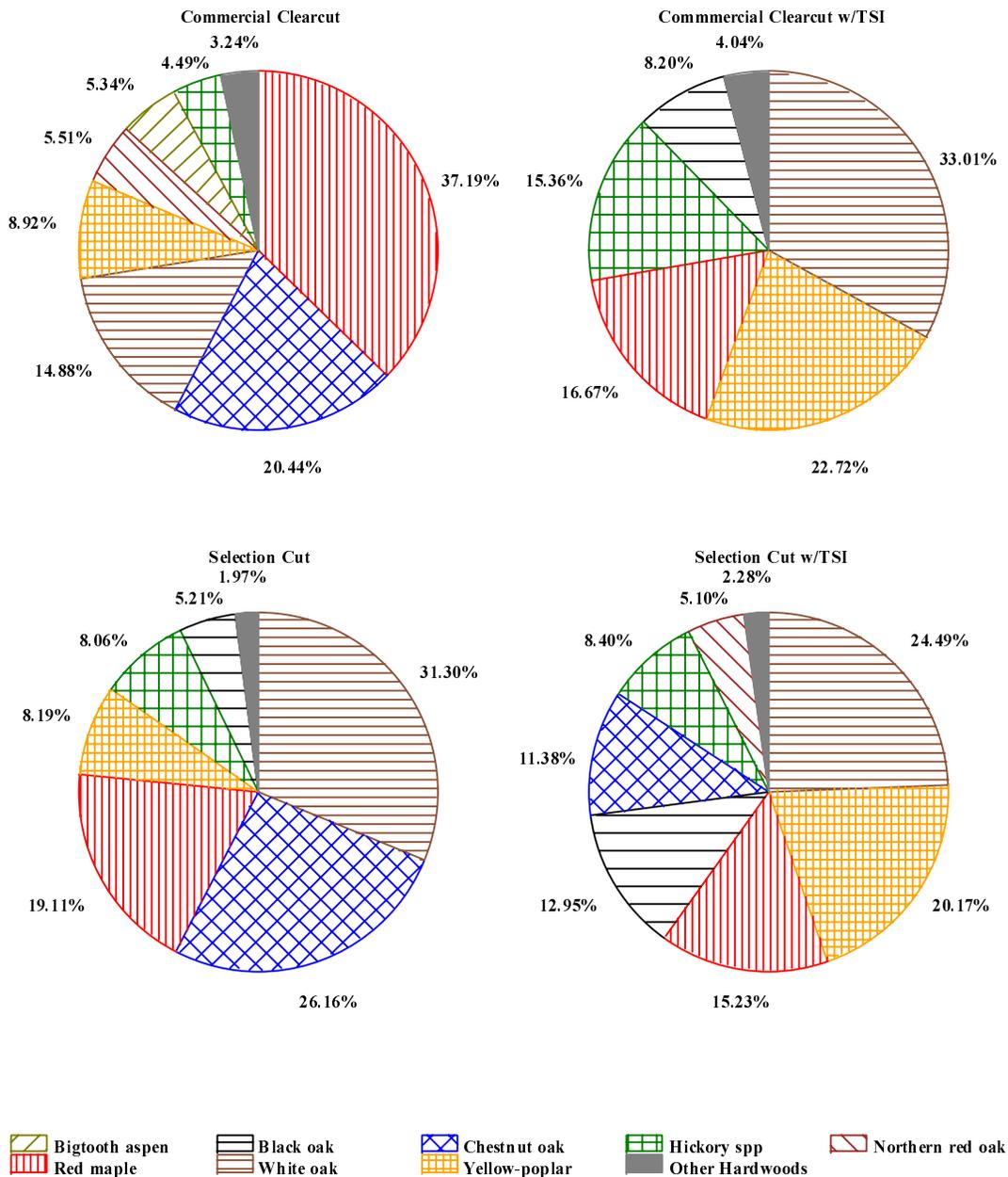


Figure 2.—Year 2000 percentages of trees (≥ 4.5 in. dbh) for each treatment summarized by basal area.

Table 3.—Current stand conditions (2000).

Treatment	No. Trees ≥4” dbh (per acre)			No. Trees ≥12” dbh (per acre)		
	n	Mean	se	n	Mean	se
Commercial Clearcut	4	161	20	4	18	1
Commercial Clearcut w/TSI	4	85	14	4	17	3
Selection Cut	4	75	14	4	18	4
Selection Cut w/TSI	4	64	11	4	14	1

Treatment	Basal Area (Trees ≥4”)			DBH (Trees ≥4”)		
	n	Mean (ft ² /ac)	se	n	Mean (in.)	se
Commercial Clearcut	4	88.1	6.3	322	9.6	0.17
Commercial Clearcut w/TSI	4	76.8	11.4	170	11.7	0.42
Selection Cut	4	78.8	11.9	149	12.9	0.43
Selection Cut w/TSI	4	63.7	7.3	127	12.3	0.50

Statistical Results

The score test for the proportional odds assumption was non-significant, p=0.775, indicating that proportional odds were satisfied. There was a significant effect of treatment on PTG, p<0.001.

In the tests of the four contrasts, only one contrast was non-significant, the comparison of TSI vs. non-TSI within selection cuts, p=0.601. The contrast of selection cut stands vs. commercial clearcut stands, of TSI stands vs. non-TSI stands, and of TSI vs. non-TSI within commercial clearcuts were all significant, p<0.001.

The first contrast to be considered is that of the selection treatments vs. the commercial clearcuts (significant at p<0.001). The odds ratio (table 4) for this contrast indicates that the SI/D odds of a tree in either of the selection cuts are 4.3 times the SI/D odds for a tree in either of the commercial clearcuts. Significantly greater SI/D odds for the selection cuts indicate the potential for better quality trees when utilizing selection cuts vs. commercial clearcuts. The PTG distribution for the commercial clearcuts contains twice as many grade four trees as do the selection cuts, while the selection cuts have twice as many grade 1 trees as do the commercial clearcuts (fig. 3).

The contrast between TSI and non-TSI stands is significant, p<0.001. The SI/D odds of the TSI stands are 5.9 times the odds of the non-TSI stands (table 4). There are 17 percent fewer grade 4 trees in the TSI

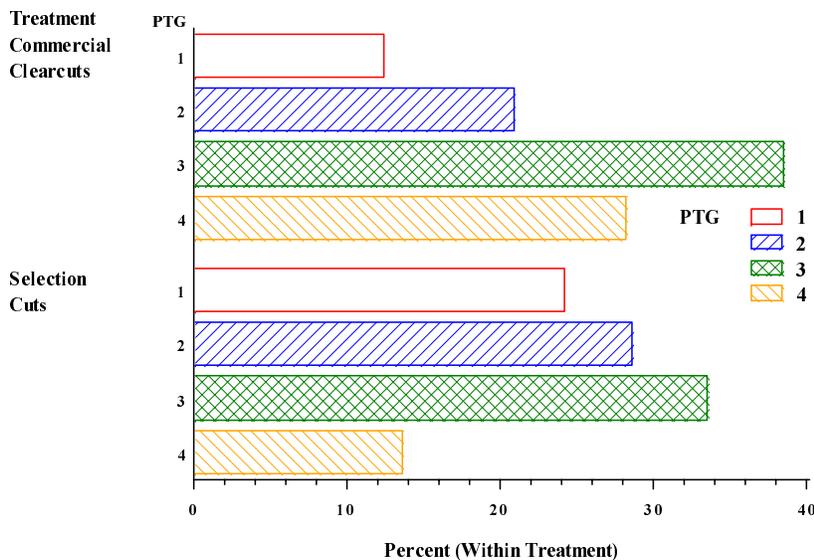


Figure 3.—Potential grade distribution of commercial clearcut stands vs. selection cut stands.

stands than in the non-TSI stands and nearly twice as many grade 1 trees in the TSI stands as than in the non-TSI stands (fig. 4).

Within commercial clearcuts, the contrast of TSI and non-TSI was also significant, $p < 0.001$.

The SI/D odds of a tree in the commercial clearcut with TSI were 5.1 times the SI/D odds of a tree in the non-TSI commercial clearcut (table 4). There are approximately 25 percent fewer grade 4 trees and nearly 20 percent more grade 1 trees in the TSI stand than in the non-TSI stand (fig. 5).

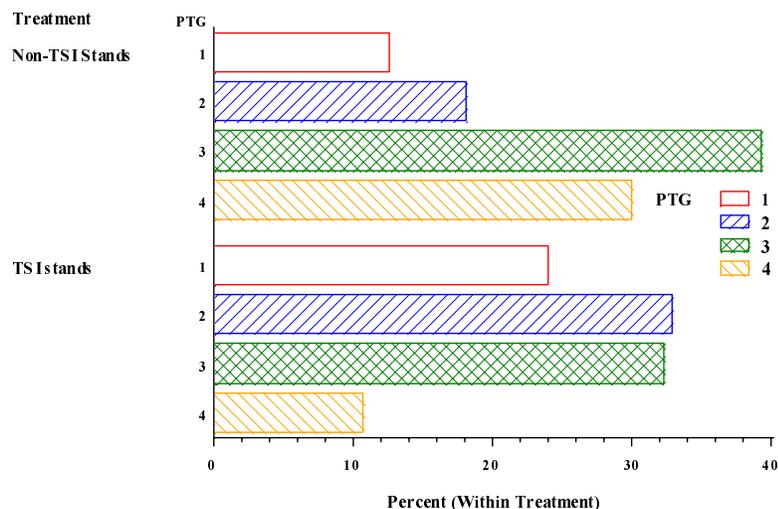


Figure 4.—Potential grade distribution of TSI stands vs. non-TSI stands.

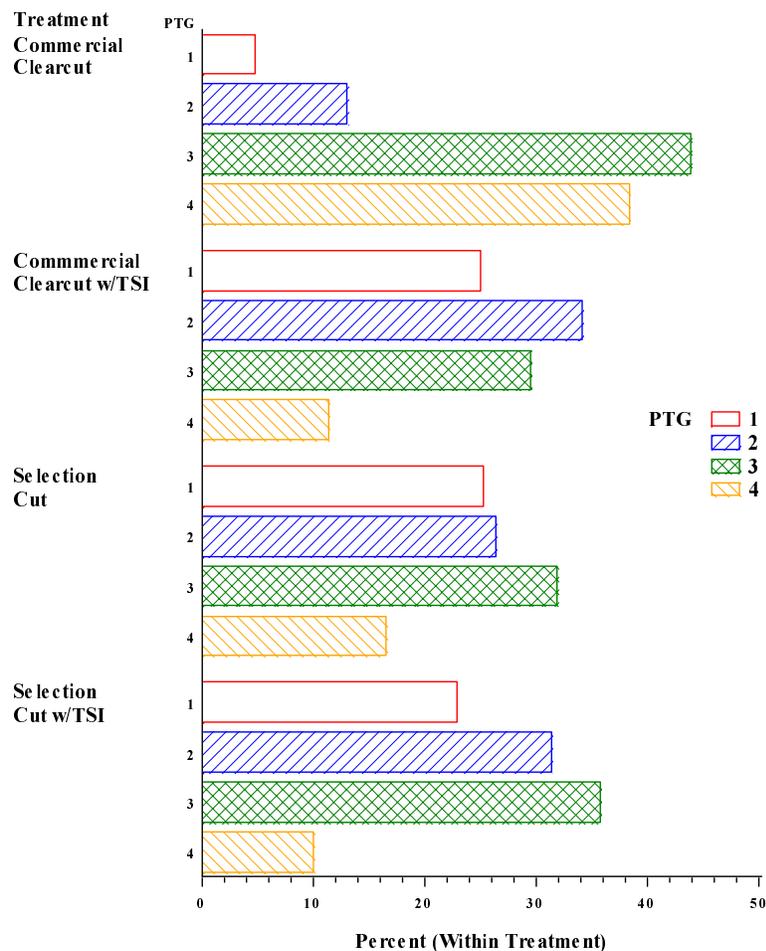


Figure 5.—Potential grade distribution of individual stands.

The lack of significance for the comparison of TSI vs. non-TSI within selection cuts is readily understandable. Selection cutting by definition attempts to remove trees of all sizes to achieve a uniform distribution of trees across age classes. As there was a harvest code to indicate TSI removed trees in 1985, scant trees were removed in order to apply TSI (table 2). The majority of removals were done for harvesting purposes, so a lack of difference in PTG within selection treatments is not surprising.

In order to assess the strength of agreement between 1989 PTG and 2000 PTG, Cohen's Kappa statistic was used. For this data, Kappa was equal to 0.42 (table 5), which would be rated moderate using Landis and Koch's (1977, p165) strength of agreement scale (see Appendix). As an alternative measure, a 95-percent CL on the proportion of matching PTG records is 0.48, 0.66, indicating that from one-half to two-thirds of the cases measured in 1990 were identical to those in 2000. Therefore, while PTG is a rough guide to future tree quality, additional measures could be useful to properly estimate future tree grade.

Table 4.—Odds Ratio estimates for selected contrasts.

Contrast	Odds Ratio*	Lower Limit	Upper Limit
Within Commercial Clearcuts, TSI vs. Non-TSI	5.1	2.5	10.2
Within Selection Cuts, TSI vs. Non-TSI	1.2	0.5	2.5
Selection Cuts vs. Commercial Clearcuts	4.3	1.5	12.3
TSI vs. Non-TSI	5.9	2.1	16.8

* Boldface odds ratios indicate significant differences at $\alpha=0.05$

Table 5.—Potential tree grade frequency counts.

PTG 1989	2000			
	1	2	3	4
1	27	4	2	0
2	13	14	8	2
3	1	12	19	5
4	0	1	4	10

$\kappa = 0.42$

Conclusions

Results from this study indicate two successful strategies for improving grade in stands of similar composition. First, TSI improved grade distribution when comparing across the two silvicultural treatments and also within the commercial clearcut options. Second, selection cutting demonstrated marked increases in grade over commercial clearcutting, with SI/D odds between 1.5 to 12.3 times those for the commercial clearcuts. Future studies including other silvicultural treatments would be of great utility in determining additional comparisons of treatments.

In addition, judging from data spanning eleven years, estimating future tree grade using PTG appears to be only moderately successful. While slightly over half of the trees maintained their original grade, the remainder demonstrated quality changes over this time period. Adding PTG measurements to ongoing inventories would provide further opportunities to determine the scope of these changes.

Acknowledgments

The authors would like to thank the staff at the USDA Forest Service RWU-4153 for long term monitoring of these stands and also two former Purdue University students, Rhett Steele and Jeff Page for taking field measurements.

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APPENDIX

Landis & Koch (1977, p.165) have suggested the following benchmarks for interpreting Kappa:

Kappa Statistic	Strength of Agreement
<0.00	Poor
0.00-0.20	Slight
0.21-0.40	Fair
0.41-0.60	Moderate
0.61-0.80	Substantial
0.81-1.00	Almost Perfect

COMPARISON OF GROWTH MODELS FOR WHITE AND RED OAK 10 YEARS AFTER RELEASE AND FERTILIZATION

Steven A. Knowe, Allan Houston and Scott Twillman[†]

Crop tree enhancement (CTE) treatments promote stand growth and quality while also providing an opportunity to maintain or improve wildlife habitat and aesthetics in hardwood forests. However, the response to crop tree release and fertilization treatments needs to be quantified for specific sites in order to assess the economic consequences of such treatments. Several methods of modeling forest growth have been proposed and utilized, but direct comparisons of these methods and models are not available for upland hardwoods. Including live crown ratio improves the performance in individual-tree growth models for western conifers compared to other modeling approaches. In addition, crown variables other than live crown ratio have not been considered for inclusion in forest growth models. Data from a CTE study in upland hardwoods at Ames Plantation, Grand Junction, Tennessee, are used to compare a relative size (stand table) projection system and an individual-tree diameter growth method. The predominant species in this study are white oak, red oak, and black cherry, with white oak being designated as the preferred crop tree species. The goal of CTE at this site was to promote volume growth and to improve the value of lumber produced. Treatments included an untreated control, fertilization only, release only, and release with fertilization, and were replicated 5 times. Measurements of dbh, total height, and crown dimensions were obtained for up to 36 crop trees in each treatment plot in 1992-93 and 2002-03. All trees included in a 10-factor prism sweep around each crop tree were designated as competitors. Total height, dbh, crown dimensions, azimuth, and distance from the crop tree were recorded for each competitor tree. Distance-dependent expressions of competition for consideration are the Hegyi index and a modified area potentially available based on basal area and crown dimensions. Distance-independent expressions of competition include: basal area of larger trees; crown diameter of larger trees; crown area of larger trees; live crown ratio of larger trees; crown volume of larger trees; and relative crown diameter, crown area, live crown ratio, and crown volume. The importance of crown variables in both modeling approaches will be evaluated.

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PRESCRIBED SURFACE-FIRE TREE MORTALITY IN SOUTHERN OHIO: EQUATIONS BASED ON THERMOCOUPLE PROBE TEMPERATURES

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ABSTRACT.—In the spring of 2001, experimental prescribed burns were conducted on three 20-ha treatment areas in the Ohio Hills of southern Ohio. Each treatment area contained ten 20- 50-m plots on which every tree >10 cm in diameter at breast height was inventoried before the burns. The plots also were instrumented with thermocouple probes which recorded probe temperature every 2 seconds during the burn. Tree mortality on the plots is described and compared with predictions from a series of physically based equations describing the flames, stem heating, tissue necrosis, and stem death. Equations for bark thickness and crown volume were developed for 13 species to help predict the ability of a tree to survive the heat from a fire. Fire effects are inferred from thermocouple probe temperature regimes and a calibrated probe heat budget. The equations provide a means of linking an important fire effect (stem death) with fire behavior and thermocouple data.

Prescribed fire has been suggested as a method to ensure the sustainability of the upland oak ecosystem of the Central Hardwoods region by controlling thin-barked, competing species such as red maple (Sutherland 1997). Bark serves as insulation, but thinner bark, present on smaller trees or thin-barked species makes a stem more susceptible to mortality from the heat applied to the lower bole by surface fires. Heat from intense flames penetrates the bark and kills the cambium, often resulting in fatal injuries. Research in western conifer forests has shown that cambium damage and crown scorch due to surface fires are closely associated with tree mortality (Reinhardt and Ryan 1989). It is assumed that bark thickness is the mechanism by which species differ in resistance to heat from surface fires.

Smoldering duff can heat root systems to the point of necrosis and subsequent mortality (Sackett and Haase 1992). Few models relate fire behavior to tree mortality in eastern hardwood forests. Absent a severe spring drought in this region, the duff does not dry sufficiently to allow burning or smoldering. After placing probes 1 cm below the soil surface, Iverson and Hutchinson (2001) recorded a maximum temperature of 27.6°C. In this paper we investigated only mortality due to aboveground heat from surface fires.

Bark thickness equations exist for tree species in the Central Hardwoods region, but most predict diameter inside bark given diameter outside bark at heights at or above 1.37 m (Hengst and Dawson 1994; Hilt and others 1983). We developed models for bark thickness below 1.37 m and crown diameter for 13 hardwood species. Stepwise logistic regression was used to estimate tree mortality based on measured or calculated tree variables, fire variables (derived from thermocouple readings), and combinations of the two.

Methods

The data used here were collected for two separate studies: the Ohio Hills site of the Fire and Fire Surrogate (FFS) study and the Ohio Hills National Fire Plan project (OHNFP). As part of the OHNFP, we used a digital micrometer to measure the thickness of three bark wedges chiseled from trees at 0, 15, 50, and 137 cm. Minimum and maximum crown width were measured by laser range finder for each tree to develop a linear equation for estimating average crown diameter based on DBH for each of the 13 species (Table 1).

A two-stage estimation process was used to develop an empirical model of lower bole bark thickness based on DBH since the measurements for an individual tree are not independent. In the first stage a nonlinear model of bark thickness by height for each tree was developed. In the second stage, DBH was used to predict the coefficients of the stage-one model for each species. Combining these models results in an equation for predicting species-specific bark thickness based on DBH for any height near the base of a tree.

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The FFS study encompasses 13 sites throughout the United States. At each site the same core variables are collected to evaluate the mechanical removal of biomass to simulate the effects of fire in ecosystems that developed with low-intensity, high-frequency fire regimes (Weatherspoon 1999). At each site, four treatments (control, mechanical removal, prescribed fire, and a combination of prescribed fire and mechanical removal) were replicated 3 times. The replications at the Ohio Hills site are in southern Ohio on the Raccoon Ecological Management Area (REMA), and the Tar Hollow (TAR) and Zaleski (ZAL) State Forests. Each 20-ha treatment area within each replication contains ten 20- by 50-m plots on which vegetation is sampled. Pretreatment measurements were taken in the summer of 2000 and prescribed burning was conducted in late March and early April of 2001. Post-treatment data were collected in 2001 and 2002. Because we are concerned with the effect of heat from prescribed fires on the mortality of overstory trees, we have included only the data from the areas that were burned but not thinned to reduce the confounding of mortality associated with factors such as logging damage and compaction. Only plots and trees with the full range of variables were used.

On the vegetation plots, we recorded the species of each tree more than 10 cm DBH as well as DBH, mortal status, height to the base of the live crown (hbc), total height, and three crown variables before treatment (Table 2). Two years after treatment, we recorded mortal status, and height of bark char. Crown assessment measurements were adapted from the North American Maple Project and consisted of:

Fine twig dieback, i.e., branch mortality that begins at the terminal portion of a limb and progresses inward:

- 0 = no or trace dieback present
- 1 = less than 10 percent
- 2 = 10 to 25 percent
- 3 = 26 to 50 percent
- 4 = 51 to 75 percent
- 5 = 76 to 100 percent

Defoliation — an estimate of the amount of foliage removed by chewing insects or foliar pathogens. The same scale for crown dieback was used.

Vigor — an impression of overall crown health. Vigor differs from dieback and defoliation in that it estimates what is not present. For example, a recent windstorm may have caused crown breakage and removed a portion of a crown:

- 1 = 10 percent or less branch or twig mortality, foliage discoloration, crown area missing, or abnormality present
- 2 = 11 to 25 percent of the crown missing/injured
- 3 = 26 to 50 percent of the crown missing/injured
- 4 = 51 to 75 percent of the crown missing/injured
- 5 = 76 to 100 percent of the crown missing/injured

During the prescribed fires a data logger located at the center of each plot recorded the temperature (every 2 seconds) of a rigid stainless steel rod with a thermocouple encased at the tip (Fig. 1). From these temperature readings we constructed the maximum temperature attained by the probe during the fire as well as the length of time the temperature remained above 30°C, a level somewhat above ambient air temperature. Few temperature profiles were recorded during the fire at TAR due to user error in deploying the data recorders. Almost all data recorders worked at the REMA and ZAL replications. Only data from plots which recorded temperatures were used in this analysis (Table 3).

With the equations for bark thickness and crown diameter, we were able to assign a bark thickness at 15 cm (representative of the area affected by surface fires) and crown dimensions to each tree from the FFS study based on pretreatment species and DBH measurements. The crowns of the trees were represented by ellipsoids based on crown length and width.

Table 1.—Mean and standard deviation (in parentheses) for datasets used to develop models for bark thickness and crown diameter (OHNFPP)

Variable	Species												
	Red maple <i>Acer rubrum</i>	Sugar maple <i>Acer saccharum</i>	Hickories <i>Carya</i> spp.	Dogwood <i>Cornus florida</i>	Beech <i>Fagus grandifolia</i>	Yellow-poplar <i>Liriodendron tulipifera</i>	Blackgum <i>Nyssa sylvatica</i>	Sourwood <i>Oxydendrum arboreum</i>	White oak <i>Quercus alba</i>	Scarlet oak <i>Quercus coccinea</i>	Chestnut oak <i>Quercus prinus</i>	Red oak <i>Quercus rubra</i>	Sassafras <i>Sassafras albidum</i>
Trees (no.)	33	16	22	15	32	33	28	35	27	11	36	11	13
DBH (cm)	14.703 (9.155)	9.975 (5.626)	10.391 (5.307)	6.980 (2.761)	10.316 (6.616)	11.745 (7.939)	8.054 (4.629)	12.400 (6.243)	13.137 (4.146)	15.064 (5.929)	13.247 (9.250)	10.491 (5.708)	10.708 (3.259)
Crown diameter (m)	5.996 (2.283)	4.425 (2.154)	3.202 (1.213)	3.591 (0.967)	5.714 (1.632)	3.638 (1.043)	3.731 (1.286)	4.033 (1.345)	3.763 (1.429)	3.626 (1.419)	4.059 (1.969)	3.380 (1.519)	3.163 (0.888)
Bark thickness (cm)	4.794 (2.597)	4.476 (2.339)	9.987 (4.621)	4.126 (1.528)	2.219 (1.488)	8.761 (4.957)	6.250 (2.907)	9.148 (4.491)	7.691 (2.653)	10.820 (3.914)	13.079 (6.283)	9.274 (4.598)	8.650 (3.388)

Table 2.—Mean and standard deviation (in parentheses) for datasets used to develop models for logistic model (FFS)

Variable	Species												
	Red maple	Sugar maple	Hickories	Dogwood	Beech	Yellow-poplar	Blackgum	Sourwood	White oak	Scarlet oak	Chestnut oak	Red oak	Sassafras
Trees (no.)	230	47	32	5	5	35	31	46	98	28	146	26	4
Dead (no.)	21	1	3	2	0	1	0	7	7	2	9	3	0
Dead (%)	9.130	2.128	9.375	40.000	0.000	2.857	0.000	15.217	7.143	7.143	6.164	11.538	0.000
DBH (cm)	17.111 (7.166)	15.945 (8.401)	27.053 (16.161)	10.700 (0.469)	12.300 (2.826)	25.643 (14.804)	15.152 (7.278)	14.435 (3.961)	37.357 (15.484)	40.564 (11.336)	37.233 (12.953)	43.123 (20.085)	30.650 (1.448)
Vigor (index)	1.235 (0.542)	1.170 (0.380)	1.125 (0.336)	2.400 (1.673)	1.000 (0.000)	1.086 (0.373)	1.097 (0.301)	1.543 (0.836)	1.459 (0.887)	1.357 (0.678)	1.411 (0.802)	1.115 (0.326)	1.250 (0.500)
Dieback (index)	1.174 (0.463)	1.085 (0.282)	1.094 (0.390)	2.600 (1.517)	0.800 (0.447)	1.057 (0.236)	1.065 (0.359)	1.804 (0.806)	1.429 (0.873)	1.643 (0.678)	1.281 (0.702)	1.231 (0.430)	1.250 (0.500)
Defoliation (index)	1.030 (0.315)	1.064 (0.247)	1.125 (0.421)	1.800 (1.789)	1.200 (0.447)	0.971 (0.169)	0.935 (0.250)	1.065 (0.533)	1.010 (0.550)	1.036 (0.508)	1.144 (0.599)	1.038 (0.196)	1.000 (0.000)
Char height (m)	0.508 (0.610)	0.334 (0.366)	0.523 (0.852)	0.480 (0.602)	0.220 (0.192)	0.900 (0.992)	0.629 (0.931)	1.296 (1.148)	0.664 (1.011)	0.829 (0.692)	0.516 (0.710)	0.919 (0.701)	0.900 (1.003)
Height to base of live crown (m)	7.081 (2.428)	5.825 (2.236)	9.728 (3.908)	4.756 (1.518)	2.378 (0.818)	9.460 (3.637)	5.881 (3.113)	6.071 (2.037)	11.193 (2.529)	11.923 (1.426)	11.452 (2.828)	12.629 (2.329)	8.155 (4.078)
Crown length (m)	5.068 (2.316)	7.045 (2.485)	4.192 (1.917)	2.683 (0.950)	8.720 (4.014)	5.174 (2.657)	4.032 (1.945)	2.757 (1.236)	5.068 (1.990)	4.617 (1.456)	4.369 (1.822)	4.468 (1.374)	7.927 (2.240)
Bark thickness (cm) at 15 cm	0.587 (0.158)	0.793 (0.291)	2.353 (1.082)	0.466 (0.006)	0.292 (0.047)	1.626 (0.616)	0.989 (0.270)	1.047 (0.169)	1.530 (0.392)	2.020 (0.292)	2.474 (0.466)	2.358 (0.658)	3.053 (0.165)
Crown volume (m ³) ^a	128.772 (113.586)	209.632 (331.624)	89.833 (100.992)	30.351 (10.495)	190.280 (124.492)	86.758 (106.475)	72.692 (67.025)	29.491 (20.707)	352.604 (348.022)	187.294 (103.272)	186.588 (142.009)	348.623 (318.276)	94.044 (30.257)
Crown scorch (%) ^a	0.624 (5.266)	1.305 (4.599)	0.000 (0.000)	13.545 (30.287)	22.853 (29.066)	3.416 (17.001)	6.672 (22.378)	6.605 (23.175)	0.000 (0.000)	0.000 (0.000)	0.773 (8.305)	0.000 (0.000)	0.000 (0.000)
Depth-to-thickness ratio ^a	0.781 (0.232)	0.496 (0.164)	0.209 (0.078)	0.816 (0.363)	1.520 (0.232)	0.293 (0.134)	0.422 (0.089)	0.415 (0.107)	0.302 (0.096)	0.226 (0.048)	0.176 (0.072)	0.217 (0.079)	0.124 (0.035)
Depth minus thickness ^a	-0.156 (0.186)	-0.430 (0.308)	-1.934 (1.066)	-0.087 (0.169)	0.143 (0.054)	-1.215 (0.654)	-0.589 (0.256)	-0.622 (0.195)	-1.099 (0.391)	-1.574 (0.307)	-20.61 (0.495)	-1.890 (0.669)	-2.671 (0.115)

^a Values calculated from modeled data, all other variables were measured.

Table 3. —Mean and standard deviation (SD) of FFS plot-level data used in logistic regression (plots: n=22)

Maximum temperature		Duration		Byram's fire intensity ^a		Height of crown scorch ^a		Depth of necrosis ^a	
Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
°C		seconds		kw/m		m		cm	
162.445	(51.269)	729.818	(380.849)	140.547	(53.605)	3.940	(1.006)	0.431	(0.091)

^a Values calculated from modeled data; all other variables were measured.

While Byram (1959) demonstrated the relationship between a subjective estimate of flame length and fireline intensity, a method of Bova and Dickinson (2003), was used to estimate Byram's fireline intensity from the maximum temperatures recorded by the temperature probes located in the center of each vegetation plot. The interaction between flames, winds, and tree boles cause uneven heating on different sides of larger trees, leading to an underestimation of mortality for those trees. As such, this method may be expected to work better for small trees that do not interrupt the fluid flow sufficiently to cause significant uneven heating. Bova and Dickinson (2003) have developed an equation that relates the depth of cellular necrosis in living trees to Byram's intensity. If this estimated depth was greater than bark thickness, cambial tissue likely would be killed, and stem death was predicted.

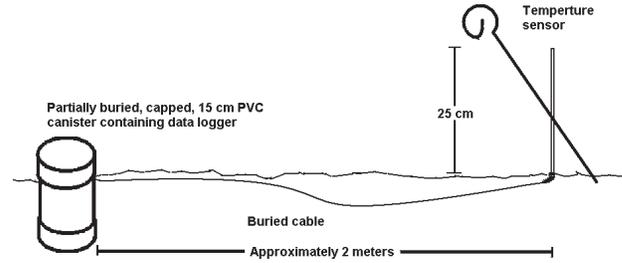


Figure 1.—Schematic of field installation of temperature probe.

Height of crown scorch was calculated with Bova and Dickinson's (2003) estimate of intensity and with the equations of Van Wagner (1973). An estimate of the percentage of the crown that was scorched by the fire was calculated for each tree in the plots based on scorch height, hbc, crown length, and crown volume.

A forward stepwise logistic regression was used to evaluate the ability of the variables, actual and constructed, to predict overstory tree mortality on the vegetation plots of the Ohio Hills site of the FFS study. Variables included measurements, calculations, and combinations of crown health, DBH, bark thickness, depth of necrosis, crown scorch, intensity, and char height (Tables 2-3).

Results

Statistics for the bark- and crown-dimension dataset and the FFS dataset are presented in Tables 1 and 2, respectively. Coefficients for the linear regression of crown diameter by DBH are listed in Table 4 for the 13 species in this study.

Graphs of several of the height-thickness relationships suggested fitting an exponential function for the first stage of the bark thickness equation:

$$BT = a + b * \exp(c * \text{height})$$

where:

BT = Bark thickness estimated by regression
 a, b, and c = Regression coefficients.

The intercept term was not significant and was dropped so that the final model fit to each tree was:

$$BT = a * \exp(b * \text{height}).$$

Table 4.—Regression coefficients for crown diameter and bark thickness models

Species	Crown diameter ^a		Bark thickness ^b		
	b ₀	b ₁	b	c	d
Red maple	2.885	0.212	-0.003	0.899	0.680
Sugar maple	1.662	0.308	-0.002	0.984	0.770
Hickories	1.753	0.141	0.002	1.810	0.778
Dogwood	1.594	0.286	0.000	2.447	0.271
Beech	3.632	0.213	-0.005	0.519	0.719
Yellow-poplar	2.440	0.102	-0.004	2.044	0.668
Blackgum	1.765	0.248	-0.001	1.851	0.632
Sourwood	1.979	0.166	-0.001	1.073	0.617
White oak	0.331	0.261	-0.002	1.843	0.600
Scarlet oak	0.700	0.193	-0.004	3.351	0.504
Chestnut oak	1.665	0.181	-0.004	4.391	0.498
Red oak	1.002	0.227	-0.007	3.291	0.557
Sassafras	2.336	0.079	-0.002	0.630	1.143

^aCrown diameter = b₀ + b₁ * DBH.

^bBark Thickness = c * DBH^d * exp(b * height).

The second stage used DBH to predict the coefficients of the stage one model for each species. Graphical observation suggested:

$$\hat{a} = c + d * DBH^e$$

where:

\hat{a} = First-stage coefficient to be estimated by second-stage regression

c, d, and e = Regression coefficients.

Again the intercept was not significant and the final model fit to each species was:

$$\hat{a} = c * DBH^d.$$

DBH was not correlated to the “b” parameter; therefore, the mean within each species was used.

Combining these models results in an equation to predict species-specific bark thickness based on DBH for any lower bole height:

$$BT = c * DBH^d * \exp(b * \text{height}).$$

Coefficients for these models also are listed in Table 4. The models differentiate bark thickness by species; beech (Table 1 includes common and scientific names) and red maple having the thinnest bark while hickory and chestnut oak have the thickest (Fig. 2). Sassafras appears to develop thick bark slowly but continually.

The final logistic regression model contains the variables for which the coefficients differed from zero at the 0.05 level. These include dieback, duration, depth of necrosis to bark thickness ratio (depth-to-thickness), and char height, none of which was correlated ($r > 0.500$). The model is of the form:

$$P_m = 1 - 1 / (1 + \exp(-8.6622 + 1.4091 * \text{dibak} + 0.00215 * \text{duration} + 3.1740 * \text{dtratio} + 0.4649 * \text{char}))$$

where:

P_m = Probability of mortality

dibak = Fine twig dieback

duration = Length of time the temperature of the probe remained above 30°C (seconds)

dtratio = Depth-to-thickness ratio

char = Char height.

These variables allow the model to attain a rank correlation of 0.730 (Somer’s D statistic).

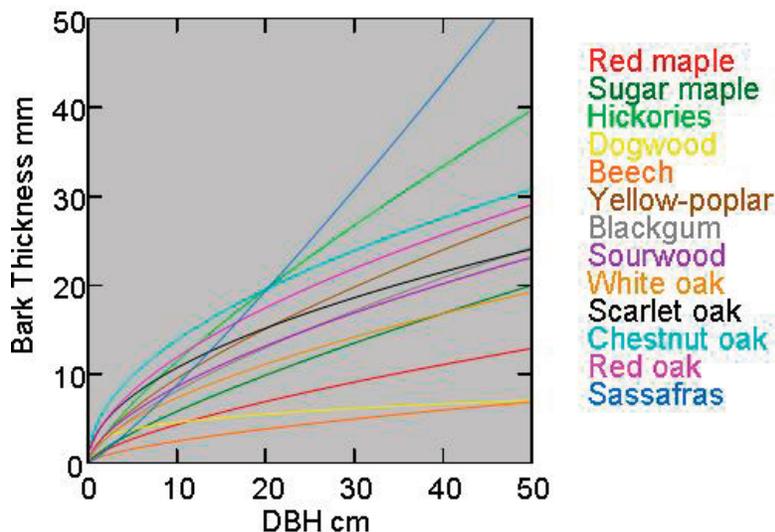


Figure 2.—Bark thickness at 15-cm height by DBH for 13 hardwood species.

Discussion

The DBH ranges of many of the species in the FFS dataset extend well beyond those of the dataset used to create the crown diameter and bark thickness equations (Tables 1-2). However, the models are well behaved within the range of tree sizes expected to be affected by fire, and are assumed to be applicable to the entire range of FFS data.

Forward stepwise logistic regression allows the user to see which variables or sets of variables add significantly to the prediction of mortality. All variables are assessed singly for their contribution to reducing the variability of the model. As new variables are entered, the remaining variables are reassessed based on their contribution to the new, larger, model. Dieback reduced the variation slightly more than vigor and was introduced to the model first. Once dieback was included, vigor no longer contributed much to the reduction of variance. Thus, although vigor was more highly correlated to mortality than all other variables except dieback, it was not included in the final model.

The five beech trees in the FFS dataset had the greatest predicted depth-to-thickness ratio and the largest predicted percent crown scorch (Table 2), yet none of these trees died. Bark char was lowest for beech, indicating that fire intensity at each bole may have been low compared to plot-level predictions. Beech had the shortest mean hbc. The relatively large crown with low limbs gives the trees high shade tolerance and may provide resistance to fire by protecting the boles from fuel build-up. If the lower limbs are killed in a fire, these trees still may have sufficient crown to maintain adequate photosynthesis—but also a higher probability of mortality in subsequent fires.

If these five beeches are removed from the dataset, percent crown scorch and depth of necrosis minus bark thickness (depth minus thickness) enter the model. Depth-to-thickness ratio and depth minus thickness are highly correlated and only depth-to-thickness ratio should be included in the model. This formulation of the data and model improved the predictive capability only slightly (Somer's D = 0.740).

Logistic regression typically is used to model tree mortality (Stringer and others 1989; Regelbrugge and Smith 1994; Reinhardt and Ryan 1989; Ryan and others 1988). Another method for modeling the effects of surface fire on tree mortality is stepwise discriminant analysis. This method results in identical independent variable selection and similar predictive capabilities.

Conclusion

Measured and estimated variables were used to construct a logistic model to predict mortality of trees two years after a prescribed fire in southeastern Ohio. The most significant indication of tree mortality following fire is the health of the tree prior to the fire. If the tree is under stress, as indicated by crown dieback or vigor assessments, it may be more susceptible to mortality with the added stress of fire effects. Equations were developed to estimate species and DBH specific bark thickness and crown structure. These estimates, along with estimates of heat transmission processes, added to the predictive ability of a logistic model. However, effective strategies may have evolved in tree species with thin bark, such as beech, to limit the build-up of fuels from around their boles enabling them to survive surface fires.

Estimates of fire intensity and duration from a single point are associated with the likelihood of mortality of trees up to 25 meters from the point, but can be refined, somewhat, with rough surrogates for intensity (char height) measured at each tree. More accurate models will be possible with microsite estimates of intensity possibly based on fuel loading and climatic conditions.

Acknowledgments

This is contribution number 41 of the National Fire and Fire Surrogate Research (FFS) Project. This research was funded by the USDA Forest Service through the National Fire Plan. Although the authors received no direct funding for this research from the U.S. Joint Fire Science Program (JFSP), it was greatly facilitated by the JFSP support of existing FFS project sites. We also thank Dr. Robert Long for the crown condition and char height measurements, James Stockwell for the bark thickness and crown dimension data, and David Hosack, Kristy Tucker, Brad Tucker among others for instrumenting the FFS plots and collecting the FFS data.

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LINKING HYPERSPECTRAL IMAGERY AND FOREST INVENTORIES FOR FOREST ASSESSMENT IN THE CENTRAL APPALACHIANS

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ABSTRACT.—Hyperspectral imagery from EO-1 Hyperion and AVIRIS were used in conjunction with continuous forest inventory (CFI) data to map detailed forest composition in the state forests of Western Maryland. We developed a hierarchical vegetation classification that conformed to the National Vegetation Classification Standard (NVCS) at the Alliance level and mapped these forest types as a function of hyperspectral reflectance using decision trees. Overall classification accuracy for vegetation at the Alliance level was very high (60-80 percent), with field validation indicating accuracies ranging from 65-70 percent. In an area dominated by oaks, the hyperspectral imagery was able to accurately distinguish plots dominated by individual red oaks with acceptable success (>60 percent). Hyperspectral imagery also differentiated between conifers more than 70 percent of the time. Overall, the accuracies were improved over similar analyses conducted using multi-date Landsat data. Our research demonstrates the capacity for hyperspectral imagery to remotely monitor, map and model forest systems in the Central Appalachians. The resulting forest composition maps can inform forest management decisions with a level of information content not previously available.

Mapping forest types is one of the primary applications of remote sensing data to forest management. In the past, standard satellite imagery available from Landsat era sensors has been limited in its ability to differentiate between specific hardwood forest types such as those described at the Alliance level of the National Vegetation Classification Standard (NVCS) (<http://biology.usgs.gov/npsveg/nvcs.html>). Forest maps with species level detail continue to be desired by forest managers and scientists alike, yet remain difficult to produce from satellite imagery without complex methods, extra data sources, and intense time and effort. Hyperspectral imagery improves upon the limits of Landsat data by measuring the reflectance of light in more than two hundred narrow bands, compared to Landsat TM's seven wide bands. Specific portions of the hyperspectral spectrum have been linked to forest indicators such as forest stress measured by canopy chlorophyll content (Sampson et al. 2003) and the biophysical content of leaves (Townsend et al. 2003, Smith et al. 2003). As such, the high data content found in hyperspectral data promises to greatly enhance forest mapping capabilities when used in conjunction with typical forest inventory data. We tested the ability of hyperspectral imagery from two sensors combined with Continuous Forest Inventory (CFI) data to map forest composition to the Alliance level and compared the results to a similar classification created from multi-season Landsat TM data.

Study Area

Green Ridge State Forest (GRSF) is a 16,000 ha working forest in Western Maryland (fig. 1). It is located in the Ridge and Valley physiographic province of the central Appalachian Mountains, and is characterized by steep northeast to southwest trending mountains, with many deep valleys. Elevation ranges from 200-700 m. Green Ridge lies in the rainshadow of the Allegheny front, receiving only 91 cm of rainfall annually, the lowest amount in Maryland. Most of the soils at GRSF are derived from either shale or sandstone with a relatively low water holding capacity (Mash 1996). The forests were largely cleared around the turn of the twentieth century, and are now mostly intact and mature. Scattered clearcuts and deferred harvest cuts at various stages of regrowth are located within the study area, though fewer than 100 ha are typically harvested annually. Large areas of uneven forest are located in places where gypsy moth defoliation over two decades has caused substantial tree mortality. Forests are comprised largely of deciduous oaks, with Virginia

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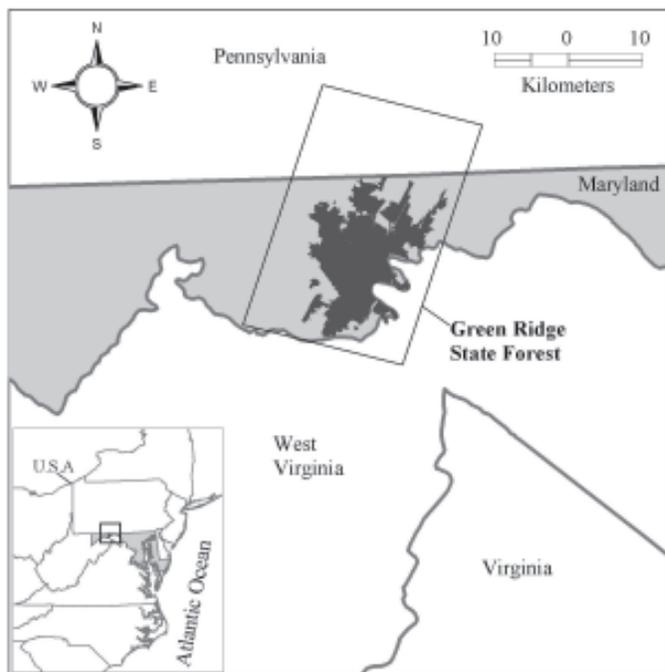


Figure 1.—Location of Green Ridge State Forest, Maryland.

pine (*Pinus virginiana*) and white pine (*Pinus strobus*) mixing in or forming pure stands and hemlock (*Tsuga canadensis*) growing on protected north-facing slopes. Of the key oak species, white oak (*Quercus alba*) and scarlet oak (*Quercus coccinea*) are both prominent on mesic slopes and at lower elevations. Red oak (*Quercus rubra*) and chestnut oak (*Quercus prinus*) are abundant on ridge tops and rocky slopes, and black oak (*Quercus velutina*) dominates mostly on mesic lower slopes. The understory is largely open, although blueberry (*Vaccinium spp.*) can be locally abundant and greenbriar (*Smilax spp.*) frequently expands in Gypsy moth defoliated areas.

Methods

Field Data

Continuous Forest Inventory (CFI) data from the Maryland Department of Natural Resources was used to sample the image data to classify forest types based on image spectra. The CFI database includes 432 plots in GRSE, all of which were sampled in 2000 or 2001. Each CFI plot is a 0.08 ha circular area on which all trees > 12 cm diameter are identified and measured. The CFI plots are arrayed on a regular grid at approximately 550 m intervals, yielding a statistical sample of the population of forest properties within the study area. All plots were geographically referenced using a Trimble Pro XR GPS.

Image Data

Hyperspectral images from sensors on two separate platforms were acquired for this research. A summer Hyperion image was acquired from the EO-1 satellite on 24 July 2001. An AVIRIS image was collected from an ER-2 aircraft on 14 May 2000. Hyperion has a spatial resolution of 30 m and covers a swath 7.68 km wide. It measures 210 bands at approximately 10 nm intervals from 400 nm to 2500 nm. High altitude AVIRIS pixels have approximately 17 m spatial resolution and 224 bands at 10 nm intervals between 400-2500 nm. Hyperion and AVIRIS images were converted to reflectance and referenced to UTM map coordinates as described in Townsend et al. (2003). The Hyperion image was also corrected for terrain illumination effects while the AVIRIS data used for this study was not (Townsend and Foster 2002). Four reflectance spectra were extracted from the hyperspectral images for each CFI plot. These spectral “bundles” were linked without averaging to their respective CFI plot data to create the hierarchical classification trees we used to map forest alliances. 423 plots overlapped with the AVIRIS image area and 283 plots overlapped with the narrower Hyperion image.

Table 1.—Forest associations found at GRSE. (Shaded classes were not validated in the field for this study and are not discussed in the results).

Class #	Class Name	Common Names	N (CFI Plots)
1	<i>Quercus rubra</i>	Red Oak	11
3	<i>Quercus rubra</i> - <i>Quercus</i> spp. - <i>Carya</i>	Red Oak - Oak species - Hickories	32
4	<i>Pinus virginiana</i>	Virginia Pine	36
5	<i>Pinus virginiana</i> / deciduous mix	Virginia Pine / Deciduous Mix	23
6	<i>Quercus prinus</i> - <i>Quercus coccinea</i>	Chestnut Oak - Scarlet Oak	7
7	<i>Quercus coccinea</i> / mix	Scarlet Oak / Mix	35
8	<i>Pinus rigida</i>	Pitch Pine	4
9	<i>Quercus velutina</i> / mix	Black Oak / Mix	52
10	<i>Quercus alba</i>	White Oak	75
11	<i>Quercus prinus</i> - <i>Quercus</i> spp. / mix	Chestnut Oak - Oak species / Mix	53
12	<i>Quercus prinus</i> - <i>Acer rubrum</i> / mix	Chestnut Oak - Red Maple / Mix	2
14	<i>Quercus prinus</i>	Chestnut Oak	21
15	<i>Prunus serotina</i>	Black Cherry	3
25	<i>Fraxinus</i> spp.	Ash species	1
33	<i>Sassafras albidum</i>	Sassafras	2
40	<i>Robinia pseudoacacia</i> / mix	Black Locust / Mix	3
42	<i>Fraxinus americana</i> / mix	White Ash / Mix	6
43	<i>Carya</i> sp.	Hickory species	14
44	<i>Acer saccharum</i> / <i>Quercus rubra</i>	Sugar Maple / Red Oak	6
45	<i>Tilia americana</i> / <i>Acer saccharum</i>	American Linden / Sugar Maple	1
49	<i>Pinus strobus</i>	White Pine	6
50	<i>Pinus strobus</i> / <i>Quercus</i> mix	White Pine / Oak Mix	24
52	<i>Tsuga canadensis</i>	Eastern Hemlock	5
53	<i>Liriodendron tulipifera</i>	Tulip Poplar	2
54	<i>Liriodendron tulipifera</i> / mix	Tulip Poplar / Mix	1
55	<i>Acer saccharum</i>	Sugar Maple	1
56	<i>Acer saccharum</i> / <i>Fagus</i> - <i>Fraxinus</i> mix	Sugar Maple / Beech - Ash Mix	5
59	<i>Acer rubrum</i> / <i>Fagus</i> - <i>Acer saccharum</i> mix	Red Maple / Beech - Sugar Maple Mix	1
TOTAL			432

A time-series of Landsat TM images from three dates was used to test the ability of multi-temporal Landsat data to map forest alliances. The time-series included an early spring image from 16 April 1997, an early summer image from 31 May 1996, and a mid-summer image from 24 July 2001. Each Landsat image was converted to planetary reflectance, orthorectified to UTM map coordinates, and corrected for terrain illumination effects using a Cosine correction (Meyer et al. 1993, Allen 2000). Both Landsat and Hyperion images were spatially accurate to within 30 m. Reflectance from TM bands 1-5 and 7 for each date for a total of 18 bands were extracted as described above and put into the CART analysis. 426 CFI plots overlapped with the TM image area.

Vegetation Classification

Vegetation types were assigned based upon the clustering of floristic data from 15,033 forest samples from the Central Appalachian region acquired from several sources, including data we have collected (3,813 plots), samples from the USFS Forest Inventory and Analysis (FIA) database (10,140 plots), and from the CFI database (1,080 plots). Sample areas and number of trees tallied were comparable among datasets, which permitted merging the data from multiple sources. Once the data sets were assembled, species names were standardized and relative basal area was computed by species by plot. The data were clustered using Ward's hierarchical agglomerative method (Lance and Williams 1967), with a small proportion of the plots eliminated as outliers. The classification yielded 41 forest alliances, of which 28 occurred in the Green Ridge study area (table 1). Subsequent to developing the vegetation classification, the image spectra

extracted for each of the CFI plots in GRSF were used to develop target hyperspectral signatures for the 28 associations. The spectral data were then used with decision trees to develop stand type maps.

Statistical Methods

We used classification and regression trees (CART) to map the Alliance level forest classes from spectral signature bundles. A variety of approaches have been used to map forest composition with hyperspectral data in mountainous landscapes (Martin et al. 1998) and with mixed forests (van Aardt and Wynne 2001). CART is being used increasingly for mapping from remotely sensed imagery (Friedl and Brodley 1997, Friedl et al. 1999, Hess et al. 1995, Cairns 2001) and is only generally described here. Classification trees – also known as decision trees – are fitted by binary recursive partitioning, in which data sets are consecutively divided into smaller subsets with increasing statistical homogeneity (Clark and Pregibon 1993). CART approaches are desirable because they are less sensitive to non-linearities in the input data than methods that require assumptions of Gaussian distributions (as do many image classification techniques) (Clark and Pregibon 1993, Venables and Ripley 1994). In addition, CART is an extremely valuable approach for data exploration when a very large set of independent predictor variables are available, e.g. with hyperspectral data sets. CART does not require data reduction, tests for normality or data transformations. One limitation to CART is that its performance is most robust and repeatable with large data sets, such as the CFI vegetation database that we used.

Field Validation

Forest maps created from the CART classification were validated using the “fuzzy sets” methodology originally proposed by Gopal and Woodcock (1994, also Townsend and Walsh 2001). Traditional measures of accuracy assessment that report simple “yes-no” accuracy are limited in applicability to complex forest landscapes because they assume all locations on a map can be unambiguously mapped to a single category. In addition, traditional methods do not convey the degree of error or mismatch between mapped categories and actual composition. Forest communities in the Appalachians are naturally heterogeneous and highly variable, grading across types in terms of both species dominance and presence. For this reason, forest composition can be described as “fuzzy,” and it can be acknowledged that for any one location it either may be difficult to identify any single best community type label or, alternatively, a location may be reasonably classified into two or more types. For example, in this work, we attempt to map forests dominated by black, red and scarlet oaks, whereas in many locations forests contain a mix of the three species. To accommodate this, in an accuracy assessment using fuzzy sets, all validation locations are assigned a fuzzy membership rating in all possible classes (from 1, completely wrong, to 5, absolutely right); fuzzy membership scores are then used to assess the magnitude and sources of confusion and ambiguity in the classification. We report 3 assessments of accuracy using the method of Gopal and Woodcock (1994): (1) percent accuracy for mapping to the “best” class vs. mapping to an “acceptable, but not necessarily the best” class), (2) the mean magnitude of errors by class, and (3) the source of errors (the average number of classes with acceptable membership for a given class).

NVC standards suggest visiting a minimum of 10 sites for each class when validating forest vegetation maps. Using an AVIRIS classification as a base map, we selected 10 validation sites for each class that was represented by more than 10 plots in the CFI plot data, for a total of 11 classes (see un-shaded classes in table 1). These sites were distributed widely and were generally accessible from forest roads. Incidental validation plots were added to replace inaccessible plots or when a unique forest type was observed. These plots increased the number of validated forest alliances to 15. Expert assessment was conducted independent of knowledge of the classification. We visited a total of 146 validation plots. Because of slight differences in the three maps from different sensors, the number of validation sites per forest type varies based on the map being analyzed.

Results

The results of the accuracy assessment of CART classification maps from each sensor are summarized in Tables 2 and 3, and require some explanation for those unfamiliar with fuzzy assessments. Table 2 shows different accuracy measures for all the field validated forest types. The learning accuracy for the CART classification trees is shown in the LEARN column and represents the percentage of the CFI forest plots

used to create the classification tree that were mapped to the correct class by that tree. According to learning accuracy, AVIRIS predicted CFI forest plot type with 80.9 percent accuracy overall, followed by Hyperion (64.4 percent) and Landsat (62 percent). AVIRIS mapped 10 of the 15 most abundant classes with a learning accuracy greater than 75 percent, of which 4 pine classes and the white oak class exceeded 90 percent accuracy. In comparison, Hyperion mapped only 5 classes with learning accuracy near or greater than 75 percent. These forest types included Virginia pine, white oak, black oak mix, white pine/deciduous mix and eastern hemlock. Landsat TM had the lowest overall learning accuracy, mapping only 4 forest types well (> 75 percent).

The remaining columns in Table 2 show the results of the fuzzy set analysis designed to measure the frequency of matches and mismatches between the mapped forest types and the independent validation data. The number of samples (N) corresponds to the number of validation sites visited for each class for a particular map. We include two accuracy metrics from the fuzzy set analysis in this table. The MAX metric represents the percentage of validation sites that were mapped to the absolute best class as seen in the field. The RIGHT metric represents the percentage of validation sites that were mapped to a good or acceptable forest type. This measure gives a better sense of map accuracy for the map user, because it illustrates the probability that a point on the map will be mapped to a class that agrees to a large extent with what is observed on the ground. The improvement in accuracy gained from the MAX to the RIGHT metric is shown in the IMP column. We can see that the RIGHT function improved the MAX accuracy by 20-25 percent on average.

According to MAX total, all three forest vegetation maps have a low overall validation accuracy ranging from 41.3 percent for Landsat to 46.5 percent for Hyperion. Fortunately, the RIGHT measure shows that the actual ability of the classifications to map good or acceptable classes ranges from 65.3 percent for Hyperion to 70.2 percent for AVIRIS. Hemlock stands, which are easy to distinguish both spectrally and in the field, were mapped to the best possible class with consistently high accuracy by each sensor. The forest type that was mapped with a relatively high RIGHT accuracy most consistently across platforms was the black oak mix class with 79.2 percent success for AVIRIS, 68 percent (Hyperion), and 82.1 percent (Landsat). AVIRIS mapped 9 out of 12 validated classes with 60 percent accuracy or greater. Hyperion mapped 7 out of 11 validated classes with 60 percent accuracy or greater and Landsat TM broke 60 percent accuracy with 7 out of 12 validated classes. Considering the degree of mixing among the forest vegetation classes being mapped, these accuracy levels are a promising accomplishment. No previous studies have attempted to map such specific classes.

Table 3 illustrates the average magnitude and direction of the map validation errors (MAG) and the average fuzzy membership statistic (MEMBER). Where a mismatch occurs between the best forest type in the field and the actual forest type found on the map, the MAG statistic shows the average difference between the maximum score given for a forest type at a given site and the next highest score given for other possible forest types. For example, a site mapped as chestnut oak – oak species / mix turned out to actually fit the exact description of the chestnut oak – scarlet oak class when visited in the field. In this case, the maximum score given was a 5 to the chestnut oak – scarlet oak class and the second highest score was a 4 for the chestnut oak – oak species / mix class. The difference value for this particular site would be the score for the mapped forest type (4) minus the maximum score (5) resulting in a value of -1. A case where a site is accurately mapped as the best possible class with no other reasonable choices would give a difference of 5-1, or +4. A MAG value of zero indicates that a site was accurately mapped to the best possible class. Thus MAG values close to zero are desirable. Slightly negative values show that although the best class was not correctly mapped for that site, it was actually mapped to the next best choice. Given the specificity and mixing characteristic of our forest alliances, we did not expect to find many cases where there were strongly positive differences except for very distinct classes such as the hemlock, which had slightly positive MAG values for AVIRIS and Hyperion.

The average difference values for each sensor hover around a value of -1 with AVIRIS having the least negative difference, indicating that when the best forest type was not matched by the map, the mapped class had a score only 1 unit below the best. This result confirms that our forest types are generally not distinct classes with discrete boundaries, but rather mixes dominated by sets of species that vary

Table 2.—Results of classification learning accuracy and fuzzy set field validation accuracy metrics for three sensors

Class Name	AVIRIS			EO-1 HYPERION			LANDSAT™		
	N	LEARN	IMP	N	LEARN	IMP	N	LEARN	IMP
<i>Quercus rubra</i>		43.0%		2	30.0%	0.0%	7	34%	0.0%
<i>Quercus rubra</i> - <i>Quercus</i> spp. - <i>Carya</i> mix	5	66.0%	40.0%	11	53.0%	45.5%	5	16%	40.0%
<i>Pinus virginiana</i>	13	95.0%	38.5%	18	95.0%	16.7%	18	99%	22.2%
<i>Pinus virginiana</i> / deciduous mix		93.0%			68.2%		3	78%	33.3%
<i>Quercus prinus</i> - <i>Quercus coccinea</i>	1	46.0%	0.0%		0.0%			0%	
<i>Quercus coccinea</i> / mix	13	77.0%	38.5%	6	51.0%	33.3%	9	32%	44.4%
<i>Pinus rigida</i>	3	94.0%	33.3%		0.0%			0%	
<i>Quercus velutina</i> / mix	24	85.0%	20.8%	25	74.0%	12.0%	28	69%	32.1%
<i>Quercus alba</i>	20	93.0%	15.0%	25	79.0%	16.0%	22	77%	13.6%
<i>Quercus prinus</i> - <i>Quercus</i> spp. / mix	12	83.0%	41.7%	1	49.0%	0.0%	13	71%	38.5%
<i>Quercus prinus</i>	7	81.0%	42.9%		22.0%		3	30%	0.0%
<i>Carya</i> sp.		64.0%		1	58.3%	0.0%		59%	
<i>Pinus strobus</i>	7	69.0%	14.3%	4	30.0%	50.0%	1	64%	0.0%
<i>Pinus strobus</i> / <i>Quercus</i> mix	5	91.0%	20.0%	1	88.8%	0.0%	5	82%	40.0%
<i>Tsuga canadensis</i>	11	75.0%	0.0%	7	77.0%	0.0%	7	36%	0.0%
TOTAL	121	80.9%	25.6%	101	64.4%	18.8%	121	62%	24.8%

Table 3.—Fuzzy set validation statistics showing the magnitude and direction of errors (MAG) and the average membership per class (MEMBER).

Class Name	AVIRIS			EO-1			Landsat TM		
	N	MAG	MEMBER	N	MAG	MEMBER	N	MAG	MEMBER
<i>Quercus rubra</i>				2	0.00	8.00	7	-0.71	5.00
<i>Quercus rubra</i> - <i>Quercus</i> spp. - <i>Carya</i> mix	5	-1.40	4.60	11	-1.09	5.36	5	-0.80	6.20
<i>Pinus virginiana</i>	13	-0.92	5.62	18	-1.56	5.00	18	-1.28	5.00
<i>Pinus virginiana</i> / deciduous mix							3	-3.00	2.00
<i>Quercus prinus</i> - <i>Quercus coccinea</i>	1	-4.00	6.00						
<i>Quercus coccinea</i> / mix	13	-1.38	5.62	6	-1.50	5.67	9	-2.22	5.56
<i>Pinus rigida</i>	3	-2.33	1.00						
<i>Quercus velutina</i> / mix	24	-0.75	6.34	25	-0.92	5.72	28	-0.93	5.89
<i>Quercus alba</i>	20	-1.00	4.20	25	-1.28	5.08	22	-1.05	4.91
<i>Quercus prinus</i> - <i>Quercus</i> spp. / mix	12	-1.83	5.67	1	-2.00	4.00	13	-2.62	5.85
<i>Quercus prinus</i>	7	-1.00	5.71				3	-2.67	5.33
<i>Carya</i> sp.				1	-4.00	3.00			
<i>Pinus strobus</i>	7	-0.57	4.29	4	-0.25	3.00	1	1.00	2.00
<i>Pinus strobus</i> / <i>Quercus</i> mix	5	-1.00	6.00	1	-4.00	6.00	5	-2.00	4.80
<i>Tsuga canadensis</i>	11	0.27	3.00	7	0.14	3.86	7	-0.71	3.86
TOTAL	121	-1.00	5.09	101	-1.13	5.16	121	-1.37	5.21

compositionally across the landscape. Most individual classes validated by more than 5 sites had difference magnitudes close to -1.

We also include an average membership statistic in Table 3 that indicates the degree of fuzziness for each forest type as mapped by the three different sensors. The MEMBER column shows the average number of classes – including the mapped class – that were potentially acceptable choices for a given site (for example, classes 4 and 5 in Table 1 are both dominated by Virginia pine and it is reasonable to assume that each could potentially be misclassified as the other). A class with high average membership values has a fuzzier distinction from other classes than a class with a relatively low membership value. Most of our forest alliances had an average membership ranging from 4-6 classes. The lowest membership was found in the eastern hemlock class. Of the oaks, the white oak alliance was the most distinct, having the lowest average membership.

Figure 2 shows the forest type map derived from the CART classification tree of AVIRIS data for Green Ridge State Forest. GRSF is bounded on the east by the meandering Potomac River and on the west by a smaller creek, and it is bisected in its northern half by the interstate. Non-forest areas have been masked out in black. Conifers and conifer/deciduous mixes are shown in shades of green, white oak is shown in light blue, and red oak and red oak mix are shown in shades of red. Black oak mix is shown in brown, scarlet oak mix in maroon, and chestnut oak and its mixes in shades of yellow and orange. Rare classes that have not been discussed here are shown in other colors.

The classification map itself can inform us about its ability to predict the distribution of forest associations at the Alliance level. A few characteristics of this particular map suggest that it is in fact mapping these forest classes with reasonable success before even considering the accuracy assessment. First, related mixed forest types tend to occur adjacent to each other or as mixes. This is especially evident with the red oak (Class 1) and red oak – oak species – hickory mix class (Class 3). Where relatively large contiguous patches of red oak dominated forests occur, they almost always occur next to or mixed with the red oak mix class. The same can be said of chestnut oak and its mixes, shown by shades of yellow and orange. In addition, the classes which are mixes of pines and deciduous species often occur as transition zones between the pine dominated forest types and the deciduous forest types. These patterns are expected based on the knowledge of how forest composition changes across species and environmental gradients. Finally, there is a distinct

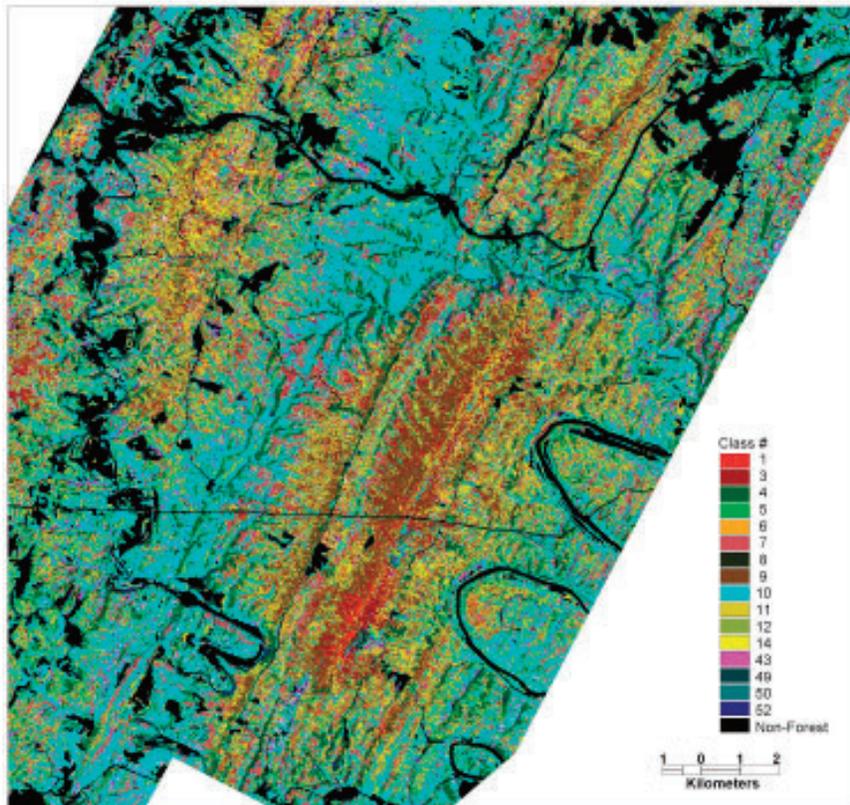


Figure 2.—Map of forest alliances derived from CART classification of AVIRIS imagery. Only the field validated alliances are referenced in the classification key.

dominance of white oak (light blue) in an oblong area running through the center of the state forest that is characterized by a shale substrate and more complex valley and drainage patterns than the rest of the area. In comparison, red oak classes in reds, browns, and yellows tend to dominate the long and linear sandstone mountains oriented northeast to southwest. These two distinct regions represent geological differences that seem to favor the dominance of either red oaks or white oaks.

Discussion

Using CART classification of hyperspectral and forest inventory data, we have produced Alliance level forest maps with accuracies unattainable from traditional methods. In the past, mapping of forest alliances from remotely sensed data typically resulted in accuracies of 40 percent or less (Czaplewski 2003). Our accuracies for maps from AVIRIS data averaged 80 percent for most alliances, which is a substantial increase in accuracy from past efforts. These successes are especially notable considering the various types of uncertainty confronted during this study.

One perplexing result was that white oak was not validated with greater accuracy. This is surprising because it is the most abundant species represented in the CFI data and generally easy to identify in the field. One possible reason for this discrepancy is that the ubiquity of the white oak class across parts of GRSF affects the expert evaluation during the field validation. More than any other class of hardwoods, the white oak class at Green Ridge is dominated by its namesake, with 26 of the 75 white oak CFI plots having a white oak relative basal area of 40 percent or greater. When a large percentage of validation sites sample these relatively pure white oak forests, oak mixes with less conspicuous dominance of white oak may suffer by comparison and subsequently be scored higher as some other type of oak mix. This tendency suggests that the most abundant classes are also characterized by an extreme range of dominance levels and may in fact be more vulnerable to incorrect scoring during fuzzy validation. If this is the case, then the high learning accuracies reported for the white oak forest alliance (93 percent AVIRIS, 79 percent Hyperion, and 77 percent Landsat) are probably closer to the true accuracy than the fuzzy validation values.

The somewhat low MAX accuracy values seen in Table 2 illustrate the difficulty of accurately mapping mixed deciduous forest alliances to the best possible class using hyperspectral data. Revisiting the descriptions of these forest classes in Table 1 underlines this. Several of the most abundant forest types are oak or pine mixes where the dominant species in the type frequently comprises from 30-40 percent of the basal area and occurs on almost all of the plots (Class 3, 5, 7, 9, 43). This level of dominance for a single species is common in Green Ridge's mixed forests, while smaller pockets of forests dominated by 40-50 percent or more relative basal area for a given species occur infrequently (except for white oak) (Class 1, 4, 14, 50). The level of specificity and mixing in our forest alliances probably pushes hyperspectral data to the limits of its current ability to separate spectrally mixed classes using the methods described here.

Nonetheless, it is important to remember that the field validation dataset consists of only one fourth the number of inventory plots used to create the classifications, and these were typically classified correctly at a much higher rate. The learning accuracy reported in Table 2 is more robust in this way because it is derived from a much larger sample size and from the quantitative forest inventory data. The field validation provides an independent assessment of the learning accuracy while also characterizing the fuzzy nature of our forest alliances.

Fuzzy set validation proved extremely useful for validating our satellite based forest maps in the field. It is particularly relevant when mapping a heterogeneous mixed forest with satellite imagery, where the minimum mapping unit is a 17 or 30m² pixel. An expert visiting a pixel-sized area on the ground will often find that multiple forest types are acceptable labels for that site. Although the map itself is limited to representing each pixel with only one discrete forest class, much of the forest area is covered by regularly varying mixes of the species that dominate the alliances in Table 1. The canopy reflectance measured for a forest pixel will be a combination of the reflectance from the crowns of all the trees in the pixel area. AVIRIS data samples a smaller area with fewer tree canopies, and thus can be expected to create the most accurate maps, since its pixels experience less spectral mixing than larger pixels.

Sources of Uncertainty

The greatest source of uncertainty for our classification and validation methods was geographic inaccuracy within our AVIRIS image dataset. The AVIRIS data caused the greatest concern because certain distortions in the data caused by the pitch and yaw of the aircraft could not be removed from the data completely. While using hundreds of ground control points created excellent spatial accuracy in most areas, some small regions in the AVIRIS image could be located as far as 80m away from their actual location. This uncertainty inevitably means that some of the spectra sampled from the AVIRIS image to represent certain CFI plots in the CART classification actually sampled the incorrect area. As a result, some proportion of the spectral signatures used to train the classification might have been incorrectly labeled. These problems associated with geographic map errors tend to dampen map learning and validation accuracy values.

We did not account for differences in basal area or forest age with our final classification maps except to mask out areas of recent forest harvests generally characterized by plot basal areas less than 10 m²ha⁻¹. We chose instead to use all the CFI plot data available to map as many classes as possible, though we didn't expect to be able to map classes represented by only one or two CFI plots with good accuracy. Other classification maps were created using only a subset of the data such as plots having a total basal area greater than or equal to 20 m²ha⁻¹. These maps had lower overall accuracy and tended to map forest types to large contiguous areas that probably represented an oversimplification of the mixed forests at Green Ridge. While the forests of Green Ridge are generally even aged, variations in basal area and maturity can introduce variance into the spectral signature bundles representing a given forest type, adding to the difficulty of distinguishing among mixed forest types.

Finally, sources of error are introduced into the dataset from the collection of the field data itself. It has been estimated that forest inventory data may only be about 80 percent accurate, with uncertainty generally coming in the form of misidentified or misrecorded tree species or incorrectly acquired or recorded geographic coordinates.

Conclusions

We were able to map Alliance level forest associations with accuracies generally ranging from 60-90 percent using Hyperspectral imagery from AVIRIS. Hyperion data were more limited in mapping ability, possibly because of its lower signal-to-noise ratio compared to the other data sets (Green et al. 2003) or alternatively because only half as many CFI inventory plots were available for training because of its narrow swath width. Multi-temporal Landsat TM achieved similar accuracy levels as Hyperion but were not as high as AVIRIS. Nevertheless, the ability to map Alliance level forest types with such a high level of accuracy opens up new opportunities to ask and answer questions about the spatial distribution of forest types and to use the information gained to improve forest management.

Acknowledgments

This research was supported by NASA EO-1 Science Validation Team grant NCC5-493. Thanks to Robert Chastain and Brian Sturtevant for their contributions to this work, and to Francis Zumbrun, Chris McCann and the Maryland DNR CFI program for contributing forest data. This paper is Scientific Series Contribution Number 3709, University of Maryland Center of Environmental Science.

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ECOLOGICALLY STRATIFIED HEIGHT-DIAMETER MODELS FOR HARDWOOD SPECIES IN NORTHWESTERN LOWER MICHIGAN

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ABSTRACT.—Ecological classification systems (ECS) have been developed for northwestern Lower Michigan to stratify the landscape into areas of similar ecological potential. ECS are useful only to the extent that they capture quantifiable differences in tree form and function between defined strata and provide valuable, ecologically-referenced information for forest managers. Ecologically referenced height-diameter functions were developed here to demonstrate the utility of an ECS as a stratification tool for modeling stem form for northern hardwoods.

Ecological classification systems (ECS) have been developed and mapped to different scales across the United States (McNab and Avers 1994). The usefulness of an ECS hinges upon observable differences in variability in ecosystem structure and process between ECS map units. In other words, ECS are basically spatial stratification systems that assume that the variability within ECS units is less than the variability between them. If forest managers are to use ECS they need to see how they can be applied in an operational context. Here, height-diameter (H-D) functions, which are useful in forest inventory, were fitted to individual tree measurement data to determine if variation in tree morphology was effectively captured by Ecological Land Types (ELT), a type of ECS, defined for the Huron-Manistee National Forest in Michigan. Huang et al. (2000) undertook a similar study for white spruce in boreal forests, but ecologically referenced H-D functions have not been developed yet for northern hardwoods.

Methods

The goals for this study were to utilize variability in individual tree measurements across the landscape to see if differences in height-diameter relationships for different species were substantially affected by a site potential defined by an ECS.

Study area and data

The area of study was the Manistee portion of the Huron-Manistee National Forest, in northern Lower Michigan. The study area was large (~290,000 ha) covering a diverse array of glacial landforms and forest ecosystems (Host et al. 1988). Ecological land types (ELT), a type of ECS, were developed and mapped for much of this area. ELTs were based on regional differences in glacial parent material for soil formation and field estimates of soil properties and ground flora composition (Host and Pregitzer 1991) and are described by Cleland et al. (1993).

ELT units included are: Outwash Plains (OWP), described as *red oak-white oak-beath* (*Vaccinum spp.*) communities occurring on excessively-drained sand plains; Dry Ice-contact and Sand Hills (DSH), described as *mixed oak-red maple-Trientalis* communities occurring on excessively-drained over-washed moraines, kame terraces and glacial spillways; Mesic Ice-contact Sand Hills (MSH), described as *northern red oak-red maple-Viburnum spp.* communities occurring on well-drained sand hills; Herb-Poor Moraines (HPM), described as *sugar maple-American beech-Maianthemum* communities occurring on well-drained morainal sands; and Herb-Rich Moraines (HRM) described as *sugar maple-white ash-Osmorhiza* communities occurring on nutrient-enriched morainal sands.

Individual tree measurements data were available from 74 experimental reference stands used for developing ELTs for the Huron-Manistee National Forest (detailed methods are presented in Host et al. 1988). Only well-stocked (Basal area > 7 m² ha⁻¹), even-aged stands of at least 1 ha. in area, with minimal evidence of

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Table 1.—Summary statistics for individual tree data for trees measured in 1983 and 2003, broken down by ecological land type (ELT). Mean values are followed by [std; min, max]

ELT	n	height (m)	dbh (cm)	age (years) [†]
OWP	795	17.9 [4.5; 2.7, 30.4]	27.0 [11.6; 6.3, 106.7]	76.7 [19.9; 32, 138]
DSH	797	20.5 [5.0; 3.5, 34.4]	28.2 [11.0; 3.8, 65.0]	82.0 [14.7; 37, 127]
HPM	923	24.7 [6.0; 4.6, 43.8]	30.4 [14.1; 4.8, 97.8]	66.0 [10.5; 42, 93]
HRM	531	25.4 [5.7; 3.0, 41.7]	32.1 [14.3; 6.0, 106.6]	66.7 [12.0; 21, 102]
MSH	1242	27.4 [5.5; 6.0, 42.9]	34.8 [12.6; 5.8, 100.6]	76.0 [12.7; 47, 116]

[†]age was measured from a smaller number of trees than n.

disturbance, were selected (these stands originated from heavy logging at the turn of the last century). Study plots were established in each stand, in 1983. Four permanent point plots were randomly located to assess variability within the stand. An iron rod or piece of PVC pipe was placed at the center of each point plot and a BAF 10 cruising prism was used to estimate the number of trees in the plot (i.e., basal area in 1983). A metal tag, embossed with a tree identification number, was affixed to the base of each tree with a nail. Stem diameter at breast height (D) and total tree height (H) were measured for each tree. Age (A) was estimated in 1983 from an extracted core, on a subset of dominant or co-dominant trees, designated to estimate site index for common trees. Point plots were relocated in 2003 and dbh and height were measured again. Age for trees measured in 2003 was computed by adding 20 to the age measured in 1983. Measurement data are summarized by ELT in Table 1.

Analysis

The exponential Michelis-Menten model was used to model H-D relationships (Pacala et al. 1994):

$$H = \mu \left(1 - e^{-(\beta/\mu)D} \right) \quad (1)$$

where H is the height (m) of a tree and D is its diameter (cm) at breast height (at 130 cm above ground). The parameter μ is asymptotic height and β is the initial slope of the H-D relationship.

Equation 1 was chosen because μ and β should have an interpretable ecological meaning and because μ , in particular, should relate to site quality. An estimate of the parameter β for a group of trees growing in the same ELT should roughly correspond to the effect of ecological conditions on the average taper of smaller (dbh), suppressed and intermediate trees. However, it should be difficult to separate an “ELT” effect on β from other factors that influence H-D ratios, namely differences in stand density (Wang 1998). The parameter μ , in the scenario described, corresponds to the average upper limits of height for the larger (dbh), dominant and co-dominant trees within a particular ecological land type (ELT). Given the well-established influence of water and nutrient availability on the upper limits of height growth (Ryan and Yoder 1997), μ should reflect major differences in site quality, for trees large or old enough that they are approaching their height limit. Differences in μ should also be much less sensitive to stand density because the largest trees were likely much less influenced by competitive effects on height and diameter growth. Since the ELTs for the Huron-Manistee N.F. were defined (through multivariate statistical methods) by differences in soil texture and herbaceous plant composition (which have been shown to be correlated with available soil water and nutrients, Host and Pregitzer 1991), significant differences the parameter μ , between ELTs, should reflect that meaningful differences in site quality on stem form.

Individual trees measured in both 1983 and 2003 were pooled into two groups for analysis: (1) all trees pooled by ELT, and (2) all trees pooled of a species by ELT, to assure that trees covering a wide range of sizes and ages (Table 1) from a range of stand conditions were represented within each ELT. Species examined were: northern red oak (RO, *Quercus rubra*), white oak (WO, *Q. alba*), black oak (BO, *Q. velutina*), northern pin oak (PO, *Q. ellipsoidalis*), red maple (RM, *Acer rubrum*), sugar maple (SM, *A. saccharum*), American beech (AB, *Fagus grandifolia*) and white ash (WA, *Fraxinus americana*). BO and PO were grouped together as in the OWP because field crews did not separate them there. Equation 1 was

fitted for ELT and species-ELT combinations and parameters μ and β were estimated using non-linear regression analysis. Some species were not present in sample plots within a particular ELT or were present in insufficient numbers to obtain a statistically valid estimate of parameters μ and β , however, so an exploration of all possible species-ELT combinations was not possible.

The precision of each model was estimated with the coefficient of determination (r^2). To determine how much additional variation in height was explained by adding species level information, r^2 estimated for each ELT model were compared to that estimated for species-ELT models. Average residual error of prediction (= observed-predicted height) was used to detect for bias in each model, i.e., systematic over- or under-prediction of height. A two-tailed t-test for differences in means was used to test for significant ($\alpha = 0.05$) differences in model parameters. Differences in model parameters between ELTs were assumed to represent the effects of an ELT on the H-D relationship of *any* tree in the total population of trees, regardless of its species. Differences in model parameters between species within an ELT were assumed to represent the combined effect of species and ELT on H-D relationships. Although other factors, including local differences in stand basal area, past disturbance and stand age should also influence H-D relationships, I assumed that pooling trees from a wide diversity of locations, stand types and ages would allow for site (ECS) and species effects on H-D relationships for individual trees to be assessed, while accounting for these other influential variables. To test for this, residuals were analyzed to determine if significant residual error could be attributed to differences in stand basal area or tree age.

Results

H-D functions for ELTs and species-ELT combinations are presented in Table 2. Values for μ and β result in essentially anamorphic H-D functions (fig. 1) with significant differences in H-D functions between 4 of the 5 ELT examined. There was no statistically significant difference between the two types of moraines (HPM and HRM). Thus, the original 5 ELTs could be effectively collapsed to represent 4 strata which meaningfully separate out differences in H-D relationships as a function of ELT.

Although tree stem diameter within ELTs was fairly well correlated (r^2 ranged from 0.63 to 0.68) with tree height, about one third of the variability in height remained unexplained by dbh. At least part of this unexplained error is likely due to the difficulty in accurately estimated the total height of hardwood species in mature, closed-canopy stands. Despite the unexplained error, average error of prediction for the ELT models was very small, ranging from 0.03 to 0.10 m (table 3). However, all models systematically under-predicted height for the tallest trees (fig. 2) due to the fact the μ represents the *average* asymptotic height (i.e., dominant canopy height) rather than a true maximum height. This is likely a problem when fitting any H-D functions with an asymptotic model, although authors more typically plot residual error as a function of predicted height (Huang et al. 2000) or dbh (Peng et al. 2001), which does not reveal this pattern. Analysis of residuals revealed no significant residual error explained by either tree age or stand basal area, thus, these factors were effectively scaled out of these H-D functions.

When species were used as an additional stratification tool along with ELT (table 2), there was very little differentiation between species on either DSHs or HRMs, except for AB, which differed substantially from other species on HRMs, due to the fact that smaller (dbh) AB were shorter than other species at the same dbh (β lower). On OWP, H-D functions were different for all three species (and groups, i.e., BOPO) with significant differences in μ for WO and in β for RO. H-D functions for RO and RM were essentially the same on MSHs, but asymptotic height for WO was about 5.5 m below them. H-D functions on the HPM were essentially the same for two groups of species, [RO and RM] and [SM and AB], with an estimated 10 m greater asymptotic height between them.

To determine if ELT models were enhanced with species information, ELT models (table 2) were used to predict height for trees in species-ELT combinations. When individual trees were grouped by species and ELT, the fit of the H-D functions improved dramatically for a few species (as indicated by differences in r^2 vs. r^2 (ELT) (table 2), but in many cases resulted in only a small improvement in fit. Average error of prediction, on the other hand, was significantly improved when species information was used (table 3). However, as this error was generally less than 1 m for the ELT-only models (table 3), the magnitude of

Table 2.—Height diameter functions for different species within ecological land types (ELT). μ and β are the asymptotic height (m) and the H-D ratio for very small trees, respectively, estimated by least squares non-linear regression from eq. 1. Model parameters which are not significantly different ($\alpha = 0.05$) between ELTs or species within an ELT are assigned the same lower case letter (w, x, y, z,) or (a, b, c), respectively. r^2 is the fit of the model and $r^2(\text{ELT})$ is the fit when the ELT model (Table 1) is used to predict height for species-ELT combinations.

	n	μ	95%CI for μ	β	95%CI for β	r^2	$r^2(\text{ELT})$
OWP	795	24.84 w	[24.05, 25.63]	1.31a	[1.25, 1.36]	0.65	-
WO	295	22.77 a	[21.33, 24.21]	1.29a	[1.19, 1.40]	0.60	0.56
RO	105	24.46 b	[22.88, 26.05]	1.68b	[1.51, 1.86]	0.67	0.40
BOPO	339	25.65 b	[24.52, 26.77]	1.27a	[1.18, 1.36]	0.59	0.58
DSH	797	27.57 x	[26.75, 28.40]	1.49b	[1.42, 1.56]	0.67	-
RO	297	26.97 a	[25.91, 28.03]	1.61a	[1.49, 1.73]	0.61	0.59
WO	225	28.69 ab	[26.60, 30.79]	1.45a	[1.32, 1.57]	0.60	0.60
RM	81	28.76 ab	[23.24, 34.29]	1.56a	[1.37, 1.75]	0.72	0.70
BO	177	31.25 b	[27.68, 34.82]	1.18b	[1.03, 1.32]	0.49	0.44
HRM	531	33.12 y	[32.12, 34.13]	1.73c	[1.64, 1.81]	0.63	-
SM	239	32.53 a	[30.99, 34.07]	1.84a	[1.72, 1.96]	0.70	0.70
WA	62	33.18 a	[30.64, 39.03]	1.77a	[1.38, 2.08]	0.49	0.49
AB	15	34.20 a	[26.23, 42.16]	1.18a	[0.83, 1.53]	0.81	0.61
RO	65	35.08 a	[33.05, 37.11]	1.58a	[1.34, 1.82]	0.53	0.49
HPM	923	34.08 y	[33.25, 34.92]	1.66c	[1.60, 1.72]	0.67	-
AB	111	28.10 a	[26.29, 29.90]	2.04a	[1.77, 2.31]	0.66	0.58
SM	344	28.38 a	[27.16, 29.61]	2.06a	[1.91, 2.20]	0.61	0.55
RO	290	37.14 b	[35.66, 38.61]	1.47a	[1.37, 1.57]	0.61	0.58
RM	61	37.19 b	[27.42, 46.96]	1.51a	[1.20, 1.81]	0.50	0.50
MSH	1242	35.20 z	[34.51, 35.88]	1.69c	[1.64, 1.75]	0.68	-
WO	114	30.05 a	[28.12, 31.98]	1.86a	[1.58, 2.13]	0.59	0.14
RO	846	35.30 b	[34.47, 36.12]	1.73a	[1.65, 1.80]	0.56	0.56
RM	199	36.56 b	[34.56, 38.56]	1.64a	[1.54, 1.74]	0.83	0.82

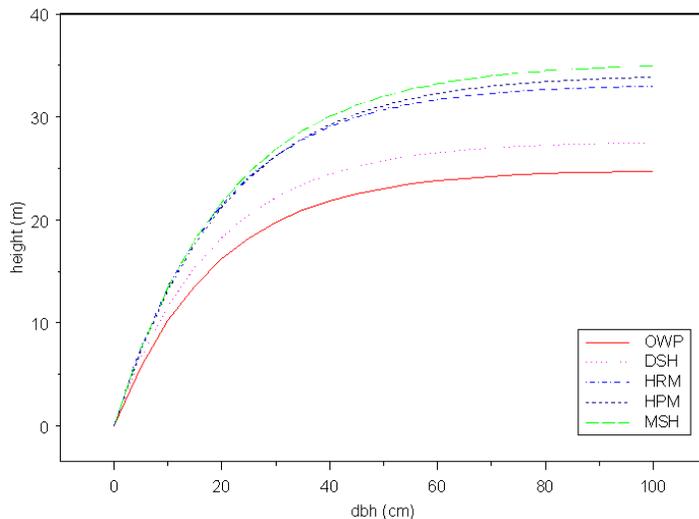


Figure 1.—Height-diameter functions for any tree species growing within an ecological land type (ELT). Curves for HRM and HPM are not statistically different.

Table 3. —Average error in height prediction (= observed – predicted height) for height-diameter functions (table 2). error(SPELT) refers to species-ELT models and error(ELT) refers to the error in the ELT model, or the error when the ELT model is used to predict height for a species within that ELT.

ELT	SPP	n	error(SPELT)	error(ELT)
OWP		795	-	0.03
	WO	295	0.01	-0.84
	RO	105	0.01	1.84
	BOPO	339	0.05	0.20
DSH		797	-	0.10
	RO	297	-0.05	0.25
	WO	225	-0.02	0.15
	RM	81	-0.07	0.56
	BO	177	0.00	-0.62
HRM		531	-	0.04
	SM	239	0.04	0.44
	WA	60	0.01	0.23
	AB	15	0.11	-3.12
	RO	65	0.01	0.64
HPM		923	-	0.10
	AB	111	0.00	-0.68
	SM	344	0.01	0.06
	RO	290	0.05	0.44
	RM	61	0.06	-0.23
MSH		1242	-	0.05
	WO	114	0.00	-1.82
	RO	846	0.01	0.30
	RM	199	0.02	0.10

improvement was small relative to the probable accuracy of the height measurements. Thus, while knowledge of species improved the power of H-D functions in most cases, these results suggest that the effect of ecological conditions (as indicated by soil-site conditions) was far more influential than the effect of species (i.e., genetics) on stem form. Hence, in this case, H-D functions developed for ELTs alone would be more robust than those developed for individual species.

Conclusion

Ecological Land Types (ELT) developed for the Huron-Manistee National Forest appear to be useful stratification tools for partitioning variability in H-D functions (i.e., stem form) across the varied glacial landscape of northwestern lower Michigan. Consistent with other studies (e.g., Host and Pregitzer 1991, Wang et al. 2000), these results suggest that effectively stratifying sites through the use of Ecological Classification Systems (ECS) can provide useful, ecologically-referenced information for managing forested ecosystems.

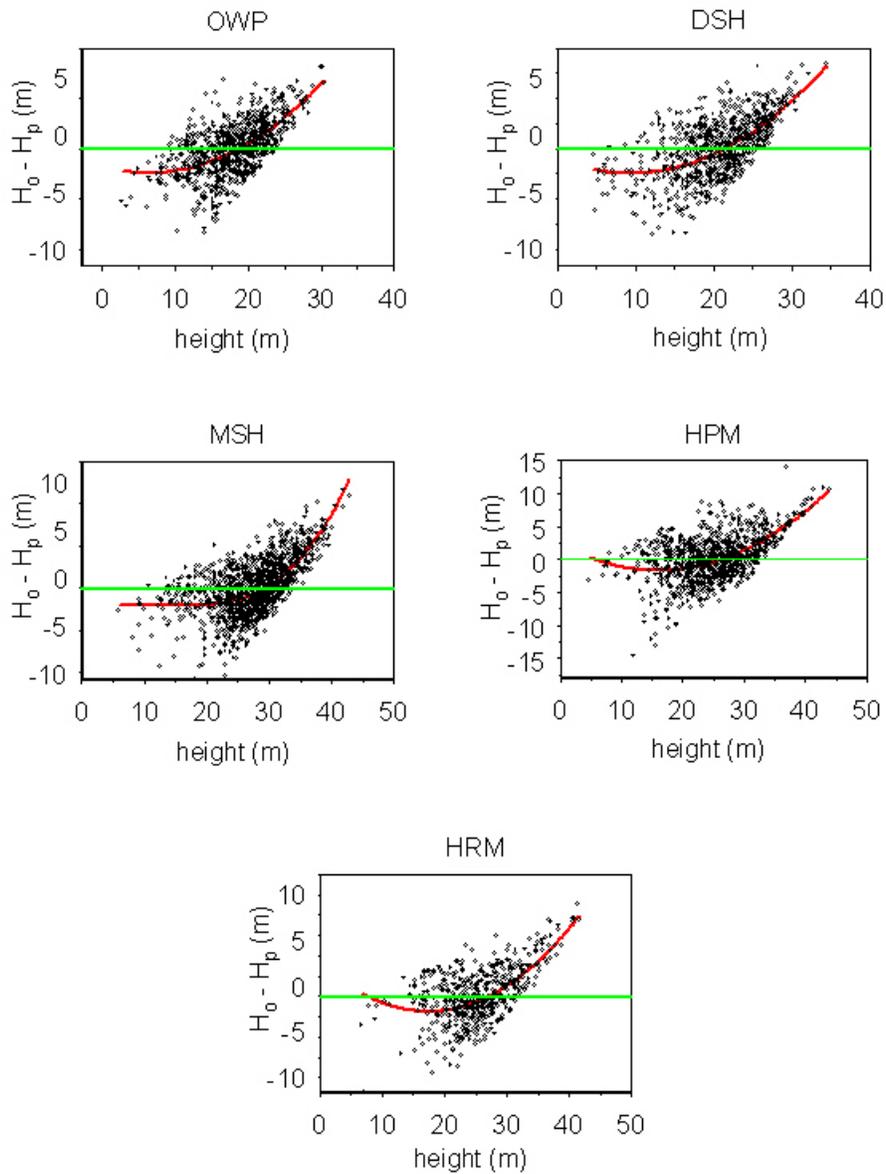


Figure 2.—Residual error of prediction (= observed height – predicted height = $H_o - H_p$) from height-diameter functions for ELTs (Table 2) plotted against observed tree height. The curved line through each scatter plot represents the best fit line for the data, using a second order polynomial model, and the horizontal line represents the zero error line.

Acknowledgment

The author would like to thank Bud Hart, Michigan State University, and Joe Gates, H.M.N.F. for help assembling and understanding the full ECS data. The author would also like to thank S.P. Meyer and L.A. Spence for numerous hours of tree measurement. Financial support of S.P. Meyer and L.A. Spence through the Michigan Agricultural Experiment Station, at Michigan State University and U.S.D.A. McIntire-Stennis Cooperative Research Funds allowed for tree measurement data to be collected.

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EVALUATION OF MULTIPLE FIXED AREA PLOT SIZES AND BAFs IN EVEN-AGED HARDWOOD STANDS

John R. Brooks and David W. McGill[†]

ABSTRACT.—A test that utilized fixed area plot size and basal area factor as a continuous variable to determine the relationship with stems (TPA), basal area (BAAC) and board foot volume per acre (BFAC) was applied to even-aged mature (70-year-old) and immature (12-year-old) hardwood stands located in northern West Virginia. For the 70-year-old hardwood sites TPA, BAAC, and BFAC decreased with increasing plot size and converged for fixed area plot sizes greater than 0.08 acre. For BAFs between 10 and 40 these variables increased in a linear fashion with BAF and did not converge. The coefficient of variation (CV) for TPA and BAAC were very similar and less than that for BFAC over the range of fixed area plot sizes and BAFs examined. The CV for all variables decreased with increasing plot size and increased with increasing BAF. In the 12-year-old stands, TPA and BAAC decreased with increasing plot size and converged for plot sizes larger than 0.06 acre. These variables decreased in a linear fashion with increasing BAF between BAFs of 1 and 40 and did not converge. The CV for BAAC was larger than that for TPA in the fixed area plot sizes tested while the reverse was found to be true for point sampling.

Early forest inventory techniques in the United States were based on the use of fixed area sampling units, typically 0.20 and 0.25 acre in size. Fixed area sampling units were the status quo for many years until Grosenbaugh introduced the concept of point sampling in early 1950's. This sampling system changed the way foresters inventoried forest populations fueled primarily by the obvious time savings associated with this technique. Early tests by Grosenbaugh and Stover (1957) in Texas showed that using a BAF of 10 did not differ statistically from estimates based on 0.25 acre and 0.20 acre plots. Though not widely accepted at the time, Husch (1955) and Clutter (1957) tested a limited number of BAFs and found that average per-acre estimates of basal area increased with increasing BAFs. Only the larger BAFs appeared to give unbiased estimates. Zeide and Troxell (1979) cite several references where small BAFs resulted in underestimation of basal area that they concluded was due to missing in-trees. The main reasons given for this apparent anomaly included the existence of heavy understorey, poor light conditions, hidden trees and miscounting trees when the number of in-trees was large. In a test of BAF 5, 10, 20, and 40 in a 70-year-old Appalachian hardwood forest in northern West Virginia, Wiant et al. (1984) found that both basal area per acre and board foot volume per acre increased with increasing BAF. Their test results were compared with repeated sampling of this tract using 0.10 acre plots. Both BAF 5 and BAF 10 estimates of volume were statistically less than the estimates using BAF 20, BAF 40 and 0.10 acre plots. They recommend using BAFs of 20 or 40 for sawtimber cruises in this region. During the last decade, foresters have slowly migrated to using a BAF of 20 in sawtimber inventories spurred by the empirical evidence that it provides less biased estimates of stand volume but more likely due to the fact that fewer in-trees would be measured thus saving field data collection time. During this same period the use of 0.10 acre fixed area circular plots also became the norm, although it is not employed as frequently as point sampling techniques.

Methods

Two Appalachian hardwood sites were selected from the Eastern Allegheny Plateau and Mountains land resource region (USDA SCS 1981) and are located in Monongalia and Preston counties, in northern West Virginia. The first location (Mature Hardwood Sites) consists of 40 plots located in 70-year-old even-aged mixed hardwood stands. These plots represent permanent sample plots distributed across both mixed oak and mesophytic hardwood sites (SAF cover types 52 and 59)(Eyre 1980). The individual plots in this site are fairly uniform with a coefficient of variation ranging from 0.31 to 0.51 for trees per acre and board foot volume per acre, respectively (Table 1). The second location (Young Hardwood Sites) consists of 18 plots established in 12-year-old even-aged stands situated on a middle slope having a northwest aspect. Plots on this site are very uniform with a coefficient of variation of 0.22 for trees per acre and 0.14 for basal area per acre (Table 1).

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Table 1.—Descriptive statistics for the mature and young hardwood study sites.

	Mature Hardwood Sites ¹				Young Hardwood Sites ²			
	Min	Max	Mean	CV	Min	Max	Mean	CV
TPA	10.0	110.0	72.2	0.31	1,690.0	3,960.0	2,773.3	0.22
BA	9.4	187.4	112.4	0.3651.3	88.2	75.4	0.14	
BF	1,173.8	33,885.5	16,482.7	0.51				
QMD	13.2	20.6	16.7	0.101.7	2.9	2.3	0.14	

¹based on a fixed area plot size of 0.2 acre

²based on a fixed area plot size of 0.1 acre

Mature Hardwood Sites

Forty permanent plots were visited and every tree within a 70-foot radius of plot center was measured for dbh, sawlog merchantable height, and distance from plot center. The distance from plot center to each tree was determined using an Impulse 200 laser hypsometer by shooting to a reflective target positioned at dbh and facing the plot center. The distance to the pith was determined by adding half the dbh in feet as determined with a diameter tape. This distance was recorded to the nearest 0.01 foot. Tree dbh was measured with a steel diameter tape and recorded to the nearest 0.1 inch. Since this study involves sawtimber products, only those trees 11.6 inches dbh and larger were included in the analysis. Merchantable height of the sawlog portion of the tree was measured using an Impulse 200 laser hypsometer and recorded to the nearest foot. Sawlog merchantable height was defined as height to an 8-inch dob top or where branching or form limited merchantability for this product. Merchantable height data were collected to estimate board foot contents of each stem using equations published by Scott (1979). Since these plots are centered at an existing square 0.2 acre permanent sample plot, most trees tallied were previously tagged. Once the tagged trees were identified and measured, a sweep of the area for sawtimber trees outside the existing plot but within the 70 foot radius were identified and measured making it quite unlikely that any trees were missed. At each plot, TPA, BAAC and BFAC (Int ¼) were determined for fixed area plots ranging in size from 0.01 to 0.30 acre in 0.01 acre intervals. In addition, the same estimates were made using BAFs from 1 to 50 in 1 BAF intervals. For this study, any tree whose distance from plot center was less than or equal to the limiting distance for that specific sampling unit was included in the tally.

Young Hardwood Sites

Eighteen 0.10 acre circular plots were established in 12-year-old even-aged stands. Every tree taller than 4.5 feet and within a 37.24-foot radius of plot center was measured for dbh and distance from plot center. Tree measurements were conducted using the same procedures used for trees in the Mature Hardwood Sites. At each plot, TPA and BAAC were determined for fixed area plots ranging in size from 0.01 to 0.10 acre in 0.01 acre intervals. In addition, the same estimates were made using BAFs from 1 to 40, in 1 BAF intervals. As with the Mature Hardwood Sites, any tree whose distance from plot center was less than or equal to the limiting distance for that specific sampling unit was included in the tally. Plot data were summarized for each fixed area plot size and BAF examined. Mean trees and basal area per acre as well as the coefficient of variation and the average number of in-trees were determined for this sample of 18 plots.

Results

Mature Hardwood Sites

Figure 1 depicts the mean TPA, BAAC, BFAC, number of in-trees, and coefficient of variation for TPA, BAAC and BFAC for the 40 permanent sample plots described previously. Mean BFAC decreased with plot size but stabilized for fixed area plot sizes greater than 0.08 acre. The small variations in volume per acre for plot sizes from 0.10 to 0.26 acre are most likely due to spatial variation of trees among the different plot sizes. The coefficient of variation for BFAC decreased with increasing plot size but only changed 10 percentage points (0.6 to 0.5) between plot sizes of 0.13 to 0.30 acre. A similar pattern is portrayed for estimates of TPA. Trees per acre stabilized for plot sizes larger than 0.09 acre. The coefficient of variation stabilized for plot sizes larger than 0.06 acre. As expected, the graph of BAAC by plot size is very similar to that of BFAC. The average number of in-trees per plot increased linearly with increasing plot size from 1

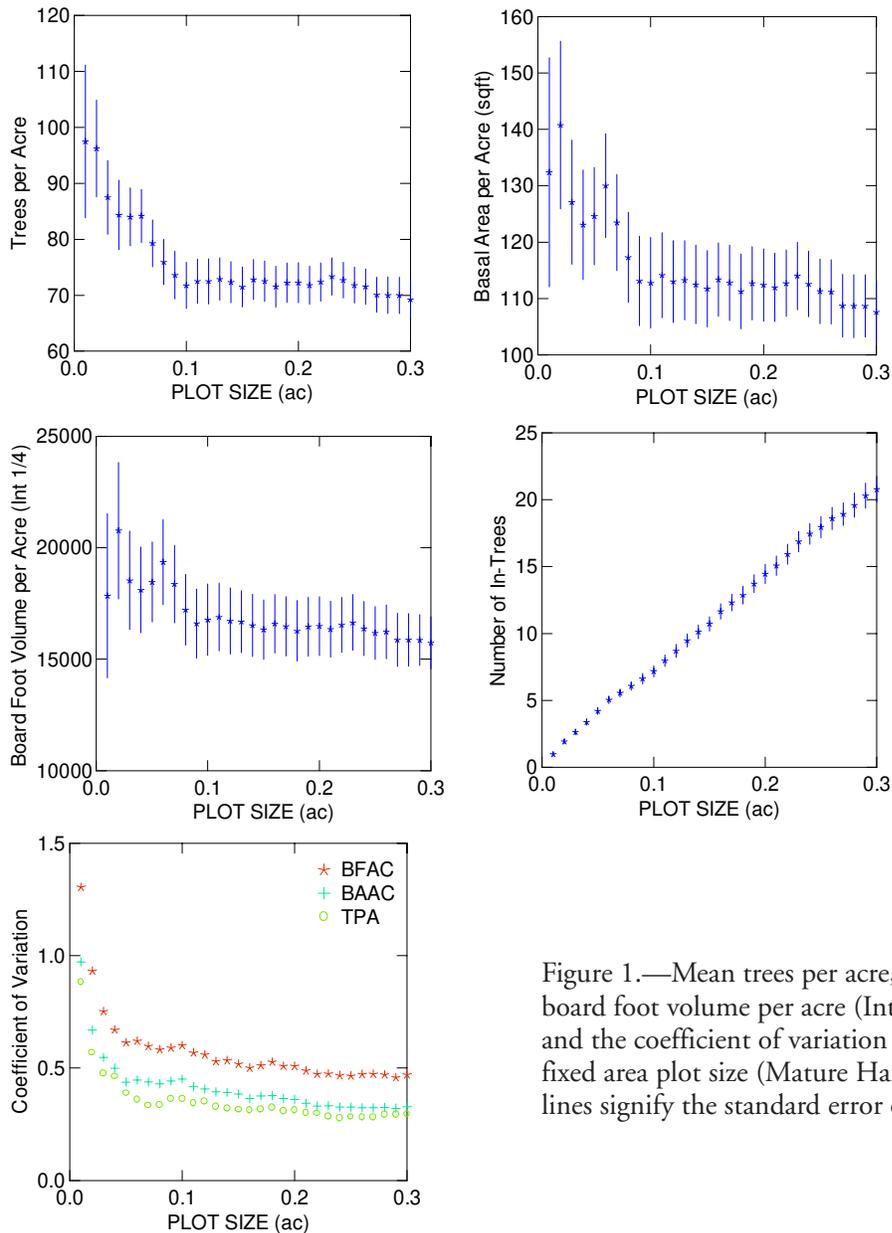


Figure 1.—Mean trees per acre, basal area per acre, board foot volume per acre (Int 1/4), number of in-trees and the coefficient of variation by stand parameter by fixed area plot size (Mature Hardwood Sites). Vertical lines signify the standard error of the mean.

tree (0.01-acre plot) to 19 trees (0.30-acre plot). The vertical lines depicted in Figure 1 signify the standard error of the mean for that sample size. A SAS General Linear Models test was conducted to determine whether the means for TPA, BAAC or BFAC were the same for all plot sizes ($\alpha = 0.05$). Only TPA indicated significantly different means. A mean separation test (Tukey) was unable to identify differences in mean TPA.

As mentioned previously, BAF was increased in units of 1 BAF starting with a BAF of 10 and culminating with a BAF of 50. Based on previous work in this region, it is realistic to believe that an appropriate BAF would fall within this range. Figure 2 shows the mean TPA, BAAC, BFAC, number of in-trees, and coefficient of variation for TPA, BAAC and BFAC with increasing BAF. The pattern shown in Figure 2 is typical of the pattern described by Wiant et al. (1984) as well as other investigators, but in this case it is very unlikely that this anomaly can be explained by missing in-trees. The irregular pattern shown between BAF 40 and 50 is currently unexplained. The coefficient of variation for BFAC increased with increasing BAF from 0.48 (BAF 10) to 0.69 (BAF 50). However, this increase is relatively small between BAF 13 and 30. The estimates for TPA increased in a consistent manner from BAF 15 to BAF 40, after which there is an unexplained drop in TPA. The coefficient of variation for TPA increased linearly with increasing BAF

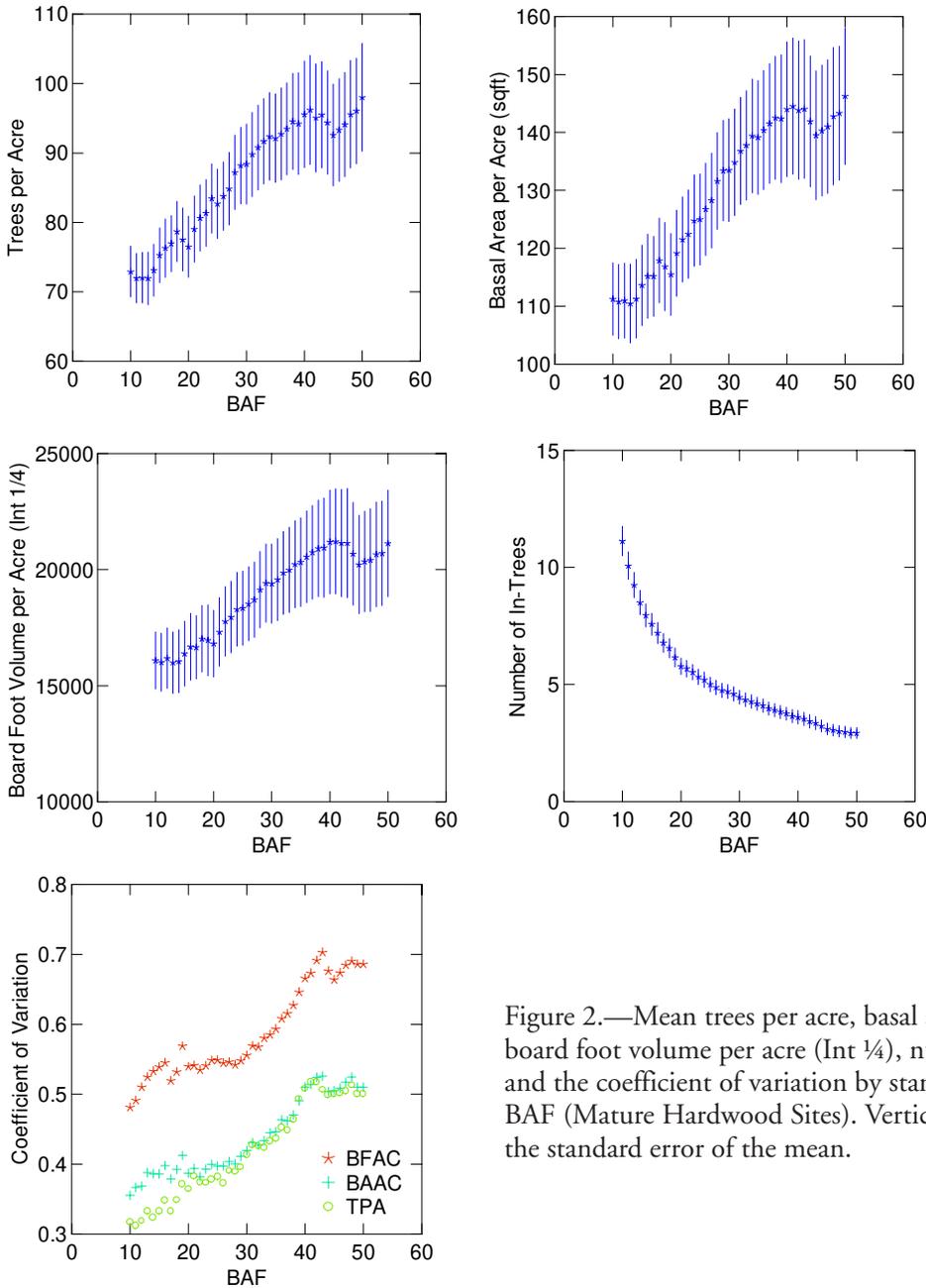


Figure 2.—Mean trees per acre, basal area per acre, board foot volume per acre (Int 1/4), number of in-trees and the coefficient of variation by stand parameter by BAF (Mature Hardwood Sites). Vertical lines signify the standard error of the mean.

up to a BAF of 40. There seems to be very little increase in variation above a BAF of 40. As with fixed area plot samples, BAAC followed a similar trend to BFAC. The mean number of in-trees per point decreased exponentially with increasing BAF with an average of 11.1 in-trees with BAF 10 and approximately 3 in-trees for BAF 50. A SAS General Linear Models test was conducted to determine whether the means for TPA, BAAC or BFAC were the same for all BAFs ($\alpha = 0.05$). Only BAAC and TPA indicated significantly different means. A mean separation test (Tukey) was unable to identify differences in either mean BAAC or TPA.

Young Hardwood Sites

Figure 3 depicts the mean TPA, BAAC, number of in-trees and the coefficient of variation for TPA and BAAC with increasing plot size. Although these estimates are based on fewer plots (18), BAAC and TPA appear to stabilize above a fixed area plot size of 0.06 acre. The number of in-trees required for each plot size increases linearly ranging from 10 (0.01-acre plot) to nearly 300 (0.10-acre plot). The mean number of

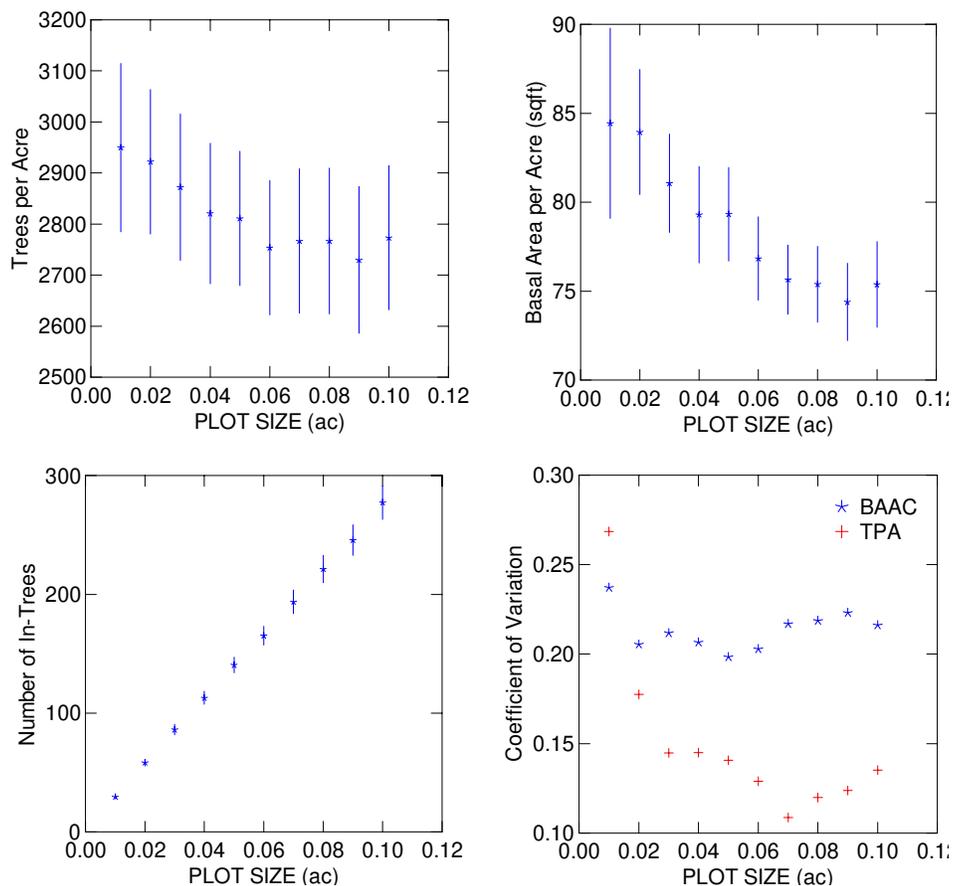


Figure 3.—Mean trees per acre, basal area per acre, number of in-trees and the coefficient of variation by stand parameter by fixed area plot size (Young Hardwood Sites). Vertical lines signify the standard error of the mean.

in-trees for a 0.06 acre plot was approximate 150 trees. No significant differences between sample means were found for BAAC or TPA.

The results obtained using point sampling are exactly opposite that found for the Mature Hardwood Sites. In this situation, BAAC decreased with increasing BAF (fig. 4). The same relationship was observed for the mean TPA. As was found with point sampling in the Mature Hardwood Sites, the number of in-trees decreased exponentially with increasing BAF ranging from 75 (BAF 1) to 1 (BAF 40). No significant differences were found between sample means.

A review of the coefficient of variation by plot size and BAF indicated that TPA is more variable than BAAC for point sampling but not for fixed area plots. These values stabilize relatively quickly in small fixed area plots and do not stabilize at all for different BAFs.

Conclusions

The effect of increasing plot size and BAF on BFAC, BAAC and TPA was examined on age 70 (Mature Hardwood Sites) and age 12 (Young Hardwood Sites) even-aged hardwood sites in northern West Virginia. Statistically significant differences were found for BAAC and TPA for the range of BAFs examined. For the range of fixed area plot sizes tested, only TPA showed a significant plot size effect. Mean separation tests (Tukey) were unable to identify significantly different means. On the Mature Hardwood Sites, average BFAC stabilized for fixed area plot sizes larger than 0.08 acre. Utilizing the same plot centers, board foot

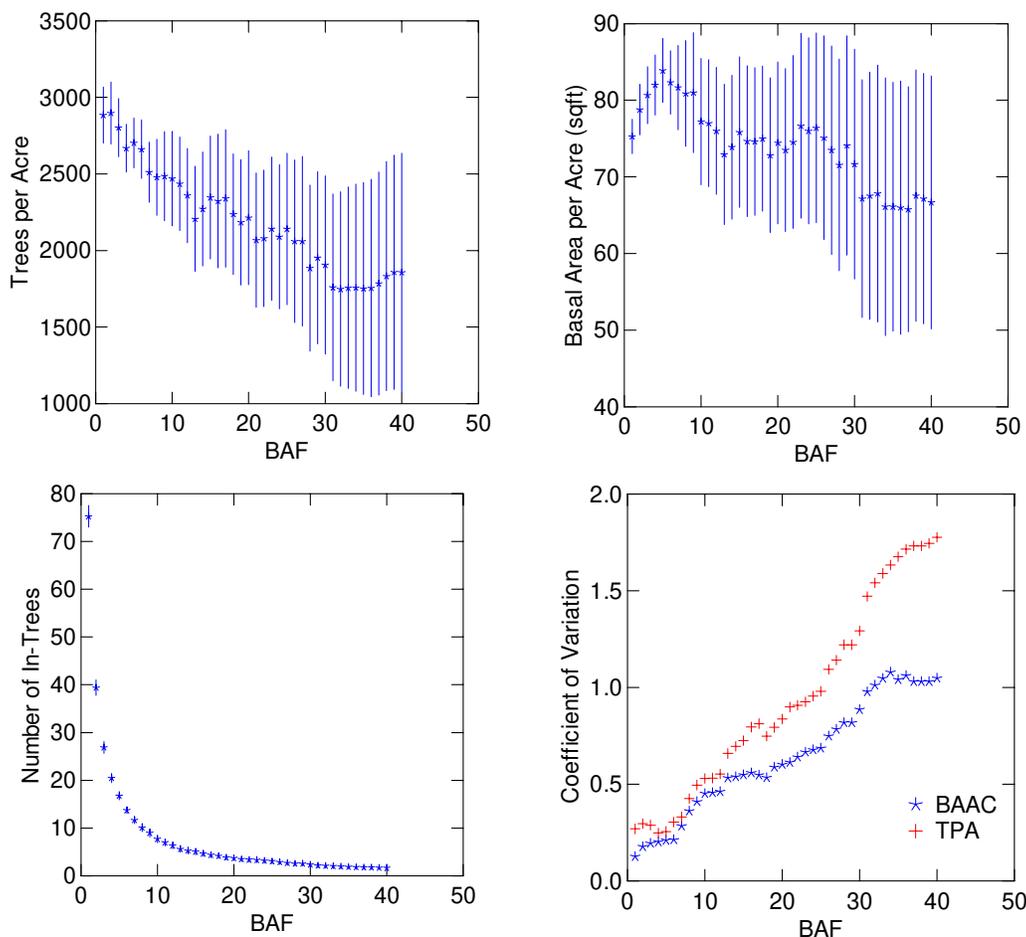


Figure 4.—Mean trees per acre, basal area per acre, number of in-trees and the coefficient of variation by stand parameter by BAF (Young Hardwood Sites). Vertical lines signify the standard error of the mean.

volume increased linearly with increasing BAFs between 10 and 40. These results support the findings of earlier investigators (Clutter 1957, Zeide and Troxell 1979, Wiandt et al. 1984) but it is unlikely that the results are due to missing in-trees, thus indicating a bias associated with the implementation of point sampling on these sites. If one would assume that the “true” value for the parameter of interest is identified by the values from the fixed area plot graphs where the means flatten out and become less variable, then an estimate of the “true” board foot volume per acre is around 15,500 bf/ac and the “true” basal area per acre is approximately 105 ft²/ac. This would indicate that the appropriate BAF to use would be between 16 and 20 for board foot volume and basal area per acre. Either of these situations would require less than 7 in-trees per point, on average. It is interesting to note that if the “true” volume is as described above, selection of a BAF of 40 would have resulted in an overestimation of volume by 26 percent. The coefficient of variation for BFAC was larger than that for BAAC or TPA for the fixed area plot sizes and the different BAFs examined. Additional study is recommended as an increase in sample size would likely result in an increased ability to distinguish significant differences between sample means.

Less prior work has been done regarding the appropriate plot size or BAF to use when sampling immature stands, due primarily to their lower inherent value. In general, foresters have employed smaller plot sizes in younger stands, though the reasoning may be more related to practicality than in selecting an unbiased estimate of the variable of interest. In this relatively small sample, no statistical difference was found between the means for BAAC or TPA for the range of BAFs and fixed area plot sizes examined. Average basal area and trees per acre stabilized in fixed area plot sizes larger than 0.06 acre. However, the average

number of in-trees at this plot size is approximately 150. In general, mean BAAC and TPA decrease with increasing BAF in these stands. It is the same phenomenon obtained with the Mature Hardwood Sites, but with a negative slope. Again, if one would assume that the “true” value for the parameter of interest is identified by the values from the fixed area plot graphs where the means flatten out and become less variable, then an estimate of the “true” basal area per acre is around 75 ft²/ac and the “true” trees per acre is approximately 2,750. This would indicate that the appropriate BAF to use would be between 10 and 27 for BAAC and 3 and 6 for TPA. Either of these situations would require less than 20 in-trees per point, a value much lower than that needed for fixed area plots. For the fixed area plots examined, the coefficient of variation for BAAC was larger than that for TPA. The exact opposite was found for the BAFs tested. Additional investigation is needed in immature hardwood stands to determine whether these relationships hold over larger sample sizes and different site types. Care must be taken in selecting an appropriate BAF in any age stand as the variable of interest has been found to change linearly with BAF in all sites examined.

Acknowledgment

This research, in part, was supported by the West Virginia Agricultural and Forestry Experiment Station. Scientific Article No. 2861.

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A COMPARISON OF TWO DOUBLE SAMPLING AUXILIARY VARIABLES WITH INVENTORIES OF VARYING SAMPLE SIZE

Wesley W. Bailes and John R. Brooks[†]

ABSTRACT. —Two auxiliary variables were tested to determine the effects on the precision statement about the mean board foot volume per acre. These comparisons were made with data from three inventories of differing sizes in West Virginia. The sum of the merchantable height on all “in-trees” per point was compared with the more commonly employed auxiliary variable of basal area per acre. The auxiliary variable based on the sum of the merchantable height on “in-trees” resulted in higher correlations with board foot volume per acre than the use of basal area in all three inventories. This increase in correlation did not produce a substantial decrease in the standard error of the mean. A time study was also conducted to investigate the additional time required to estimate the number of logs on “in-trees” per point when compared to just recording basal area per acre. On average, estimating merchantable height on “in-trees” took an additional 13 seconds per point.

Double sampling, also referred to as two-phase sampling, was originally mentioned in forestry literature by Spurr (1952), while the first detailed description of its application was presented by Freese (1962). This sampling technique was developed to take advantage of the relationship between the variable of interest and some easily measured and highly correlated auxiliary variable. The primary objective of double sampling is to reduce total inventory time without sacrificing the precision about the point estimate. This can be accomplished through the use of a carefully selected auxiliary variable. A prerequisite for a good auxiliary variable is one that is easily and thus quickly measured and is also highly correlated with the variable of interest. The most commonly utilized auxiliary variable with double sampling for volume estimates is basal area. This is due to the high correlation between basal area and volume and the fact that basal area can be determined very quickly when using point sampling. Although basal area is an excellent candidate for double sampling, other auxiliary variables can be used. In a New Jersey study by Wiant and Zeide (1979), the sum of the estimated merchantable logs on “in-trees” was initially examined for use as an auxiliary variable. Their results indicated that the sum of the number of logs on “in-trees” provided a higher correlation with board foot volume than basal area per acre. This is a plausible statement since a 20-inch tree with 3-logs and a 20-inch tree with 1-log have the same basal area, but different merchantable heights, and thus different volumes. Another study that utilized merchantable height as an auxiliary variable was reported by Myers (1985) for a simulation study conducted in a 36-year-old even-aged hardwood forest located in northern West Virginia. He concluded that the use of the sum of estimated merchantable logs as the auxiliary variable provided a more precise estimate of board foot volume than basal area using repeated sampling with samples sizes ranging from 20 to 60. Although limited testing of this auxiliary variable appears promising, its use in large-scale forest inventory applications has not been documented. This study was designed to investigate the use of basal area per acre and the sum of the merchantable heights on “in-trees” as auxiliary variables using several inventories that included both large and small sample sizes. In addition to determining the efficiency of these auxiliary variables, field data recorders were used to record point measurement times for the intensively measured points as well as both types of non-intensive points on one of the larger inventories.

Methods

Three separate inventories were utilized as the basis for this analysis. All sample inventories were based on a systematic point sample using a BAF of 20 to inventory sawtimber board foot volume on approximately 70-year-old hardwood forests in West Virginia. Actual inventory area ranged from 11,995 acres to 138 acres (Table 1).

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Table 1.—Tract size, sample size and intensive to non-intensive sampling ratios for three hardwood forest inventories in West Virginia.

Inventory	First Phase (n)	Second phase (n)	Actual ratio	Forest Size (acres)
Coopers Rock	3,124	1,041	1:3	11,995
Kumbrabow	530	216	1:2.45	8,683
Compartment 14	52	18	1:2.89	138

The Coopers Rock inventory, consisting of 11,995 acres, was inventoried during 2000-2001. This inventory includes both the West Virginia University Research Forest (7,594 acres) and a subset of the Coopers Rock State Forest (4,401 acres) located in northern West Virginia. The Kumbrabow inventory was conducted during the summer of 2002 and represents a majority of the area (8,683 acres) that comprises Kumbrabow State Forest located in Randolph County, West Virginia. The third inventory was included to examine the effects of inventory size on the precision statement and is based on a compartment level inventory (Compartment 14) conducted on the West Virginia University Research Forest in 2003. The Kumbrabow inventory was initially designed to test the efficiency of these two auxiliary variables while the Coopers Rock and Compartment 14 inventories were initially comprised of all intensively measured points that were converted to a double sample inventory with an approximate 1 to 3 intensive to non-intensive ratio. The actual plot ratios for each inventory are displayed in Table 1. In each sample inventory both auxiliary variables are known on every point. For the Kumbrabow and Coopers Rock inventories, board foot volume was based on equations by Scott (1979) which are a function of merchantable height in feet. The Compartment 14 inventory utilized form class 78 equations developed by Wiant and Castaneda (1977) where merchantable height was recorded in logs and half logs. All board foot volumes are based on the International ¼-inch log rule.

Results

The relationship between each auxiliary variable and board foot volume per acre was examined on the second phase samples. A ratio estimator was selected since the data appeared to cross at the origin and exhibited increasing variance with increasing values of the auxiliary variable (Figures 1 through 3). There was an increase in the correlation coefficient in all three inventories due to utilization of sum of merchantable heights as the auxiliary variable (Table 2). The estimates of mean board foot volume for the two auxiliary variables produced similar mean values (Table 3). Since the Coopers Rock and the Compartment 14 inventories were originally based on all intensively measured points, the actual mean volume per acre is known to be 10,686 and 12,657 board feet per acre, respectively. In both cases, the sum of the merchantable heights on “in-trees” as the auxiliary variable provided less biased estimates of the mean volume per acre. There was not a substantial increase in precision from utilizing merchantable height as an auxiliary variable (Table 3). The largest increase in precision occurred in the Compartment 14 inventory, which was 1.1 percent lower than that obtained using basal area as the auxiliary variable (Table 3). An evaluation of the mean time to measure each non-intensive plot type indicated that estimating the merchantable heights on “in-trees” took only an additional 13 seconds per point.

Conclusions

Based on a 1:3 ratio of intensive to non-intensive sample points, the use of merchantable height as an auxiliary variable increased the correlation coefficient with board foot volume per acre by 3.8 to 9.8 percent in all three inventories. However, this increase in correlation with volume was not large enough to substantially reduce the standard error of the mean. Although the standard error was reduced in each of the three inventories, reduction only ranged from 0.02 to 1.12 percentage points. Based on a comparison of those inventories where complete enumeration of all points was available, the use of the sum of merchantable heights on “in-trees” as the auxiliary variable resulted in slightly less biased estimates of the mean board foot volume per acre. Additional investigation of these auxiliary variables utilizing intensive to non-intensive ratios with fewer intensive points and their effects on the estimated mean and standard error is recommended.

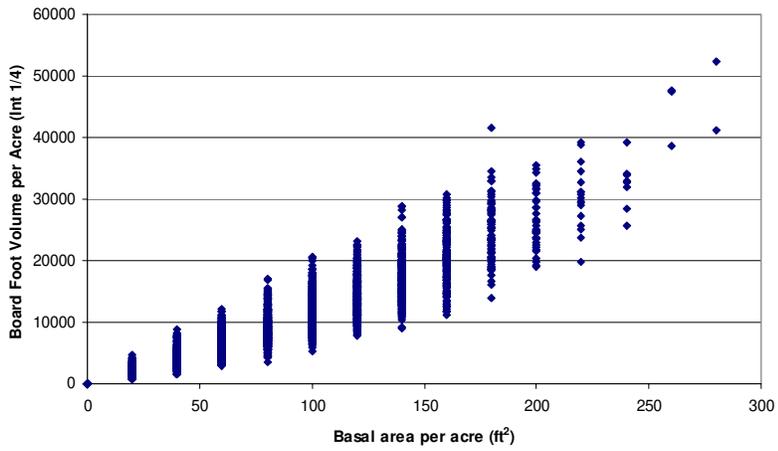


Figure 1.—The relationship between the sum of the merchantable heights on “in-trees” and basal area per acre with board foot volume per acre for the Coopers Rock inventory.

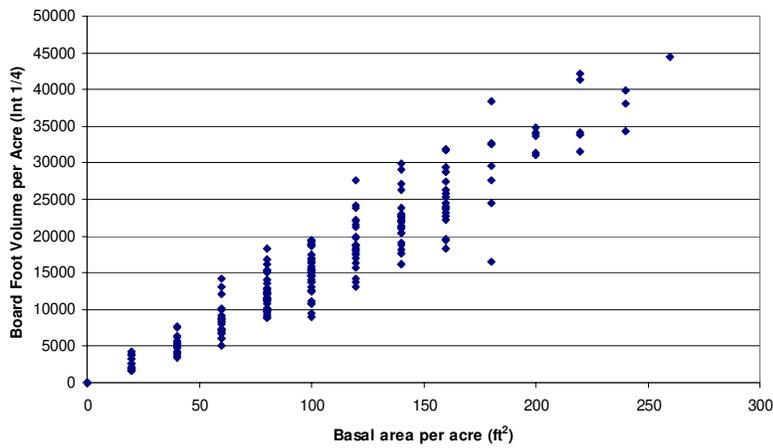
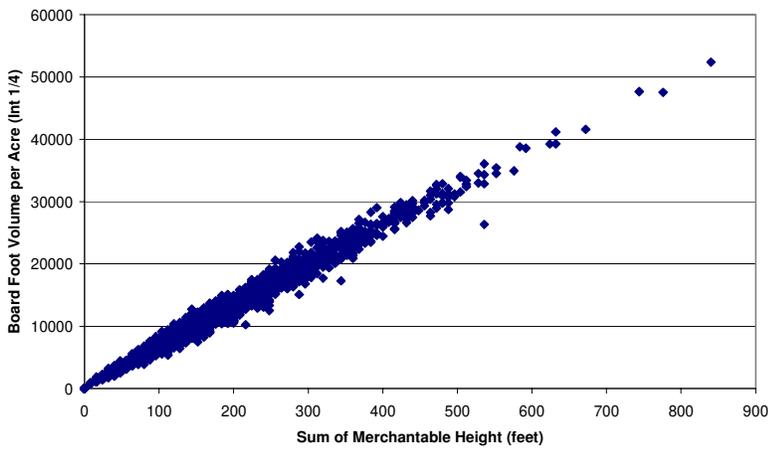


Figure 2.—The relationship between the sum of the merchantable heights on “in-trees” and basal area per acre with board foot volume per acre for the Kumbrabow inventory.

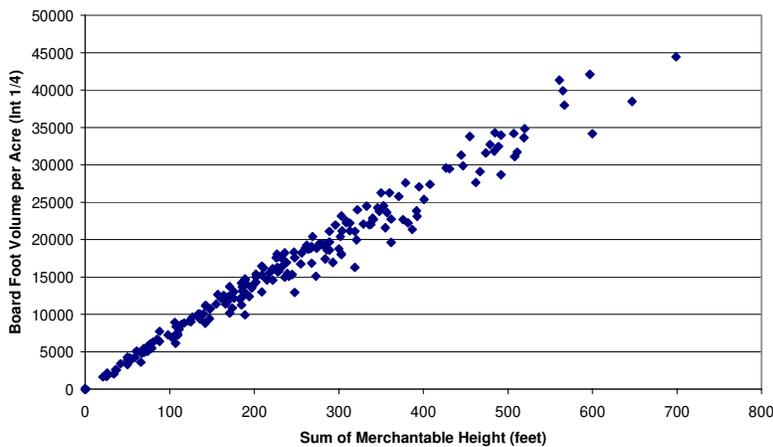


Table 2.—Correlation coefficients by auxiliary variable and inventory.

Auxiliary Variable	Coopers Rock	Kumbrabow	Compartment 14
Basal Area (BA)	0.910	0.952	0.908
Merchantable Height (MHT)	0.991	0.988	0.997

Table 3.—Mean estimates and statistics by auxiliary variable and inventory.

Statistic	Coopers Rock	Kumbrabow	Compartment 14
Mean with BA	10,826	14,445	12,345
Mean with MHT	10,693	14,260	12,504
Standard Error with BA	149	435	1,079
Standard Error with MHT	130	427	953
Standard Error BA (%) ¹	1.38	3.01	8.74
Standard Error MHT (%) ¹	1.22	2.99	7.62

¹Standard error as a percent of the mean

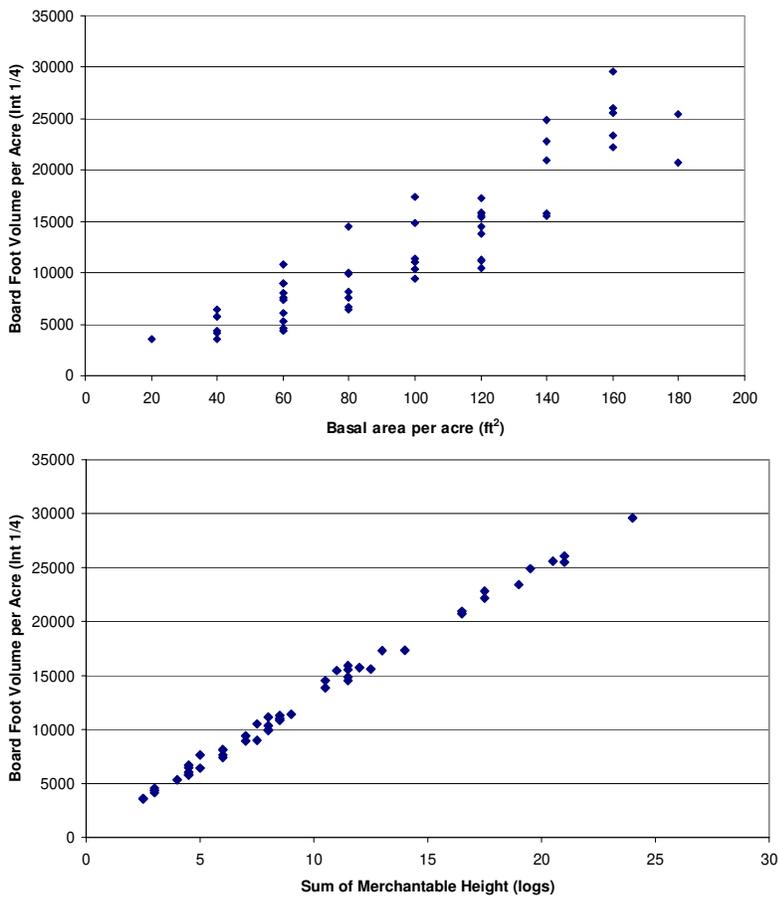


Figure 3.—The relationship between the sum of the merchantable heights on “in-trees” and basal area per acre with board foot volume per acre for the Compartment 14 inventory.

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GROWTH STRESS IN HARDWOOD TIMBER

Daniel L. Cassens and Jose R. Serrano[†]

ABSTRACT.—Longitudinal growth stress, particularly in small diameter trees, can result in severe splitting and warping of lumber and can even ruin veneer quality logs. This phenomenon is well documented in the literature. The problems it causes are common and occur wherever wood or veneer is being processed. However, foresters, industrial managers, and researchers generally do not understand its significance. This paper will present examples of wood manufacturing problems resulting from longitudinal growth stress. It will also briefly review the literature and present findings from a study measuring longitudinal growth strain in different plantations of pole sized yellow poplar and the relationship of growth strain to warp and checking in lumber. Growth strain should be considered in future genetic studies.

Longitudinal growth stresses are present in all standing timber and cut logs. In fact, if they did not exist, trees could not maintain a vertical position. Growth stresses are not visible although they can be measured and are called growth strain (GS). When trees are felled and cut into logs and logs processed into lumber the results of growth stresses being released, particularly in small diameter timber become evident. Growth stress-related degrade includes splits, cracks and warp and thus loss of potential economic value. This degrade is so commonplace that many individuals will dismiss it as being “characteristic of wood” or “wood just does that”.

Heart checks in the butt of felled trees and cut logs are the first evidence of the presence of longitudinal growth stress relief in timber. These checks will occur as soon as the tree is felled or the log is bucked and may increase with time. In severe cases, logs may literally quarter themselves (fig. 1). “S” or “C” irons or plastic “I” pieces are commonly used on high valued logs to help control the problem.



Figure 1.—High quality walnut veneer log which has been severely degraded due to the release of growth stress.

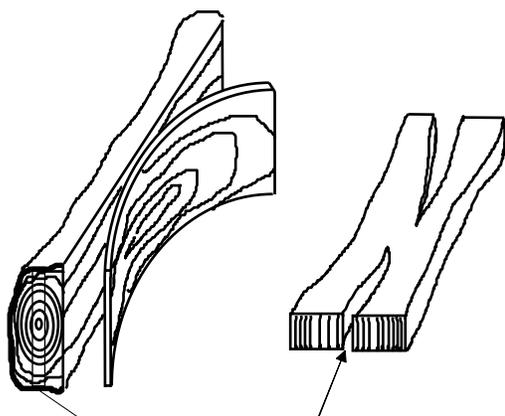


Figure 2.—Bow from longitudinal growth stress release in a flat sawn board (left). Tension in the outer portion of the log pulls the ends of the board outward and together. Longitudinal growth stress release in a diametrical edge grained (quartersawn) blank resulting in deep end splits and crook.

Chain saws may bind when cross cutting logs (Wilhelmy 1971) due to stress. As the saw first enters the cross cut, tension in the outside of the log pulls the wood away but as the cut deepens the center portion of the log which is in compression is now free to expand and the saw is pinched.

Bow in flat grained lumber occurs when tension in the outer part of the log cause boards to pull outward as they are cut loose from the log or cant (fig. 2). The log or cant will curve slightly in the opposite direction. If another board is cut on the same log face, it will be thinner on the ends and thicker in the middle due to movement of the cant. At mills which cut logs one board at a time, the cumulative effect of thin ends and thick middles is seen as a slight or low arch at the top of tall lumber stacks on the air drying yard. The effect is particularly noticeable in long lumber

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Crook often occurs in edge grained or quartersawn lumber. The region near the bark is in tension and tends to shrink while the region near the pith expands due to release of compression forces. The region near the bark ultimately ends up in compression (Jacobs 1965).

Boards and timbers cut with a centered pith or boxed heart may also develop deep end checks or splits (fig. 2), sometimes with an explosive force. Simply dropping one of these pieces during processing can result in the split developing.

Although growth stresses can often explain at least in part bow and crook, warp is usually the result of several growth characteristics, processing procedures, and wood properties interacting with growth stresses. These may include juvenile wood, pith, drying stresses, wood density and stiffness as well as anatomical alterations. Some drying stresses may act to offset growth stresses. Because several factors may be interacting it is difficult to identify a single cause for splitting and warping.

In order to better understand longitudinal growth stress and its effect on logs and lumber a series of experiments were conducted. Unlike previous reports, this research measured longitudinal growth stresses in standing trees and cut logs and then related the amount of stress to warp and checking in the lumber produced. Serrano (1999) provides additional information on the relationship of stress to warp and end checking in air dried and surfaced lumber. A complete statistical analysis and model are presented in the same document.

Literature Review

Jacobs (1945, 1965) defined growth stresses as the forces found in green stems disregarding stresses due to the weight of the tree's crown and sap tension. According to Post (1979) growth stresses are the forces that develop in the wood of growing plants. These stresses are different from those developed as a consequence of drying, although they may interact (Jacobs 1965, Chafe 1979A and B, Kubler 1987). Growth stresses in wood are distributed orthotropically in longitudinal (axial), tangential (ring, circumferential) and radial directions in the stem. Growth stresses originate when growing wood cells contract in the longitudinal direction and expand in the transverse direction (Munch 1938). Since the contraction is restrained by older cells, the new cells generate longitudinal tension, while obstruction of the lateral expansion by neighbor cells leads to tangential compression stress (Kubler 1987). The longitudinal tension force compresses the adjacent interior layer and reduces the tension of older cells (Jacobs 1939). The macro-cumulative effect of additional woody layers is converted to severe compression near the pith.

The tree generates growth stresses in response to mechanical requirements. Growth stresses are important in that they help reorient stems and branches. For example, reaction wood generally tends to exhibit high levels of longitudinal growth stresses that allow leaning trees to regain a vertical position. Growth stresses are present in the tree as internal or residual stress even without the action of external forces. As a result, the mean growth stress in vertical standing trees and the corresponding felled trees differ only slightly (Nicholson 1973). Conversely, the pattern of stress distribution around the circumference changes significantly due to lean and wind. The release of growth stresses during processing generates many complications such as splits and distortions in both logs and lumber.

Stress appears in woody plants in the form of tension and compression. The stress acting on wood produces strain, which is a change in dimension. Since stress cannot be measured directly, strain is used to evaluate defects caused by growth stress (Kubler 1987). In the longitudinal direction, the tree is pre-stressed in tension on the periphery and in compression near the center. Compressive longitudinal growth stresses tend to increase toward the pith and when felling and crosscutting, the internal-stress balance is altered resulting in splits and heart checks in the log ends (Malan 1979). Longitudinal growth stresses are largest in magnitude and the most serious in processing because they result in warp (Boyd 1950, Hillis 1984, Maeglin 1987, Kubler 1988). Warp in lumber is fundamentally driven by growth stresses, and differences in shrinkage within the wood as it dries (Simpson 1983). Warp in sawn green timber occurs essentially due to differential or asymmetry in growth stress levels. Since growth stress release causes warp and splits in green lumber, researchers are interested in finding ways to predict or to reduce its impact. To predict lumber warp, surface longitudinal strain measurements in standing trees and felled logs have been suggested as an

alternative to the more complicated inside-stem strain determination method (Okuyama and Sasaki 1979, Archer 1986 and Kubler 1987). Because growth stress release will occur during the conversion process, sawing pattern plays an important role in warp development. Longitudinal position in the tree may also influence lumber warp (Balaban 1982, Maeglin and Boone 1983, Koch 1986). Other causes of lumber warp might be due to the anisotropic nature of wood as well as the variability of wood properties within and among trees. As a result, growth stresses and growth factors may increase or decrease the tendency of lumber to warp. Kubler (1987) and Archer (1986) note that large variations of surface growth stresses occur within and between species, around the tree circumference especially in leaning trees (reaction wood) as well as with tree height, silvicultural treatments and seasons. However, only limited experimental information has been generated. In addition, surface growth strains have been found to be closely related to variations of wood properties such as, MOE, basic density, shrinkage, fiber wall thickness, lignin/cellulose content, and microfibril angle (Nicholson et al. 1975, Okuyama et al. 1994, Bailleres et al. 1995, Hillis 1997, Malan 1997). Most of these wood properties show strong relationships with growth stress when evaluated in leaning trees.

Today, second growth forests along with fast growth plantations represent an important source of many commercially important species. As a result, the size and quality of logs are continuing to decrease, resulting in utilization of smaller and smaller trees with less mature, clear wood. Moreover, wood which was initially managed for pulp is being processed for lumber even at younger ages (Senft et al. 1985). These younger trees have a higher proportion of juvenile wood, reaction wood, and growth stress as compared to older trees (Maeglin 1987). Growth stresses are more critical in smaller logs from young fast growing trees because of steeper growth stress gradients (Boyd 1950, Hillis 1997). In addition, growth stress release in hardwoods is of a much greater magnitude than in softwoods (Archer 1986, Kubler 1987). Well documented differences have been observed among the properties of wood from second growth forests, plantations, and old growth forests (Rendle 1960, Bendtsen 1978, Lewark 1985, Zobel and van Buijtenen 1989). As timber demand and resulting economic pressures impose short rotations, more quality problems with shrinkage, warping, low strength and other factors, impacting the performance of wood products will be found.

Materials And Methods

Surface longitudinal growth strain (GS) of 36 trees was measured during May and September on the west and east side of a sample of plantation grown yellow poplar (*Liriodendron tulipifera* L.) trees from three different plots at the Purdue University Martell Forest in Tippecanoe County, Indiana. Trees in the plots were planted in 1965 and initially spaced at 6x6 or 6x9 feet. Dominate trees were selected. The plots were on level ground. An additional set of the GS values were taken in August and included measurements on the west, east, north and south sides of the stem. Twenty of these trees were felled and processed into lumber. GS measurements were taken at diameter breast high (DBH) on the butt logs as well as at 12.50 feet from the ground on the second logs. Side-matched GS measurements were obtained on the felled trees (two 8 foot logs). A slightly modified Nicholson's method (fig. 3) was employed to determine GS measurements (Nicholson 1971 and Serrano 1999). Growth strain is measured as $\mu\epsilon$ or micro-strain. " ϵ " is the change in length divided by the original length and " μ " is equal to 10^6 thus micro-strain. By using GS average values measured on the butt logs, the trees were grouped into HI and LOW GS trees. Those trees with GS levels above the mean value were considered "high GS level", and below the mean value "low GS level." In order to avoid the effect of outliers the trimmed (Tr) mean¹ was used. The trimmed mean was (1301.9 $\mu\epsilon$; and the range was from 500 to 3420 $\mu\epsilon$). Tree samples from the three plots of yellow poplar that were harvested during August were processed using the balanced sawing technique by means of the (SDR) process and the CANT sawing

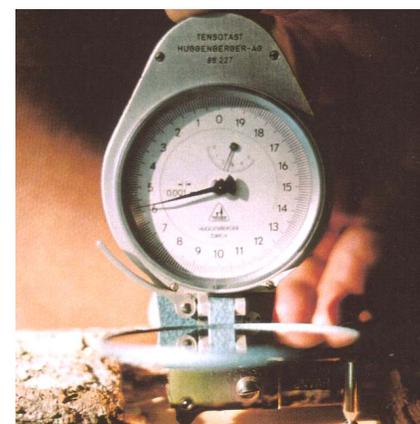


Figure 3.—Huggenberger tensotast (accurate to 0.001mm) used to measure growth stress. It is the movement of the two Phillips head screws (bottom) which is measured after the wood is cut free on each side. The instrument is provided with calibration bars.

¹Removes the smallest 5% and the largest 5% of the values (rounded to the nearest integer) and then averages the remaining values.

pattern. This technique is based on the alternate cuts of similarly located pieces from opposite sides of the log or flitch in order to counterbalance the longitudinal stress release. Figure 4 is a graphical presentation of these sawing patterns. In both sawing patterns, a portable sawmill with thin-kerf bandsaw blades was used. The bandsaw blades produced a 1/16 in. nominal saw kerf. The taper of the logs was split which permitted full thickness for the pieces obtained from the center of the log. For the SDR process, the opening face was at least 4 inches wide in order to produce a nominal 4 inch stud from the subsequent flitch.

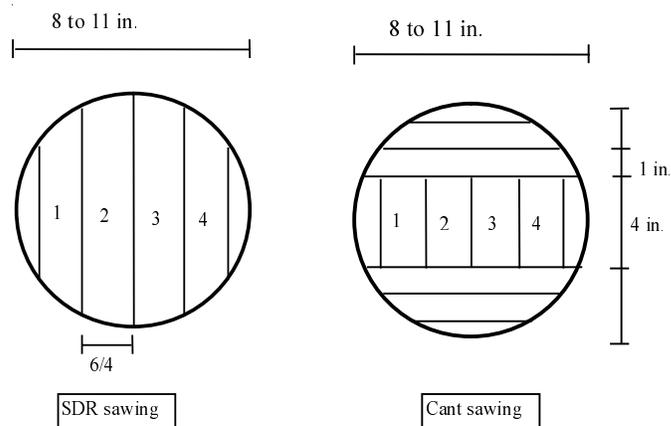


Figure 4.—Typical sawing patterns to produce 2 by 4 yellow poplar studs.

It was estimated that a bolt diameter inside bark (DIB) of 8 to 11 inches (Figure 4) would be required to apply the sawing patterns proposed in this study.

The diameter breast high (DBH) of the study trees ranged from 8 to 13 inches. As a result, some studs containing wane were produced. Only material from the butt and the second eight foot log was used.

Stud position in the tree was evaluated through log position or height (butt and second log) and radial (flat and quartersawn pieces) differentiation. From the sawing pattern in Figure 4, stud position 1 and 4 produced “flat grained” pieces while positions 2 and 3 generated “edge grained or quartersawn” pieces from the juvenile wood zone of the log.

Warping measurements in the green condition included crook and bow. These measurements were determined according to Hallock and Malcolm (1972), Milota (1995), and Serrano (1996). The number of end checks and the length of the longest end check were also determined. For the cant sawing process, these measurements were taken on 2 x 4's but for the SDR process, they were taken on flitches which still contained the waney edges.

Results And Discussion

Table 1 shows that the magnitude of GS increased significantly from May to September (p -value 0.0001). The GS for August was in an intermediate position. At the same time, the average GS readings on the west side were significantly higher than the east side (p -value 0.0263). Thus, harvesting and processing the trees before the dormant period will result in lower GS values and perhaps less warping and end checking in the lumber. This recommendation is further supported because fewer difficulties in debarking logs from May to August as compared to September were encountered. Okuyama et al. (1981) reported similar results in an experiment with the conifer tree sugi (*Cryptomeria japonica*), and Nicholson (1973) indicated the existence of some seasonal and other periodic variations of longitudinal GS. Archer (1986), based on theoretical premises, suggests that softwoods would start with slightly negative or nearly zero GS at the beginning of a growth season and then increase to some maximum value in the latewood near the end of the growing season. However, this GS seasonal change may be less pronounced in hardwoods, based upon the small microfibril angle variation over the annual ring. Kubler (1987) rejected the idea of GS seasonal change, arguing that without a new stimulus, GS should be the same in earlywood as well as in latewood. However, his position is not supported with experimental data. Okuyama et al. (1981) indicates that lignification is the most important process for growth stress generation. It is also known that the lignification process occurs over a period of time. Therefore, this time factor must be considered in the models (Yamamoto 1998).

The GS averages of the butt logs were significantly lower than the second logs (p -value 0.008, table 2). The interaction of plot and log position generated important effects. The significance level for this interaction effect was 7.10 % (p -value 0.071). The GS averages of the second logs tended to be higher than the butt logs, but in the case of plot number 2 the mean value was much higher. GS differences by polar points

Table 1.—Surface longitudinal growth strains (GS in $\mu\epsilon$) at DBH on standing trees for two polar points and three time periods.

Polar points	Time Periods		
	GS May	GS Aug.	GS Sep.
East	1099	1169	1743
West	1286	1639	1978
N	36	20	36

Note: -GS May and GS Sep. are side-matched observations on the same tree, while GS Aug. were taken on a sample of the 20 trees that were harvested and processed by the SDR and cant sawing method.

-GS Aug. means were adjusted by plot size.

Table 2.—Surface longitudinal growth strains ($\mu\epsilon$) on standing trees according to plot number and log position.

Log position	Plots			
	1	2	3	Average
Butt	1404	1333	1455	1397
Second	1481	1826	1541	1616
Average	1443	1580	1498	1507
N	80	40	40	160

Note: -Mean values by plot number include all 4 polar points for both the butt and the second log.

-GS values in this table were adjusted by plot size.

regardless of log position for the standing trees were only significant at 6.80 % (p -value 0.068, table 3). The mean values for the butt logs tended to be systematically lower than the second logs for all polar points. This large GS difference between the butt logs and the second logs from trees of plot number 2 was consistently reflected in higher levels of crook and end checking in the lumber specimens produced from this plot (table 2, figures 5 and 6). Log end split percentages were also significantly higher in the logs produced from trees of plot 2. Yao (1979) and Chafe (1981A and B) have reported similar results. However, other authors have not found a significant variation of GS with height (Sasaki et al. 1981A and B, Trenard and Gueneau 1975). Kubler (1987) stated that in a stable, immediate environment, peripheral strains are probably height-independent.

The plot number or the source of the trees affected the level of crook found in the green 2 x 4's and flitches (figure 5). The mean value of crook from plot 2 was significantly higher than from plots 1 and 3 (p -value 0.0315). The interaction of plot and GS level (HI and LOW) was significant (p -value 0.0406). Therefore, the level of crook found in the green material was dependent on the plot number and GS level. Processing trees with low levels of stress will reduce crook in green lumber. Since crook averages for that condition (Plot 2 and HI strain trees) include lumber pieces from the butt and the second logs, it is likely that the large difference of GS between these two log positions found in plot 2 explain such a high level of crook.

Crook in the green 2 by 4 studs was significantly higher than that in the flitches. In other words, the CANT sawing pattern generated higher levels of crook than the SDR process (p -value 0.0001) (fig. 7). On average, the amount of crook found in green 2 by 4 studs was around 3/32 in., but only 1/32 in. in the green flitches.

Just as with the crook results, the 2 by 4 studs had a higher level of green bow than the flitches (p -value 0.0102, fig. 7).

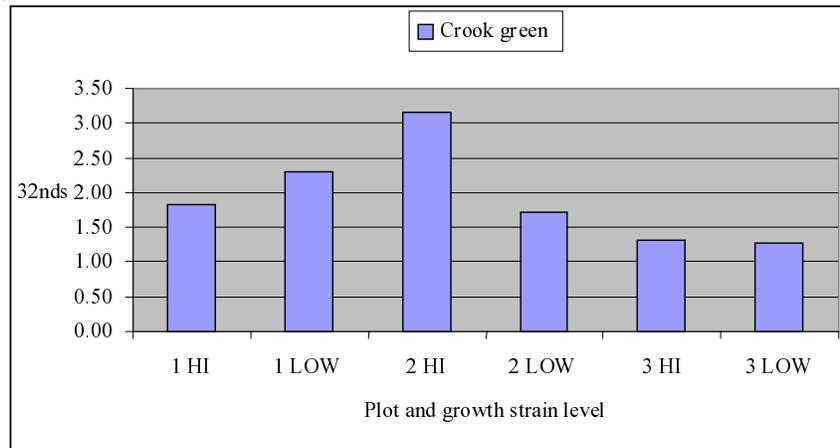


Figure 5.—Crook by plot number and growth strain level in the green condition.

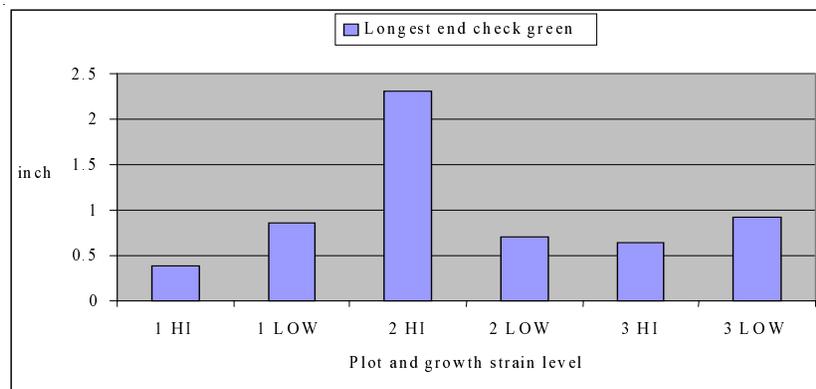


Figure 6.—Length of the longest end check by plot number and growth strain level in the green condition.

Bow in the green condition was affected by radical position. Higher levels of bow were found in the flat sawn pieces as compared to the quartersawn ones (p -value 0.0001). The interaction between radial position and plot number was also significant (p -value 0.0372). The flat sawn pieces from plot 2 exhibited the highest level of bow. Again, the large GS difference between the butt and the second logs from plot 2 may also explain such behavior. By subtracting the GS differences for the butt and second log (table 2) differences are 77, 493 and 86 $\mu\epsilon$ for plot 1, 2 and 3 respectively. The trees from all three plots of yellow poplar were grouped according to LOW and HI growth strain gradient. The trees classified as LOW had an average GS gradient between the butt and the second logs of 266 me as compared to 626 me for the trees classified as HI, and the average bow was 1/4 in. and 13/32 in. for LOW and HI respectively. An increase of about 135 % of the GS gradient from LOW to HI generated an increase of 63 % of bow in the green lumber specimens.

The longest end checks in the green state appeared only in the studs and fitches from quartersawn pieces. Another important factor was the significant interaction between plot and GS. Specimens from HI growth strain trees from plot 2 presented the highest level for the end check defect (fig. 6). This result also confirms the benefits of processing trees with low levels of stress in order to reduce the length of the longest end check in the green state, which is in complete agreement with the literature. Considering that the averages for this end check defect included pieces from both the butt and the second logs, it is conceivable that the large GS difference between these two log positions found in plot 2 (table 2) explain such a high level of end checks.

Table 3.—Surface longitudinal growth strains ($\mu\epsilon$) on standing trees according to the four polar points and two log positions.

Bolt	Polar points			
	west	south	east	north
Butt	1639	1313	1169	1469
Second	1719	1499	1668	1578
Average	1679	1406	1418	1524
N	20	20	20	20

Note: -GS values in this table were adjusted by plot size.

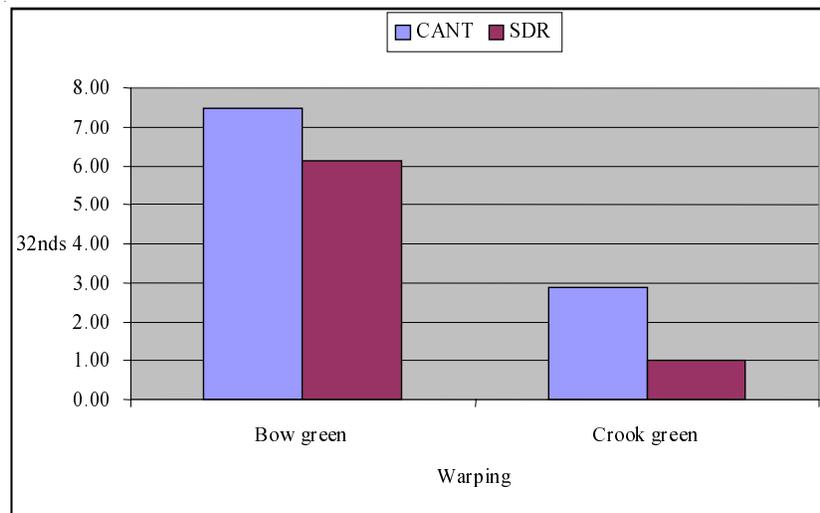


Figure 7.—Bow and crook level by sawing pattern in the green condition.

Conclusions

Longitudinal growth stress results in tension toward the outside of a tree stem or log and compression towards the center. These stresses are measured as growth strain. As the tree is processed first to logs and then to lumber, the stresses are unbalanced and log splitting, lumber warp, and checking occur. This study has demonstrated that yellow-poplar trees with higher GS produce green lumber with greater amounts of bow, crook and end checking and that these results are also affected by sawing pattern.

Crook found in the green specimens, as a result of growth stress release, was very important. Sixty percent of the crook in the 2 by 4 studs from the cant sawing pattern appeared right after the sawing operation as compared to 35 % for the unedged flitches from the SDR process. This also indicates a higher stress release as a result of the sawing operation in the green 2 by 4 studs than in the flitches.

From the merchantability point of view, quarter-sawing produces lower quality 2x4 studs than flat-sawn pieces. The important challenge is to overcome the poor performance of these pieces.

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ECONOMIC FEASIBILITY OF COMMERCIAL MAPLE SYRUP PRODUCTION IN ILLINOIS

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ABSTRACT.—For Illinois farmers, the maple resource is poised to be tapped given that 1.8 of the total 4.3 million acres of Illinois woodlands exists on farms. Industry standards suggest that a properly managed maple tree resource producing an average sap sugar concentration of 2 percent and an average volume per tap per season of 10 gallons of sap is necessary for a commercial maple syrup venture to succeed. IMPLAN results show that a 20-acre sugarbush, supporting 1,000 taps, will add an additional \$7,146 to the returns of an existing farm producing 50 acres of mulch corn and 50 acres of mulch soybeans. Such an operation would impact the local economy by requiring \$1,792 of locally purchased inputs and creating a total farm commodity value impact of \$12,564. Graphical results using ESRI's ArcView 3.3 illustrate that commercial maple syrup production has potential throughout several counties in Illinois, especially within the southern portions of the state. These results indicate that maple syrup production maybe an economically viable alternative agricultural enterprise for Illinois farmers and landowners to incorporate into their whole farm plan.

Introduction

Maple syrup, once referred to as “white gold”, is one of our oldest agricultural crops native to North America (Koelling et. al. 1996). As subsistence was a way of life for many of the early settlers, colonists, and farming families of the early 20th century, their life revolved around an agricultural operation whose primary role was to provide for the immediate needs of the family but also to produce an ample amount of surplus products to be bartered or sold for products that weren't produced on the farm. On farms where a maple resource was present, maple sap was collected and processed into syrup and sugar. It wasn't long until maple sugaring caught on to become an integral part of the spring farm experience, occurring at a time of year when other farm activities of necessity slowed down or ceased.

Historically, Illinois forests have been dominated by oak-hickory forests. Due to a lack of disturbances, such as the fires present prior to European settlement, fire tolerant tree species such as oaks and hickories are being replaced by fire intolerant species such as the sugar maple, also referred to as “maple takeover” (Fabry and Patterson 2000). Both silver and sugar maple (*Acer saccharinum* and *Acer saccharum*, respectively) can be used to produce syrup. Both are present within a wide natural range on upland and bottomland sites in eastern North America and are increasingly becoming more common in Illinois. Currently, maple syrup production is an alternative, value-added agricultural enterprise underdeveloped in the state of Illinois. Therefore, a potential exists for many Illinois farmers to incorporate maple syrup production into their whole farm plan, helping them to diversify their farm-income and establish a “safety-net” in years of crop-struggle.

Even though Illinois is not considered as a commercial maple syrup producing state, two commercial producers do exist within the state. Industry standards suggest that a properly managed maple tree resource producing an average sap sugar concentration of 2 percent and an average volume per tap per season of 10 gallons of sap is necessary for a commercial maple syrup venture to succeed (Vogt 1994, Heiligmann 2002). Funk's Grove Pure Maple Syrup (Shirley, Illinois) is the largest commercial syrup producer in the state while the Forest Glen Preserve (Part of the Vermillion County Conservation District) is the second largest commercial producer of maple syrup in the state. Currently, Funk's Grove puts approximately 6500 taps

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out and produces an average of 1600-1800 gallons of syrup per year (2000+ gallons on exceptional years). Forest Glen Preserve's production peaks at approximately one hundred gallons of maple syrup in a season. Consequently, Illinois has to import most of its syrup and candies since the local market demand is outpacing the local market supply.

This study uses IMPLAN (Impact Analysis for PLANning, Social Accounting and Impact Analysis Software, Minnesota IMPLAN Group, Inc., Stillwater, MN) to demonstrate the economic impacts that a hypothetical maple syrup industry would have on Illinois, both statewide and locally. Additionally, a geographic information system (GIS) is used in conjunction with the Forest Service Forest Inventory and Analysis database (FIA) to investigate the availability and production potential of the maple resource in the state.

Methods

Economic Impact Analysis

An adapted version of IMPLAN (Impact Analysis for PLANning, Social Accounting and Impact Analysis Software, Minnesota IMPLAN Group, Inc., Stillwater, MN) was used to model the economic impact that an expanded maple syrup industry would have on Illinois. A preliminary phase in this study used IMPLAN to construct an input-output model representative of the state of Illinois thus establishing a foundation upon which a hypothetical maple syrup industry could be evaluated. The IMPLAN-software is capable of assessing the potential economic impact on the region's economy as well as contributions to the local community. Results of the IMPLAN model can then be used to determine the economic feasibility of establishing maple plantings with or without cost-sharing such as the Conservation Reserve Program (CRP).

In order to more accurately reflect regional conditions, southern Illinois-specific enterprise budgets were constructed on a per acre basis for commodities such as mulch corn and mulch soybeans. Farm budgets were originally created by the University of Illinois extension office (UIUC Farmlab 1999) and then modified by researchers at Southern Illinois University Agribusiness Economics department (Peterson, 2003). The modified maple syrup enterprise budget used for this analysis was established from information obtained from the North American Maple Syrup Producers Manual (NAMSPM) (Koelling et al. 1996). This budget example assumes an operation consisting of 1,000 taps within a 20-acre sugarbush. Koelling et al. (1996) made several additional assumptions when they created the enterprise budget. For instance, a separate sugarhouse building is assumed to be located on the farm property, sap is collected using a vacuum tubing system, the evaporator is wood fired, and a reverse osmosis machine is used. The modified enterprise budget used for this study shares many of the same assumptions as the NAMSPM does, except that this study assumes the 20-acre sugarbush as well as the tractor and trailer are already in service, integrating a maple syrup enterprise into an already established "working" farm.

Once the enterprise budget was created, production function coefficients were calculated by taking the value of production and dividing through by total receipts (i.e., the gross revenues from retail, wholesale, and other products). This calculation created the gross absorption coefficients that are recognized by IMPLAN. In addition, value-added coefficients were established for employee compensation, proprietary income, other property income, and indirect business taxes. Final steps necessary to run the IMPLAN model included entering the aforementioned coefficients and assigning an employment level to the model.

After running the IMPLAN model, specific results (e.g., taken from the industry balance sheet) were aggregated (i.e., 526 "specific" sectors grouped into 21 "generalized" sectors) and selected output data were taken from impact tables including output, employment, personal income, total value added, employee compensation, proprietors income, other property income, and indirect business taxes to create the following summary tables. These tables include: Table 1. Economic Impact in Local Community of a Farm Producing Maple Syrup, Table 2. Economic Return to Farm of a Farm Producing Maple Syrup, Table 3. Total Economic Impact in Local Community of a Farm Producing Maple Syrup, and Table 4. Multiplier Effect on the Local Community of a Farm Producing Maple Syrup. These individual tables were then linked to another spreadsheet capable of demonstrating the economic impacts a 20-acre maple syrup enterprise would have on an established 100-acre Illinois farm already producing mulch corn and mulch soybeans. For ease of comparison, summary information from these four tables was compiled in Table 5.

Table 1.—Economic Impact in Local Community of a Farm Producing Maple Syrup

	Farm Level Commodity Production (dollars)	Gross Inputs (dollars)	Inputs Purchased Locally (dollars) (State)
1—20 acre sugarbush	9500.00	2,354	1,792

Table 2.—Economic Return to Farm of a Farm Producing Maple Syrup

	Farm Level Commodity Production (dollars)	Return to Farm* (dollars)	Percent of Commodity Production that is Return to Farm
1—20 acre sugarbush	9500.00	7,146	75%

*Return to Farm includes Employee Compensation, Proprietary Income, Other Property Income, and Taxes

Table 3.—Total Economic Impact* in Local Community of a Farm Producing Maple Syrup

	Total Farm Commodity Impact* (dollars)	Total Farm Commodity Employment Impact*	Total Farm Commodity Value Added Impact* (dollars)
1—20 acre sugarbush	18,475	0.42	12,564

*Total Economic Impact = Direct (Farm Level) Impact+ Indirect Impact + Induced Impact

Table 4.—Multiplier Effect on the Local Community of a Farm Producing Maple Syrup

	Total Farm Commodity Multiplier*	Total Farm Commodity Employment Multiplier*	Total Farm Commodity Value Added Multiplier*
1—20 acre sugarbush	1.94	1.32	1.76

*Multiplier = (Direct (Farm Level) Impact+ Indirect Impact + Induced Impact)/Direct Impact

Table 5.—Economic Impacts of Maple Syrup on Representative 100 Acre Farm

Economic Impacts	Without Maple	With Maple
Farm Level Commodity Production (dollars)	31,495	40,995
Gross Inputs (dollars)	16,093	18,447
Inputs Purchased Locally (dollars) (State)	7,920	9,712
Return to Farm ¹ (dollars)	15,402	22,548
Marginal Return to Farm (dollars)		7,146 (46%)
Percent of Commodity Production Returned to Farm	49%	55%
Total Farm Commodity Impact ² (dollars)	57,563	76,038

¹Return to Farm includes Employee Compensation, Proprietary Income, Other Property Income, and Taxes

²Total Economic Impact = Direct (Farm Level) Impact+ Indirect Impact + Induced Impact

Mapping Industry Potential

County level data for Illinois, obtained from the U.S. Forest Service Forest Inventory and Analysis (FIA) database, was used to map potential syrup production by county. FIA is a continuing endeavor mandated by Congress in the Forest and Rangeland Renewable Resources Planning Act of 1974 and the McSweeney-McNary Forest Research Act of 1928. The FIA database provided this study with extensive data on forest area attributes (e.g., size class and frequency) and on the status of live and standing dead hard and soft maple trees by county in Illinois. While the FIA database is not an inventory of every tree, a continuous sampling of multiple plots is used. For the purposes of this analysis, data obtained for the multiple plots was statistically aggregated up to the county level. While a majority of the counties in Illinois used 1998 (cycle 4) data, Edgar County used 2001 (cycle 5) data and the counties of Massac, Cass, Bureau, Scott, and Kendall, used 1985 data (cycle 3). The counties of Boone, Carroll, Champaign, Cook, De Witt, Douglas, Du Page, Ford, Kane, Kankakee, Lake, Mc Donough, McLean, Menard, Morgan, Piatt, Stephenson, and Will couldn't be included in the analysis because no data was available. The DBH tapping criterion outlined in the NAMSPM was used to estimate the number of taps each tree could support. For example, a DBH of 11-14.9 inches is capable of supporting one tap, 15-18.9 inches is capable of supporting two taps, and 21-inches and greater capable of supporting 3 taps. Assuming that each tap produces 1 quart of syrup per year, the total number of taps was divided by 4 to calculate the total gallons. This calculation represents the number of gallons of syrup that each county in Illinois could potentially produce, assuming that every tree was tapped and every tap had a potential of producing an average of 10 gallons of sap per tap per year at 2 percent sugar. This table was then inserted into ESRI's ArcView 3.3 and a map was generated to predict the potential gallons of syrup per county that could be collected in Illinois.

Results

Economic Impact Analysis

The results in Table 5 compare the economic impacts of a farm producing 50 acres of mulch corn and 50 acres of mulch soybeans with and without 20-acres of sugarbush. A farm producing 50 acres of mulch soybeans and 50 acres of mulch corn (without a maple syrup enterprise) returns \$15,402 to the farm. In other words, 49 percent of the money generated from producing these commodities is returned to the farm. Money returned to the farm includes employee compensation, proprietary income, other property income, as well as indirect business taxes. Incorporating a 20-acre sugarbush adds an additional \$7,146 to the returns to farm bringing the total to \$22,548, an increase of 46 percent. In addition, maple syrup increases the percent of the commodity production that is returned to farm to 55 percent, an overall increase of 6 percent. The maple syrup also added an additional \$18,475 to the total farm commodity impact bringing the total to \$76,038. The total farm commodity impact includes a direct (farm level) effect, an indirect effect, and an induced effect. These results demonstrate the positive economic potential of incorporating a maple syrup enterprise into a whole farm plan. Future research will assess the economic contributions that a maple syrup enterprise will have on different farm diversification scenarios.

Mapping Industry Potential

Figure 1 illustrates the potential unconstrained maple syrup production per county, assuming that every tree is tapped. The darkest areas in the figure represent counties with the greatest number of maple trees. Mapping data from the FIA database demonstrates that the resource potential for commercial maple syrup production is predominately concentrated in southern and western Illinois but is also concentrated in the central portions of the state. Potential syrup production under the unconstrained scenario could reach as high as 344,400 gallons in some counties.

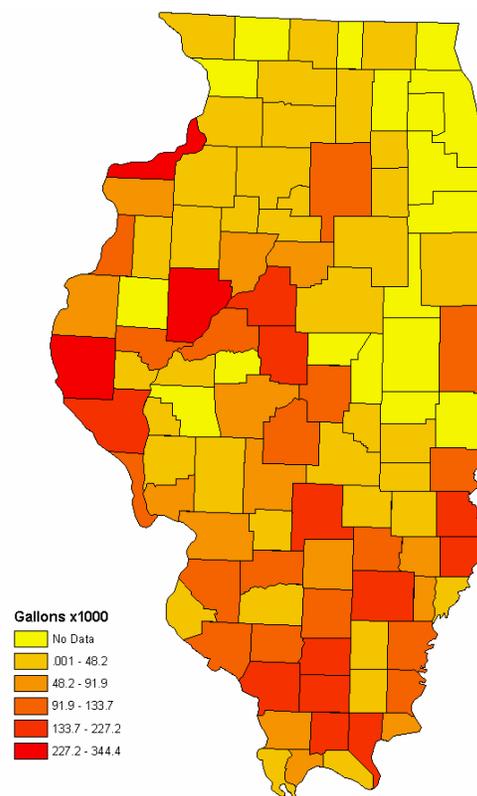


Figure 1.—Unconstrained Maple Syrup Production Potential

Discussion And Conclusions

Economic Impact Analysis

Because small to medium sized farms are more predominant in the southern part of Illinois, this analysis used a representative farm size of 100 acres. A 20-acre maple grove was incorporated into a representative farm already producing 50 acres of mulch corn and 50 acres of mulch soybeans for a total of 120 acres, assuming the farm had a maple resource poised to be tapped. While there are integrated and diversified maple syrup operations that exist with the potential to provide enough income to support a family solely from receipts obtained from maple products alone, most maple syrup produced on farms provides a supplemental source of income to the other commodities being produced. The appeal of producing maple syrup is that it is part of the seasonal farm experience, occurring at a time of year when other farm activities of necessity have slowed down or ceased. Additionally, farmers who want to enter the maple syrup industry are often advised to consider the maple syrup-product business as an addition to other agricultural enterprises by allowing for expanded use of some of the available resources (e.g., buildings, tractor, trailer, labor).

With regard to large capital expenditures common in new or expanding commercial operations, several planning options need to be considered. When planning a new sugarhouse, for example, the degree of public access plays a major role in the determination of a specific location. If the intent is to use the sugarhouse as a marketing location, it should be accessible by a hard surface road with adequate parking, and should have electrical services, sanitary facilities with a potable water supply, adequate product display areas, and sufficient room for visitors. The building and location should also be prominent, professional, and attractive to the greatest extent possible. In Illinois, compliance with local governmental regulations (i.e., Illinois Department of Public Health—IDPH) relating to land use and public accommodations may be necessary. If on-site dining facilities are in the future design, it is imperative to be in compliance with local and state regulations regarding food service areas.

Producing maple syrup is an alternative agricultural enterprise that utilizes a farm's natural resource base (both existing and potential) and employs best management practices (BMPs) to help improve the quality of farm life, diversify farm income, and promote rural sustainability. What distinguishes maple syrup production from other crops is the relative permanence of the trees that yield the crop. In other words, instead of harvesting maple trees for wood products and generating an income right away, maple syrup allows income to be generated over the life of the tree. Although maple syrup enterprises tend not to be the main source of income for many producing households, they are still economically important to the producers. With maple syrup production integrated into the full array of farming activities, many producers emphasize how the enterprise helps manage risks by diversifying farm income in an uncertain economy and exercising the seasonality of rural resource use and employment. For some in rural communities, resource-based self-employment complements available non-farm employment options.

Acknowledgments

The authors would like to acknowledge the C2000 Sustainable Agriculture Program of the Illinois Department of Agriculture for their support in this project. The authors also acknowledge the assistance of two anonymous reviewers whose suggestions greatly improved the manuscript.

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RIPARIAN SILVER MAPLE AND UPLAND SUGAR MAPLE TREES SAP SUGAR PARAMETERS IN SOUTHERN ILLINOIS

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ABSTRACT.—Forty two upland sugar maple trees and 59 riparian silver maples were tapped in 2003 to characterize their sap sugar parameters within the southern Illinois region. The mean sap sugar concentration (SSC) among all sugar maple trees was 2.03 percent (1.53 to 3.18 percent range) and the mean sap volume was 133.7 liters per tree or 44.6 liters per tap. Tree basal area was significantly related to total sap volume but not SSC in sugar maples. Sugar maple trees receiving additional taps than recommended guidelines had significantly reduced mean volume per tap but similar total sap volume compared to guideline-tapped trees. Silver maple mean SSC for the season was 1.71 percent (1.10 to 2.53 percent range) and mean total sap volume was 32.6 liters or 23.5 liters per tap. Silver maple total sap volume and sap volume per tap was dependent upon number of taps per tree. Total sap volume and SSC was positively related to basal area of silver maples.

Introduction

A sustainable maple forest resource provides several benefits to the environment and landowners. These benefits include wildlife habitat, timber, carbon sequestration, riparian zone filters, wind and water breaks, and other products. Maple syrup production is an important sustainable forest-based industry within the United States, especially in the northeast. In 2002, the United States production of maple syrup was 5.3 million liters and valued at \$38.4 million. The primary maple syrup producers are located in Vermont, Maine, and New York. In 2001, Vermont alone had a maple syrup production value estimated at \$8.4 million (Davis 2003). Of the 13 native North American maple species, Sugar maple (*Acer saccharum* Marsh.) and black maple (*Acer nigrum* Michx. f.) are preferred for commercial tapping because of their relatively high sap sugar concentration (SSC) (averaging between 2.0 and 2.5 percent) and long sap collection season (Heiligmann and Winch 1996).

Production of maple syrup is most cost effective if trees produce large volumes of sap with high sugar concentration. This is in part because of the effort, energy, and resources needed to tap trees, collect sap, and produce syrup from sap. While it takes 87.2 liters of sap with a sugar concentration of 1 percent to make one liter of maple syrup, only half the sap is required if SSC is 2% (Willits and Hills 1976). Typically, the sap sugar concentration for sugar maples can range anywhere from 2 to 6 percent, with a tree producing anywhere from 4 to 8 liters of sap per day during peak days (Tyree 1983).

Silver maple (*Acer saccharinum* L.) trees have often been disregarded for tapping because of their shorter collection season due to early budbreak, lower average SSC of 1.5 to 2.0 percent, and higher levels of sugar sand production when processed (Walters 1982, Heiligmann and Winch 1996). However, studies (Larson and Jaciw 1967, Zaczek et al. 2003) have shown considerable variability in sap sugar parameters of individual silver maples, suggesting that progress could be made with tree improvement efforts. Additionally, silver maples grow on a wide range of sites, are flood tolerant, and grow quickly (Hardin et al. 2001). These characteristics may allow maple syrup production on traditionally underutilized bottomland sites, including riparian buffers.

Much of southern and western Illinois is considered outside of the range of commercial maple syrup production (Heiligmann et al. 1996). This may be due in part to an unsuitable climate, a scarcity of maple trees, or poor sap characteristics. Climatic conditions that induce sap flow occur when temperatures oscillate above and below the freezing point anytime during the tree's dormant season (Heiligmann et al. 1996). Although climatic conditions may not be as favorable in this region when compared to others,

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southern Illinois weather conditions include these oscillating periods of freezing and thawing albeit, occurring earlier in the year than areas farther north.

Even though the oak hickory forest type is most common in Illinois there is a considerable existing and currently increasing component of maple in the region (Bretthauer and Edgington 2002, Bretthauer and Edgington 2003). It was reported that the amount of commercial oak-hickory forests dropped 14 percent while maple-beech forests increased by 4,219 percent (10,290 ha to 434,200 ha) from 1962 to 1985, reflecting land use changes and successional processes (Iverson et al. 1989, Fralish 1997, Bretthauer and Edgington 2002, Bretthauer and Edgington 2003).

There is a paucity of information about the sap parameter performance of maple trees in the region. Recent published information (Keeley 2000, Zaczek et al. 2003) has primarily focused on the relative differences among young and small diameter clones from wide-ranging provenances of silver maple that are being grown in southern Illinois. Information on mature larger-sized endemic maple trees is essentially lacking for the region. This study was conducted to provide information and comparisons on the sap performance parameters (SSC and sap volume) of native sugar maple and silver maple trees growing on representative field sites in southern Illinois.

Methods

Study Area

The sugar maple trees used for this study were located on the Buchheit farm near Lick Creek, IL, in Union County, approximately 48 kilometers southeast of Carbondale. The mature forested upland site had variable aspect and rolling topography and was of the oak hickory type with associated sugar maple in the overstory and midstory. Study trees in dominant or co-dominant crown classes were located along a portion of an access trail selected as a fixed population of sugar maple that had been tapped in previous years by the landowner. The site was used as part of a low intensity rotational cattle pasture. Silver maple trees were located along a relatively narrow riparian zone adjacent to open agriculture row crops and grassland within 10 meters of Little Crab Orchard Creek, bounded by Chautauqua and Rowden Roads on Southern Illinois University property near Carbondale, IL, in Jackson County. The site was composed primarily of early successional tree species trees with tapped silver maples in the dominant or co-dominant crown classes.

Field Procedures

Tapping procedures and equipment were similar for both sugar maple and silver maple. Starting in mid-January 2003, 42 sugar maples and 59 silver maples were tapped using a cordless power drill with a 7.9 mm in diameter drill bit. Holes were drilled 3.8 cm deep with a slight upward slope. This is the recommended depth for 7.9mm diameter spiles (personal communication from Timothy Perkins on February 3, 2004 based on research at the Proctor Maple Research Station, University of Vermont). Tree Saver food grade nylon spiles, (Sugar Bush Supplies Co. Mason, MI) were then hammered into the holes using a rubber mallet. These spiles are smaller in diameter at 7.9 mm and purportedly allow the tree to heal-over tapholes more quickly than more traditionally-sized spiles with diameters of 11.1 mm (Perkins 1999). Polyethylene tubing was run from the spile end to 19 liter plastic food grade buckets to collect sap.

The number of taps for each sugar maple tree (ranging from 2-6) was established by the landowner. This was done in previous years by estimating tree size and increasing the number of taps as tree size increased. These estimates did not strictly follow traditional tapping guidelines for the number of taps per tree of a given diameter at breast height (DBH) as recommended in the North American Maple Sugar Producers Manual (NAMSPM) (Heiligmann et al. 1996). Tapping silver maple trees also did not strictly follow NAMSPM tapping guidelines and included some trees slightly smaller in DBH than recommended. Since an unknown response of silver maple to tapping guidelines using smaller diameter taps exists, a comparative analysis of the sap parameters was performed for trees categorized as; guideline-tapped (guideline recommended number of tapholes per tree DBH); undertapped (fewer tapholes per tree DBH); and overtapped (greater numbers of tapholes per tree DBH).

The data collection period for SSC analysis and sap volume of the sugar maple trees was from January 29 to March 2, 2003. There were six sap collections for SSC analysis and there were eight volume measurements. For silver maple, there were four collection periods for SSC analysis and four volume measurements, starting January 31 and ending on March 5.

Diameter at breast height in centimeters was measured on July 22, 2003. Basal area (cm²) was determined for each tree by calculating the cross-sectional area for each stem forking below breast height ($\pi * (1/2 \text{ DBH})^2$) and summing the total for each stem of the tree.

Laboratory Procedures

To determine SSC, samples were collected from each tree directly from the tubing with 1.5ml polypropylene microcentrifuge tubes (Fisher Scientific Pittsburg, PA) while sap was actively flowing. The sap samples were brought back to the lab, refrigerated (1.1°C to 4.4°C), and analyzed within 48 hours. SSC was measured using a temperature compensated refractometer (Fisher Scientific Pittsburgh, PA). The refractometer was initially calibrated and periodically recalibrated using distilled water. Sap sugar concentration was determined by placing a drop of sap from sample tubes onto the lens of the refractometer, exposing it to a fluorescent light, and reading the scale to the nearest tenth of a percent. The lens was cleaned between each sample. Sap volume (ml) was determined by measuring the depth of the sap in the bucket in the field to the nearest 0.1 cm using a ruler. The bucket depth to volume conversion was determined by calibrating the buckets with known volumes of water and measuring depth, producing a linear regression equation with a $R^2=0.99$ ($p<0.0001$).

Analysis Method

Mean DBH, basal area (BA), SSC, total sap volume, and sap volume per tap were determined for each species to provide baseline information on sap parameter performance and general comparisons between the two species. Variance analysis using JMPIN software (SAS institute, Cary, North Carolina) at $\alpha=0.05$ compared stem basal areas of each tree between species to determine if the sample populations of each species differed in stem size. Comparing the two species for sap parameters was not warranted. This is because the species did not co-occur on each site and sites were considerably different physically and developmentally potentially confounding species to species comparisons. Analysis of variance was used to determine if differences existed among silver maple and among sugar maple trees using individual tree SSC for each collection period. Analysis of total sap volume and sap volume per tap was made for each species by tapping categories (undertapped, overtapped, and guideline-tapped). Since there were only two sugar maple trees that were considered undertapped according to NAMSPM guidelines, this category was omitted when analyzing sugar maple by tapping guidelines. For each species, linear regression was used to determine if significant relationships existed between SSC and BA, SSC and sap volume, and BA and sap volume. Basal area was used in regression analyses to sap parameters instead of DBH because previous studies have shown stronger relationships (Keeley 2000, Zaczek et al. 2003).

Results

The mean DBH of sugar maple trees was 56.6 cm with a range from 26.2 to 122.3 cm. Mean basal area of sugar maple trees was 1823.8 cm² with a range from 539.1 to 5229.6 cm². Silver maple mean DBH was 43.3 cm (range 22.4 to 101.1 cm) and mean BA was 1622.2 cm² (range 394.1 to 8027.7 cm²). Basal area of the trees was similar for both species ($p=0.4961$).

Sugar maple mean SSC was 2.03 percent ranging from 1.53 to 3.18 percent for individual trees. The season high SSC for a single tree during an individual collection date was 3.70 percent. Among sugar maple trees there was no significant relationship between SSC and basal area ($p=0.5243$).

Silver maple mean SSC was 1.71 percent ranging from a season-wide 1.10 to 2.53 percent for individual trees. The highest SSC among silver maple trees for a single collection date was 2.70 percent. Among silver maples there was a significant but weak relationship between a tree's basal area and SSC ($R^2=0.07$, $p=0.0386$).

Table 1.—Sap sugar concentration (SSC) characteristics of upland sugar maple and riparian silver maple trees growing in southern Illinois.

Species	Number of trees	Mean SSC (%)	Range SSC (%)	% Trees with a SSC at or above 2.00
Silver Maple	59	1.71	1.10-2.53	27%
Sugar Maple	42	2.03	2.53-3.18	60%

Sap volume produced from the population of 42 sugar maple trees totaled 5614.4 liters for the season (a mean of 133.7 liters per tree or 44.6 liters per tap). One individual tree produced 338.6 liters of sap for the season. There was a significant positive linear relationship between total sap volume of sugar maples and their basal area ($R^2=0.69$, $p<0.0001$) but this may be confounded by larger trees having more taps in general. In order to eliminate this, an analysis for trees with only 2 taps ($n=27$) determined that the total sap volume of a tree was still positively related to its basal area ($R^2=0.28$, $p=0.0049$). Accounting for variable number of taps by determining a tree's sap volume per tap, regression analysis also determined that there was a significant but weak positive relationship of sap volume per tap to basal area ($R^2=0.15$, $p=0.0101$). No relationship existed between a sugar maple's SSC and total sap volume for the season ($p=0.4222$).

The 59 silver maple trees produced 1925.3 liters of sap over the season (mean per tree 32.6 liters, and a mean of 23.5 liters per tap) with the highest total volume produced by a tree being 141.9 liters of sap. There was a significant positive linear relationship between total sap volume per tree and BA ($R^2=0.30$, $p<0.0001$). This may have been related to the number of taps per tree generally increasing with increasing tree size. Considering this, the sap volume per tap and BA were not significantly related ($R^2=0.07$, $p<0.0584$). However, a significant positive relationship existed between SSC and total sap volume ($R^2=0.29$, $p<0.0001$) and between SSC and volume per tap ($R^2=0.16$, $p<0.0015$).

Total sap volume examined according to tapping category per tree (Table 2) showed no significant difference among guideline-tapped and overtapped sugar maples ($t=0.6680$). Considering sugar maple sap volume per tap with regard to tapping categories, guideline-tapped trees tended to produce more sap per tap (48.8 liters) than overtapped trees (38.9 liters) ($R^2=0.12$, $t=0.0265$).

Silver maple total sap volume (Table 2) was significantly greater for overtapped and undertapped trees compared to guideline-tapped trees ($R^2=0.16$, $p=0.0089$). Among silver maples, sap volume per tap was greater for undertapped trees compared to guideline-tapped or overtapped trees ($R^2=0.15$, $p=0.0098$).

Discussion

Within each maple species there was considerable variation in sap volume production as well as SSC. Sap sugar concentration is a very important factor in production efficiency and cost. Sugar maple SSC (mean 2.03 percent) in our study fell into the range 2.0 to 2.5 percent expected in other regions and may vary due to genetics, site and environmental conditions such as weather (Heiligmann and Winch 1996). Silver maple SSC was 1.71 percent which is in the range expected for the species in other regions (Walters 1982, Heiligmann and Winch 1996). However, a previous study of woodlot-grown silver maples found that mean SSC was higher at 2.6 percent and ranged from 1.7 to 5.1 percent (Larsson and Jaciw 1967). In the current study, 27 percent of the 59 silver maple trees tapped, had a SSC at or above 2.0 percent whereas 60 percent of the sugar maples had SSC at or above 2.0 (table 1). Sap sugar concentrations of individual sugar maple trees have been found to be relatively consistent from year-to-year (Taylor 1956, Kriebel 1960). A two year examination of silver maples also showed a year-to-year consistency (Keeley 2000). Therefore, trees with lower SSC may be culled out, leaving only those trees that have a higher SSC.

Sugar maple SSC was not related to total sap volume. A study done by Blum (1971) also found no significant relationship between SSC and total sap volume among three sugar maple stands. Additionally no relationship was found between SSC and basal area, indicating that larger sugar maple trees are not necessarily sweeter trees. However, when considering all sugar maples, higher DBH trees produce more

Table 2.—Comparisons of basal area (BA), total volume, total volume per tap, and sap sugar concentration (SSC) for sugar maple and silver maple trees receiving the recommended number of taps (guideline-tapped), more taps (overtapped) or fewer taps (undertapped) than guidelines published in the North American Maple Syrup Producers Manual (Heiligmann and Winch 1996).

Silver Maple					
Tapping categories	Number of trees	Mean BA (cm ²)	Mean total sap volume (L)	Mean sap volume/tap (L)	SSC (%)
Guideline	33	1250.4	22.6 (b)	18.7 (b)	1.60 (b)
Overtapped	7	1531.4	54.9 (a)	16.7 (b)	2.03 (a)
Undertapped	19	2301.5	41.8 (a)	30.9 (a)	1.81 (a)
		NS	*	*	*
Sugar Maple					
Tapping categories	Number of trees	Mean BA (cm ²)	Mean total sap volume (L)	Mean sap volume/tap (L)	SSC (%)
Guideline	13	1897.0	125.2	48.8 (a)	2.12
Overtapped	27	1766.8	138.1	38.9 (b)	1.99
		NS	NS	*	NS

Note: Means by species in a column followed by the same letter are not significantly different alpha=0.05 using LSD mean separation procedures

NS = Not Significantly Different

* = P<0.05 (Silver maple, F-test with 2 degrees of freedom; Sugar maple, t-test with 1 degree of freedom)

total sap volume although this may be a result of larger trees tending to have more tapholes or having larger root systems. There was a significant relationship between tree basal area and total sap volume of sugar maples. Accounting for this and examining the 27 individual sugar maple trees with just two taps, it was still found that as the basal area increased so did the total sap volume. Sugar maple trees totaled 5614.4 liters of sap for the season or a mean of 133.7 liters per tree or 44.6 liters per tap. In other studies sugar maple total sap volume was found to be relatively consistent from year to year (Marvin et al. 1967), suggesting that this can be used to select for tree with desirable sap volume production.

Contrary to sugar maple, silver maple SSC was positively related to both total sap volume and basal area. This indicates that larger silver maple trees tend to be sweeter and also produce more sap volume. Silver maple is one of the fastest growing hardwoods and the fastest growing of the North American maple species (Dirr 1975, Heiligmann and Winch 1996). Since it may be possible to select for sweeter silver maples by selecting larger silver maples, their fast growth rate would be a positive species characteristic to consider. Silver maples have been found to grow from 1.3cm to approximately 2.5cm in DBH each year (Gabriel 1990); this would mean that they would reach a large enough size for tapping between 15 to 20 years compared to the 40 to 60 years for sugar maples to reach a size appropriate for tapping (Heiligmann and Winch 1996). Silver maples, unlike sugar maples, can be readily clonally propagated (Preece et al. 1991). A study done by Kriebel in 1989, indicated that SSC among sugar maple grafted clones was relatively consistent over the years. Therefore there may be potential to select for silver maple trees with desirable sap characteristics, and receive similar results from their propagated clones when grown in comparable environments.

Environmental influences and species differences, probably affected sap volume yields in this study. In addition, a tapping depth of 3.8cm and the use of relatively smaller 7.9 mm diameter spiles may have also influenced sap volume yield. Tapping depth has been shown to be related to sap volume yield, with deeper tapholes resulting in higher volume yields (Cope 1949, Perkins 1999). While past research has shown taphole diameter did not affect sap yields (Robbins 1965). A more recent study found that using smaller diameter spiles sap volume yield was 80-100% the sap volume yield of standard-sized spiles (Perkins 1999). The potential effect on the sap volume yield for this study from reduced tapping depth and smaller diameter spiles, would be a reduced sap volume yield.

Examination of NAMSPM tapping guidelines was done to understand more clearly whether guidelines established for maple tree tapping in the northeastern U.S. are appropriate for southern Illinois using silver and sugar maple tree species. In earlier research, volume of sap collected per taphole has been found to be independent of the number of tapholes per tree (Willits 1967). However, in the current study, both silver and sugar maple sap volume per tap was dependent on the number of taps per tree producing greater sap volume per tap on trees with fewer taps. This is in accordance with Heiligmann et al. (1996) who reported that using fewer tapholes can substantially increase the yield of sap per taphole. In sugar maple, overtapping trees did not produce greater total volumes than guideline tapped trees as a result of reduced sap volume per tap. In addition, overtapping trees is a concern because wounding from tapholes can influence tree health more than harvesting too much sap (Heiligmann et al. 1996).

Undertapped silver maple trees produced more volume per tap than both guideline-tapped and overtapped trees. Overtapped and undertapped trees had a significantly higher SSC and total sap volume than guideline-tapped trees. This agrees with silver maple SSC being positively related to total sap volume, (as shown earlier) regardless of guidelines. This may suggest that the traditional guidelines recommended in the NAMSPM (1 tap for 25-38cm, 2 taps for 38-51cm, 3 taps for 51-64cm, and 4 taps for tree bigger than 64cm) should be reconsidered when tapping. More research is needed to determine if tapping guidelines are appropriate for silver maple.

Conclusion

This study has characterized maple sap sugar parameters of sugar and silver trees in southern Illinois. Both species had a mean SSC that was within the expected range reported in other regions. With past research showing SSC and total sap volume of sugar maple to be relatively consistent over time, there is potential to select for just those trees with desirable SSC and total sap volume production (Taylor 1956, Kriebel 1960, Marvin et al. 1967). Silver maple SSC was positively related to total sap volume and BA. Silver maple SSC has been shown to be consistent from year-to-year (Keeley 2000). Selecting silver maple trees with desirable SSC, may include relatively high total sap volume production. The number of tapholes per tree was shown to influence the amount of sap volume per tap of both species. Each species' sap sugar parameters showed a different response to tapping guidelines. Sugar maple guideline-tapped trees produced as much total sap volume as overtapped trees with similar SSC. Silver maple had a different response with both undertapped and overtapped trees producing more total sap volume with higher SSC. This was unexpected, and reflects the potential need for further research on tapping guidelines.

Acknowledgments

The authors would like to acknowledge the support of the C2000 Sustainable Agriculture Program of the Illinois Department of Agriculture, and cooperator Buchheit Farms.

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FACTORS DETERMINING THE SUITABILITY OF TREES AND LOGS FOR THE FACE VENEER INDUSTRY

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ABSTRACT.—Trees and logs suitable for use as fine face veneer command the absolute highest prices of any fiber products entering the commercial hardwood market. Many factors determine whether a tree or log will be suitable for face veneer and its value. These factors include species, size, geographic and site location, growth rate, and most importantly, uniformity of growth rate, color, concentricity of growth rings and a centered pith. Other factors include absence of mineral; absence of pin knots, especially in oaks and walnut; absence of glass worms in ash; absence of sugar streaks in hard maple; absence of gum spots in cherry; and absence of bird peck in several species. This paper will review these factors and explain their significance to foresters and other land managers.

Hardwood veneer logs command a premium price in comparison to sawlogs and other fiber products of the forest. As an example, Hoover and Gann (2002) report that the prices paid for the highest quality delivered white oak sawlogs in Indiana averaged \$634 per thousand board feet, Doyle Scale in 2002. In the same report, several diameter categories and two quality classifications are used to report delivered veneer log prices. For the lower quality classification and the smallest DIB class (13-14 inches) the price for white oak veneer logs was \$770 per MBF but increasing to nearly \$2940 per MBF for 28 inch DIB logs of the highest quality. Thus, it makes good economic sense to market high quality logs of the appropriate species to the face veneer industry.

In addition to the increased economic value, the use of hardwood timber for veneer has other positive attributes. First processing a log into veneer ranging from 1/32 to 1/50 inch in thickness greatly extends the resource in comparison to cutting standard 4/4 lumber which is somewhat over 1 inch thick. Veneering also allows for the production of matched grain patterns, inlays, and other artistic designs. Also, veneer can now be wrapped around profiles made of reconstituted wood thus reducing the need for long, thick clear moulding blanks. It can also be formed over machined panels of reconstituted products such as for raised panels in cabinet doors.

Veneering is a fascinating industry and successful marketing of veneer quality logs or trees is dependent upon the seller knowing what constitutes veneer quality logs and an understanding of the marketing process. Unfortunately, most landowners and many individuals who market timber are not aware of the basic requirements for trees or logs to be considered “face veneer quality”. Buyers will likely not spend much time visiting tree or log offerings if they detect the seller does not understand the quality levels needed and the material is over represented due to the sellers lack of knowledge. This paper will provide insight into what constitutes veneer log quality.

Methods

The hardwood face veneer industry is composed of only 30 flat slicing operations in eastern North America. Within this industry, there are no scientific studies and very few current publications which have attempted to define log and veneer quality from an appearance aspect. These companies purchase the quality of logs as dictated by their customers, cut them into flitches (log halves or quarters) and slice them into fancy face veneers (figs. 1, 2A, B). Rotary cut veneer or peeling a log around the circumference constitutes a different industry.



Figure 1.—Veneer slicing operation showing the flitch or log half (back and right) being moved up and down against a stationary knife and the individual veneer sheets being stacked in the order that they are sliced.

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Figures 2A, B.—During processing for the domestic market, three sample sheets are taken from each flitch. These sample sheets are shown to potential buyers. Considering B, the sheet on the left is from the outside of this walnut log and the one on the right is from near the heart. Note the absence of defects, excellent color and uniformity, and cathedral patterns in the first two sheets on the left of this high quality flitch. A much lower quality walnut flitch (B) with an irregular color, knots, and excessive sapwood is shown on the right. Each flitch is bar coded as to origin, log buyer, square footage, grade and value.

However, the Department of Forestry and Natural Resources at Purdue University has held numerous workshops on the subject and developed some written materials on log quality (Cassens 1992, Cassens et al. 2001). In these workshops the head log buyer for different companies will show, by using logs, what characteristics are important and the range in quality which can be accepted. Veneer sample sheets are used to demonstrate how the various log characteristics affect the appearance and value of the resulting veneer. Quality standards and species desirability vary between companies and change over time.

Basic Requirements For All Species

There are certain basic requirements that any tree or log intended for the face veneer industry must meet. In addition, there are specific features which are unique to each species. Bark surface irregularities such as overgrown branch stubs, insect damage, old mechanical damage, etc. will likely disqualify the log as a potential veneer log. It is generally assumed that no surface indicators of interior defects are present in a quality veneer tree or log.

Veneer logs or the trees from which the logs are produced should be straight and well rounded. Bow and crook in a log creates an aesthetic problem by causing the cathedral pattern in flat sliced veneer to run in and out of the sheet. Tension wood is frequently present in leaning trees and buckle can occur when the veneer is dried. Logs which are not well rounded or have an off-center pith also result in veneer with less than a desirable grain pattern and are also likely to result in veneer buckle (fig. 3).

Growth rates and thus ring width need to be uniform across the entire cross section of the log. Thus, thinning to encourage faster growth of potential veneer trees is not desirable. Growth rates of six to nine rings per inch are usually acceptable. Fast growth or a very slow growth rate are not preferred.

Veneer quality trees should be healthy, well formed trees on well-drained timber sites. A past history of grazing and or fire will reduce the quality and value of any potential veneer tree.

Most hardwood species grow over a wide geographic range. As such, climatic conditions, soil types, elevations, insect and disease potential, and other factors vary. Within the geographic range of each species there are certain specific areas where buyers feel the highest quality trees come from. Buyers will indicate that high quality trees can come from other regions as well but the probability of finding a superior tree is much reduced.



Figure 3.—Buckle or irregular surface in walnut veneer. Note the irregular appearance of the tag.

Cherry (*Prunus serotina* Ehrh.) is currently one of the most valuable hardwood veneer species and an excellent example of the importance of geographic location. Figure 4 shows two 19 inch DIB by 10 foot logs each containing 141 board feet Doyle Scale. The log on the left is from Indiana and it is valued at about \$280 while the one on the right is from Pennsylvania and it is valued at about \$1200. The Indiana log has irregular, darker reddish color and numerous gum spots. The Pennsylvania log has a lighter pink color and is much less prone to gum spots. On the other hand, walnut (*Juglans nigra* L.) from Indiana and hard maple (*Acer saccharum* Marsh.) from Michigan and the Northeast would be preferred, as compared to that from other locations.



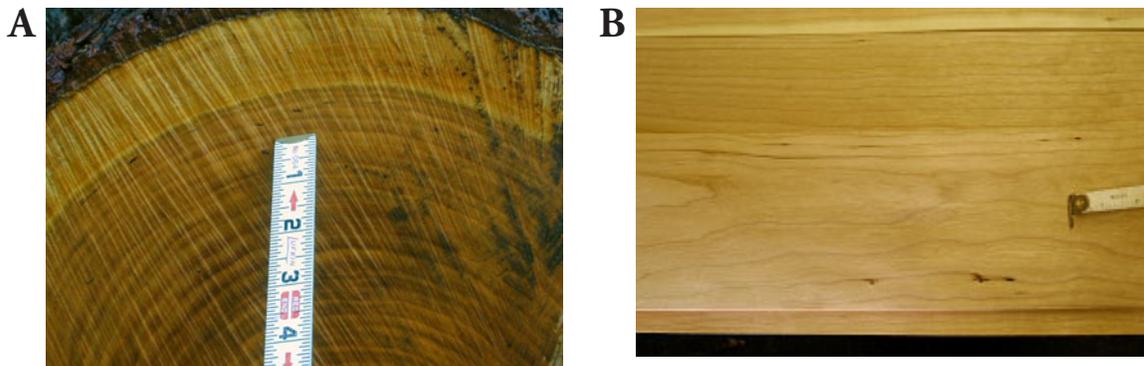
Figure 4.—Geographic origin can be very important in determining veneer log value. The cherry log on the left is valued at \$280 while the one on the right is valued at \$1200. The diameter and length is the same for each.

Cherry

Gum (figs. 5A, B) is probably the most serious defect affecting veneer quality in cherry. In addition to small spots, gum can be found in large patches, probably as a result of wounding, or even in a ring completely circling a portion of the stem. The presence of gum cannot be detected in standing trees unless a surface residue happens to be present. Buyers prefer to purchase cherry timber in those regions of the country where gum spots are least likely to be found. These regions include the higher elevations in Pennsylvania and parts of New York and West Virginia. Cherry grown in other parts of the country almost always has some gum spots present.

Most gum spots are caused by at least two different groups of insects. The most common insect is the peach bark beetle (Rexrode, 1981). These beetles occur throughout the geographic range of cherry and sometimes actually kill relatively large trees. The beetle can be found in the gum which is exuded from the trees. The beetles attack the cambium layer, and the gum is formed in response to the insect. Unfortunately, it appears that peach bark beetles may build up in large numbers in tree tops after a timber harvest. The beetles then emerge and attack the residual crop trees, causing permanent gum spots in the main tree bole. Therefore, it would appear that cherry veneer which is relatively free of gum spots from peach bark beetles would come from undisturbed stands which also have had little natural mortality or damage.

Cambium miners (Rexrode and Baumgras, 1980) can also cause gum spots in cherry. By carving its galleries, the cambium miner destroys a portion of the cambium, which later becomes covered over by healthy cambium growth and wood. These galleries consist of damaged parenchyma cells and insect feces. The cell damage can, but does not always, result in the production of gum. The parenchyma flecks or damaged parenchyma cells are seldom a defect in themselves. Parenchyma cells are just one of several different cell types which make up the wood of hardwood trees. These cells are generally used for food storage and are relatively thin-walled compared to wood fibers.



Figures 5A, B.—The small dark spots on the end of this cherry log are gum spots (A). They disfigure the otherwise clear veneer shown (B).

“Tear drops” or an irregular bump in the growth rings (fig. 6) can also be found in the ends of cut cherry logs. These tear drops cause a dimple in the veneer and are objectionable.

A light reddish brown heartwood color is preferred in cherry. A dark red color, variable color, or a greenish cast which can develop at the heartwood sapwood interface and wide sapwood is objectionable.

Walnut

Black walnut (*Juglars nigra L.*) was the premier domestic North American species since the beginning of the sliced veneer industry. Due to increased costs, over cutting, and consumer preference its popularity declined in the 1980’s. Now, it is once again in good demand and the availability of logs is also good.

Wood color as well as uniformity of color in walnut is an important factor. The best colored walnut when first cut is light greenish or mint color. As the wood is exposed to the air, it turns a gray or mousy brown color (fig. 2A), which is considered ideal. Unfortunately, the color can vary or it can lack uniformity (fig. 2B). Muddy walnut, that which is dark or splotchy, is objectionable. The color of walnut can also be affected by manufacturing variables such as cooking schedules and processing time before drying.

Small amounts of figure, sometimes called flash are also not desirable (fig. 7). Bird peck, also called worm by the veneer industry, is an important defect in walnut (fig. 8). Yellow-bellied sapsuckers probably cause most of the bird peck in walnut. It is generally believed that the bird pecks a hole to cause the flow of sap. Insects are attracted to the sap, and the bird then feeds on them. A small hole plus stain or flagging can result in the veneer.

Pin knots (fig. 9A, B, C) like bird peck can be hard to recognize in standing walnut trees, especially when only a few are present. These defects are the result of suppressed dormant buds which persist for many years as a bud trace or pin knot. As the name implies, the buds may not actually break through the bark, so in some instances they cannot be easily detected. However, sometimes, due to a stimulus such as thinning and light, the bud may sprout. The sprout may develop into a small limb that often dies, but normally the bud trace continues to form. Pin knots are best observed on the ends of the log after the tree is cut or where the bark has peeled loose and they appear as sharp spikes. On flat-sliced veneer, they appear as pin knots, but on quartered surfaces they appear as a streak or “spike” across the sheet of veneer.

Growth rate is important in walnut. The industry uses the word “texture” to define growth rate. Soft texture refers to a slow growth rate while hard texture refers to a fast growth rate. Many buyers will find 8 to 9 rings per inch of diameter the most desirable.

Fast growth trees also tend to have a wide sapwood zone (fig. 10). The sapwood is the light colored wood to the outside of the darker heartwood. Sapwood is usually discarded in high quality walnut veneer. Deeply furrowed bark which is not patchy tends to be faster growth and have a wide ring of sapwood. Figure 11A, B shows the bark on fast growth and slow growth trees.



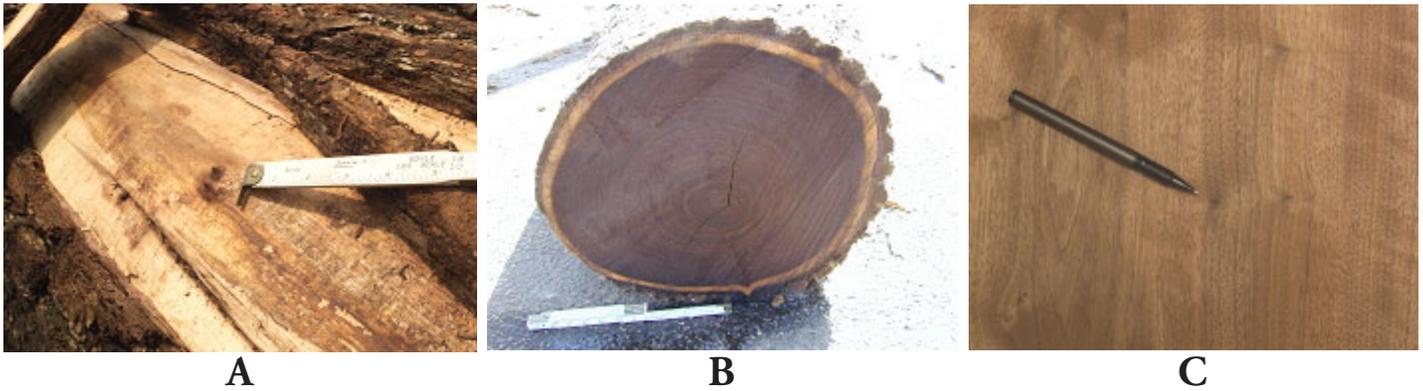
Figure 6.—Cherry log showing irregularity in the concentricity of the growth rings or “Tear Drop” which causes a dimple effect in the veneer sheet.



Figure 7.—Figure or flash in walnut veneer. Small amounts of figure as shown here are usually not desirable. A highly figured flitch, on the other hand, commands a premium price.



Figure 8.—Excessive bird peck or “worm” in black walnut and irregular color throughout the heartwood region.



Figures 9A, B, C.—Pin knot in black walnut. Note the “spike” where it penetrates the bark (A) and the light radial streak (B) and finally the pin knot in veneer (C).



Figure 10.—Fast growth walnut trees tend to have a wide sapwood (left) while slow growth trees tend to have a narrow sapwood (right)

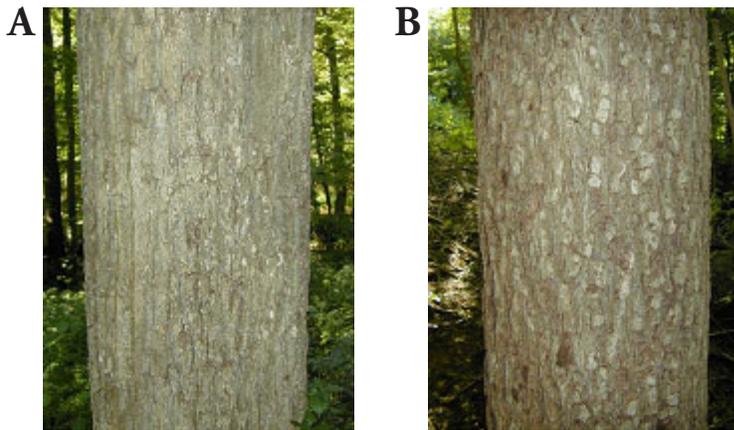


Figure 11.—Bark of fast growth (A) and slow growth (B) walnut trees.

White Oak

White oak is another very important veneer species, particularly in the export market.

True white oak (*Quercus alba* L.), especially trees with large patches of flaky bark in the upper portions or “forked leaf” white oak, are the most desirable in the white oak group. Chinkapin oak (*Quercus muhlenbergii* Engelm.) sometimes is used, but the resulting veneer has a greenish to brownish cast. Bur oak (*Quercus macrocarpa* Michx.) is also used, but careful selection is required to avoid its more common dark brown color and possible “scalloped” appearance of the growth rings which can be seen on the ends of the logs. The scallops result in shiny spots on the veneer sheets.

Color in white oak, like all veneer species, is critical. Current markets prefer a very light, uniform-colored white oak. Contrast in color and dark colors are objectionable (fig. 12A, B). Obviously, color cannot be judged in standing trees. Buyers do, however, develop preferences for certain geographical areas and site



Figures 12A, B.—Note the unique light color and centered uniform cathedral of the white oak flitch (A) as compared to the coarse, irregular flitch (B).

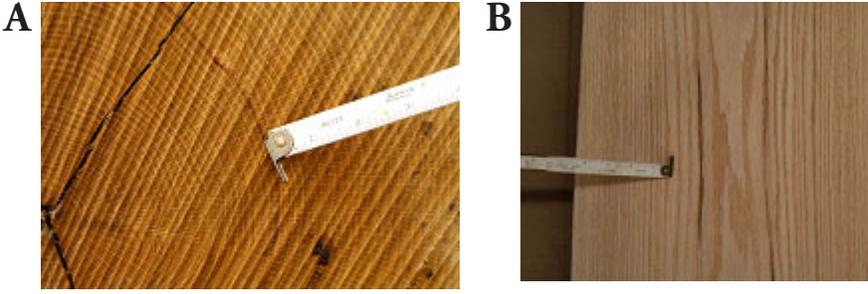


Figure 13A, B.—Mineral stain following a growth ring on the end of a white oak log (A) and appearance of mineral in white oak veneer (B).

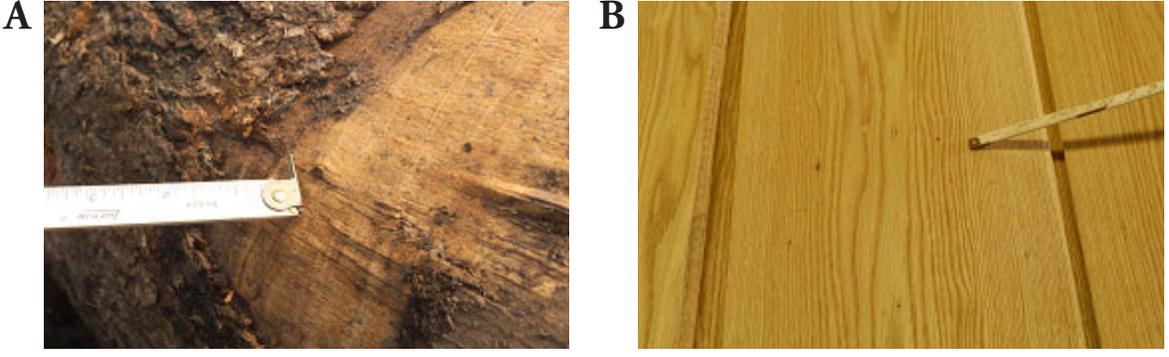


Figure 14A, B.—Pin knots in a white oak log (A) and in the veneer (B).

characteristics because their past cutting experience has taught them that those areas produce the desired color and quality of veneer. Also, very old and slow-growth white oak trees tend to be pink in the center and brown to the outside. Mineral streaks are also a common defect (figs. 13A, B)

Epicormic branching or sprouting from latent buds is a common defect in white oak (figs. 14A, B). Several buds may form a cluster. The resulting veneer will have a small pin knot or cluster of pin knots.

Stump worms and the surrounding dark flagging or mineral stain is associated with white oak grown in areas which are poorly drained or have been pastured. This defect is generally concentrated in the bottom two feet of the butt log, and it is often impossible to detect until cutting occurs.

A number of different species of borers can affect white oak (Solomon, 1980). White oak borers attack trees less than about 8 inches in diameter (fig. 15). Thus, they are generally not detectable in veneer-sized trees, nor do they damage the outer more valuable portion of the tree. Other borer species also attack white oak, but normally the damage is restricted to declining trees. Bore damage is difficult to see in standing trees. Consequently, when it is found, buyers will assume that more damage is present than what can be seen and severely degrade the tree.

Finally, some large white oak trees will have a “bulge” (fig. 16) near the base of the tree that resembles an old time coke bottle. This is not considered a defect for veneer logs.



Figure 16.—Bulges at the base of some large white oak, also called “coke bottle effect” are not considered defects.

Red Oak

Red oak (*Quercus rubra* L.) is also commonly veneered. Color and mineral stain are two of the most common problems associated with red oak veneer, in addition to obvious defects such as overgrown limbs, borers, wounds, etc. Again, the premium material demanded by the market is a very light-colored veneer. Mineral stain is common in red oak and may take the form of isolated spots or follow along the annual rings (fig. 17). It is objectionable in most finished products particularly if a natural finish is being applied. In addition, the wood often tends to split or break apart when mineral stain is present.

Mineral-free, light-colored red oak is more commonly found in certain regions of the country such as lower New York, Pennsylvania, northern Indiana, and southern Michigan. Therefore, it would seem that site or soil might also be a factor. Regardless of the cause, the presence of mineral in a particular area will result in veneer log buyers offering reduced prices for standing trees of potential veneer quality.

There are three major types of borers which attack red oak. According to Donley and Terry (1977) these are red oak borers *Enaphalodes rufus* (Haldeman), carpenter worm *Prionoxystus robiniae* (Peck) and *P. macmurtrci* (Guerin-Meneville), and the oak timber worm *Arrhemodes minutus* (Drury). Borer holes can range from 1/100 to 1-1/2 inches in diameter. Smaller borer damage is nearly impossible to detect. Larger borer holes which have healed over can be seen by experienced people and if “sap wet” are easily detected. Carpenter ants often enter trees through large borer holes and keep the wood open and further damage the tree. Old trees or stressed trees are more prone to damage.

Since most borer damage is hard to detect, buyers will be very cautious if any borer holes are found.

Crossbars or oblong horizontal bumps on a red oak log may indicate a defect that goes all of the way to the heart. However, buyers have also indicated that in some regions no defect results.

Sugar Maple

Sugar or hard maple (*Acer saccharum* marsh.) is prized for its white sapwood (fig. 18). The whitest maple is reported from Michigan and the Northeast. However, maple from other regions is also veneered. The color of maple



Figure 15.—Bore damage in the butt log of white oak. This log also exhibits slow and fast growth rate, ring shake, and very poor color.



Figure 17.—Mineral stain along an annual ring in red oak.



Figure 18.—High quality hard maple log from Vermont. Note the near absence of any heartwood.

sapwood is also affected by the season of the year and length of log storage as well as processing variables. From a color perspective, it is a very difficult wood to process.

The heartwood in sugar maple is a brown color and is considered “false” heartwood (Shigo and Larson, 1969). False heartwood is normally caused by a wound or opening in the bark of the tree such as a broken or dead limb stub.

The extent and intensity of false heartwood depends on the vigor of the tree, the severity of the wound, and the time that the wound is open. Discoloration continues to advance as long as the wound is open and is often irregularly formed throughout the stem. If the wound heals, the entire cylinder of wood present when the tree was wounded may not become discolored. The cambium continues to form new growth rings that are free of discoloration. Thus, vigorous fast growing trees with few branch stubs or wounds will produce the widest sapwood.

Mineral streaks ranging from 1 inch to several feet long are a common defect in maple and result from wounds such as broken or dead branches, bird peck, and mechanical damage (figs. 19, 20A, B). After a wound occurs, the living cells in the wood surrounding the injury react by forming materials that

inhibit the infection. These materials are deposited in the cells and may appear green initially but later turn different shades of brown. High percentages of mineral, especially potassium, are found in the cells. Some of the wood is very hard and cutting tools can be damaged.



Figure 19.—Mineral stain in hard maple.



Figures 20A, B.—Bird peck on a hard maple tree (A) and resulting defect in the veneer (B).

Sugar streaks or flecks, narrow brown-colored marks about 1/4 to 1 inch long, can also occur in sugar maple (fig. 21) The streaks are caused by cambium miners. Cambium miners attack and disrupt the cambium or growth layer of the tree (Anderson, 1960, Graham and Knight, 1965 and Hardwood Research Council, 1987) The tree plugs the gap in response, but the grain pattern has been disrupted. Cambium miners may bore from the top of the tree all of the way to the rail soil before they exit.

White Ash

White ash (*Fraxinus americana* L.) is sliced into veneer and it is also the white sapwood of this species that is preferred. Like hard maple, white ash has a false brown heartwood. With this species it is not uncommon for the heartwood to be very small at the butt of the tree but then expand to a significant portion of the tree diameter further up the stem. Unfortunately, the extent of objectionable heartwood in the top of the butt log is not known until after it is crosscut often to a shorter length veneer log as compared to a longer sawlog.



Figure 21.—Sugar streaks or flecks in hard maple

Glass worms, also called “turkey tracks” or “worm tracks”, occur in white ash. This zig zag pattern of light-colored wood is caused by the cambium miner. In some cases the wood associated with the glass worm damage turns nearly black. The characteristic is objectionable because it will not accept stain and finish like normal ash wood.

Other Species

There are several other central hardwood species all of which are processed into veneer at various times. In terms of volume and or value, these are relatively minor species and thus not included in this limited discussion.

Acknowledgments

The following companies have been most generous in sharing their knowledge of log quality with the author and many other members of the forestry, sawmilling, and logging industry as well as with landowners.

Coldwater Veneer, Inc., 548 Race Street, Coldwater, MI 49036-2122; 517-278-5676

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Thiesing Veneer Company, 300 Park Drive, Mooresville, IN 46158; 317-831-4040;
thiesing@indy.net

David R. Webb Company, Inc., Box 8, Edinburgh, IN 46124; 812-526-2601;
drw@davidrwebb.com
www.davidrwebb.com

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KEY ATTRIBUTES ASSOCIATED WITH VENEER QUALITY TIMBER THAT MAY BE IMPACTED BY FOREST MANAGEMENT PRACTICES

Delton Alderman, Jan Wiedenbeck, Paula Peter, and David Brinberg[†]

ABSTRACT.—Veneer log prices are commonly 4 to 10 times the price of grade 1 sawlogs yet veneer quality trees constitute less than 1 percent of the hardwood timber resource in the northeastern United States. Veneer manufacturers and their customers impose guidelines in specifying wood quality attributes that are very discriminating but poorly defined (e.g., exceptional color, texture, and/or figure characteristics). In order to better understand and begin to define the most important attributes that distinguish veneer logs from sawlogs and high-end from low-end veneer logs, we are conducting a series of studies in which we are collecting veneer log attribute information from veneer log buyers and sellers, veneer manufacturers, and veneer sales personnel. Broad veneer log quality requirements and specific requirements for the most highly demanded veneer log species (cherry, red oak, white oak, white ash, black walnut, sugar maple, and yellow-poplar) are being elucidated. The relative importance of critical, but often subtle, veneer log quality attributes are examined for each of these species. Guidelines for assessing the true veneer value potential of high-quality trees can be based on buyer-cue based log attribute metrics. Important veneer attributes include log form, growth-ring consistency, heartwood/sapwood proportion, and wood color as well as the more obvious defects. Forest managers can consider the relative importance of the different attributes when they initiate silvicultural techniques to produce high-quality hardwood timber.

Introduction

Veneer Log Value and Markets

The pinnacle of log quality for hardwood products manufacturers is the appearance-grade veneer log — logs that are capable of producing veneer that is highly visually appealing (as opposed to veneer that is used in hidden applications). But high-quality trees that contain these top-quality logs are relatively rare, representing less than 1 percent of the hardwood sawlog inventory in the northeastern United States (Hoover and Gann 1999). Because of the high value of the veneer product that comes from appearance-grade veneer-quality trees and their relative scarcity in the forest, these trees command a significantly higher price than do trees that contain only sawlogs. The veneer logs cut from these highest quality trees typically cost from 1.5 to 6 times the price of grade 1 sawlogs. Because of the exceptionally high prices that are paid for veneer-quality trees, a large portion of a quality stand's timber value may be derived from only a small fraction of the trees in the stand. These price differentials can provide significant economic incentive for both the landowner and the logger to manage their resources to optimize the production and recovery of veneer logs.

When managing a timber stand, there are many things that, if done wrong, can damage trees and greatly reduce the price that the landowner receives when he/she decides to sell his/her timber. Timber value is lost if the timber harvest is mistimed; for example, when veneer-quality trees are removed before they are of sufficient size. Also, if potential veneer-quality growing stock is cut or damaged during thinning operations veneer log yield(s) will be reduced. Alternatively, if too much growing stock is removed during thinning, wide growth-ring spacing may ensue, which decreases the value of a veneer log. Other timber management strategies that might be less obvious also affect the value returned to the landowner. By enhancing our knowledge of veneer-quality requirements, we can better understand how these other management practices influence the yield of veneer-quality timber. With lower value products (e.g.,

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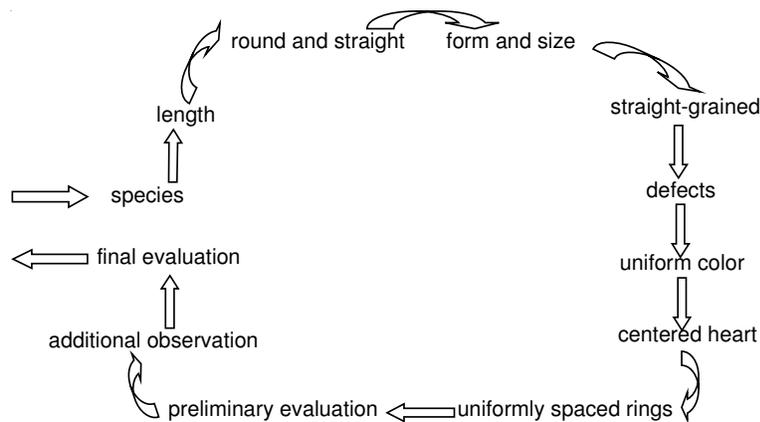


Figure 1.—Veneer log procurement is an iterative process.

rubberwood, medium density fiberboard, plywood) coming into greater use in the construction of furniture, worldwide demand for U.S. hardwood veneer is rapidly increasing. This will lead to continuing price inflation for veneer-quality timber and stronger incentives to manage prime timber stands to promote the yield of the highest-grade sawlogs and veneer logs.

Veneer Procurement Judgment Tasks

In our previous research, wide variations in the decisions made and values assigned in veneer log procurement have been observed. For example, the definition of acceptable ring count (e.g., fine, medium, or loose texture) varies widely among individual buyers. Ring count also is an attribute on which secondary manufacturers place great importance. This attribute is important for forest management as well. Forest management decisions such as choosing between clear-cutting, single-tree selection, or crop-tree release treatments can be directly affected by the weight buyers place on ring count. The most familiar judgments made by buyers during log procurement have to do with species and several log quality attributes: 1) butt or upper logs (log form and size); 2) freshness of cut; 3) roundness and straightness; 4) straight-grained; 5) free of knots, bark distortions, decay, seams, and bird peck on each of the four log faces; 6) heart (pith) centeredness; 7) color uniformity; and 8) uniformity of ring spacing.

The diagnostic evaluation of logs can be considered an iterative process (fig. 1) that begins with the species and moves through an evaluation cycle until a final decision is arrived at (e.g., Prime plus, prime, select, No. 1, No. 2, etc.).

Social Judgment Theory

SJT is a systems-oriented viewpoint for analyzing human judgment in discrete ecological (i.e., environmental) situations. According to SJT, an individual does not have direct access to information about objects in the environment. Instead, one's perception of those objects influences judgment(s). Perception is an indirect process, mediated by the set of proximal cues one receives. It is assumed that judgments result from the integration of "cues" or sources of perceptual information arising in the "ecology" (environment). SJT includes representative design, cues (i.e., information), multiple correlation, and regression, and the Lens model (fig. 2).

Brunswik's SJT focuses on achievement and the degree to which a subject successfully attains his/her goal is *achievement*. One of the primary benefits of SJT is the derivation of criterion values (correct values) or achievement that arises from analysis and permits the researcher to compare judgment processes to the environmental processes and also judgments between the subjects. Processes and judgments have a common interface that includes the proximal cues in perception and the *task system* and the *cognitive* (judgmental) system. The task system is defined in terms of the relations between the cues (X_i), the distal variable (Y_c), and relations among the cues (X_i). The cognitive system is defined in

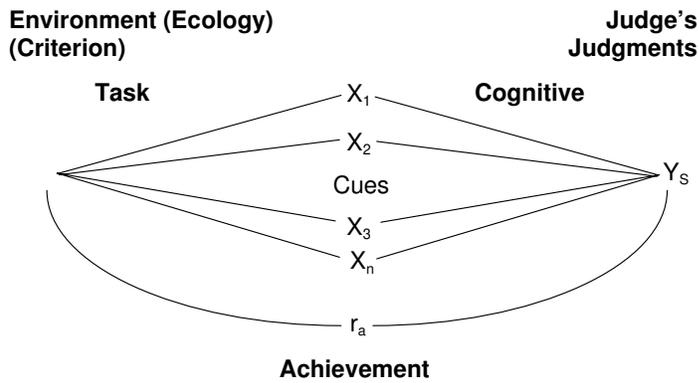


Figure 2.—Brunswik's Lens Model.

terms of the relations between the cues (X_i) and the judgment (Y_s). R_a is achievement, which simply is how good the judge's judgments are when compared to other judges, criterion, or other distal variable(s) of interest. Comparison results in cognitive feedback as a tool to facilitate discovery and learning (Cooksey 1996).

SJT allows for the decomposition of the judgment process after the judgments have been rendered (*a posteriori*) and is accomplished by multiple regression analysis to recover both cue weights and prediction equations. "Brunswik's Lens Model" is employed next, it allows for comparisons between judgments, subjects, and ecological systems. Brunswik's representative design emphasizes sampling subjects in their environment.

SJT is appropriate for researching veneer log procurement. The tasks for log buyers involve making decisions under uncertainty that require the evaluation of visual cues (e.g., species, diameter, length, attributes, and defects). For example, log buyers must decide whether to purchase logs based on the aforementioned cues, which also include color, grain pattern, and blemishes as well. Other, but not all, uncertainties include: is the color consistent throughout the log, is the grain pattern consistent throughout the log, are blemishes extensive or are they confined, and market uncertainties. There are no heuristics for predicting buyer judgments or success. Buyers must make many such judgments daily, but little is known about how buyer judgments are made or how accurate the judgments are.

Decisions about the end-user market are equally important and judgments are typically made during the evaluation process and before log evaluation is complete. From an SJT perspective, buyers generally do not have access to information about cue weighting and the degree of uncertainty in their judgments. From a researcher perspective, minimal attention has been paid to the utility of these evaluation cues in distinguishing one attribute or defect from another. This has occurred because necessary information was either unknown, or unavailable, and varies from setting to setting (i.e., regional differences in veneer log attribute requirements).

Objectives

Our overall project goal is to identify and learn how to influence those tree growth attributes that have the greatest effect on the quality and value of high-grade hardwood logs. The objectives of this study are to:

1. identify critical factors in the selection of logs for veneer,
2. identify relationships between those attributes/defects that may assist veneer buyers in assessing the quality of logs,
3. discern the allowable variability along those factors, and
4. produce metrics that can be used to evaluate the veneer potential of trees grown under different forest management systems.

Methods

To determine the most important attributes that distinguish veneer logs from sawlogs and high-end veneer logs from low-end veneer logs, we conducted informal surveys with 3 veneer log graders (buyers for a veneer producer), 4 veneer log brokers, 15 veneer producers, and 1 veneer broker (table 1).

Subsequent to our initial survey, Social Judgment Theory (SJT) will be utilized to conduct a more rigorous assessment of veneer log characteristics. SJT has been used to evaluate decision-making in several disciplines (e.g., physician decision-making, weather and climate forecasting, managerial decision-making, workplace judgments, etc.). By investigating the decision-making process, we will be able to ascertain salient veneer log attributes and/or defects via analysis of cues and the weights of log buyers, and then develop evaluation metrics. SJT will be employed in evaluating decision-making among log graders and buyers as they appraise the quality of veneer logs. If graders and buyers utilize different criteria to evaluate veneer log quality, then inefficiencies are created in log procurement. For example, suppose log buyers place substantial weight on the quantity or discrete types of defects (character marks) in the log, while others do not and thus reduce the value of a log as a result of these marks. Greater efficiency and cost savings can occur given a higher level of coordination among the criteria used to evaluate veneer logs.

Applications of SJT to Procurement Judgments

In this research project, we will incorporate a strategy-capturing policy at the idiographic level. Strategy capturing refers to research that analyzes how buyers weight cues in making judgments (i.e., the right hand side of the lens model). A single system design (fig. 3) will be utilized where buyer judgments are measured in the absence of final criterion (Y_c). Strategy capturing does not provide a method to measure the accuracy of buyer judgments because the final criterion have not been established (e.g., research on final clipping decisions, consumer preferences). Essentially, we are researching the value systems or establishing which cues are important to judgments and how they are utilized (Ullman and Doherty 1984). However, it should be noted that ongoing and future research includes both double and triple system designs where buyers, clipping technicians, and consumers' judgments will be evaluated simultaneously. This will allow for the opportunity to discern inefficiencies in the procurement, manufacture, and marketing chain.

This research is addressing several pertinent issues (adapted from Wigton 1996):

1. Do buyers use cues that are generally recognized as important?
2. Do they use previously unrecognized or irrelevant cues?
3. Do buyers use cues similarly?
4. Do buying strategies become more similar as experience increases?
5. Do buyers utilize all of the information available in making judgments or just a few cues?
6. How do weights elicited in this manner compare with self-described strategies?
7. Can strategies be characterized or clustered in meaningful ways?
8. Can we identify clusters of buyers with similar strategies?

These questions relate to a buyer judgment model. This research design also has a non-ubiquitous aspect in that all buyers respond to the same set of cues: species, logs, and their respective attributes

Table 1.—Initial participants in survey to determine the most important veneer attributes.

Business type	Location	Number
Veneer producers	IN	6
	KY	1
	OH	2
	PA	2
	VT	2
	WV	2
Veneer brokers	PA	1
Veneer log brokers	PA	3
	WV	1
Log buyers	NH	2
	WV	1

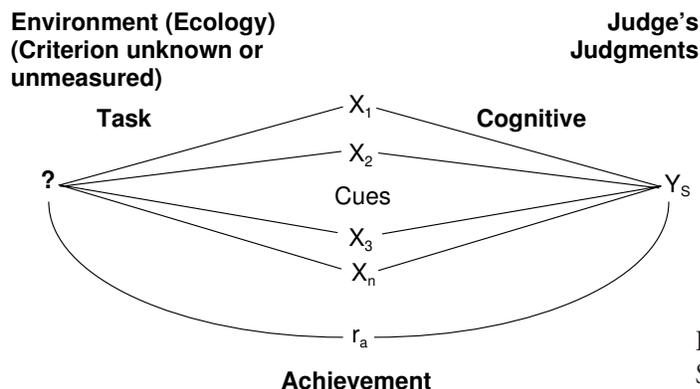


Figure 3.—Brunswik's Lens Model – Single System Design.

and defects. Wigton (1996) references Ullman and Doherty (1984) and states this will allow for the opportunity to examine additional questions regarding the variation among buyers:

9. Do buyers have similar means and variances?
10. What is the correlation among judges (buyers)?

Population and Sample Frame

The sample frame consists of members of the Hardwood Plywood and Veneer Association (HPVA), which includes approximately 30 firms. The subjects will include veneer graders and buyers, or both. A census (i.e., 100% sample of graders and buyers) will be attempted for each species.

Cue Development and Testing

In pretesting, multiple photographs of cherry logs (as a stimulus) have been created for use during the interviews with the log buyers. These photographs vary from extremely high quality logs, to those that have various character marks. After pretest interviews, revisions will be undertaken, if necessary. The stimuli (i.e., photographs of the log and veneer slices) and a questionnaire instrument will be administered to assess perceptions of the log attributes in the sample.

Data Analysis

Data analysis will take two basic forms: multidimensional scaling (MDS) of the attributes associated with the veneer product, and a regression approach for the policy-capturing component of the research. MDS will allow for a perceptual map to be produced and regression will yield the relative weights buyers place on the veneer log characteristics. The goal of a strategy policy capturing approach using multiple regression is to produce a linear equation that optimally weights each cue in terms of the discrete cue's predictive contribution to the judgments (Cooksey 1996). The judgment model is presented below:

$$Y_s = \frac{[b_0 + b_1 X_1 + b_2 X_2 + b_3 X_3 + \dots + b_k X_k] + \varepsilon}{\dot{Y}_s}$$

where;

- b_0 = regression constant,
- b_i = cue regression coefficient,
- X_i = cue,
- Y_s = judgment,
- ε = error.

Upon judgment model capture, judgment predictor values can be produced for each case in the judgment task. Correlation between authentic judgments and corresponding predicted judgments across m profiles results in a goodness-of-fit measure for the regression model, which R_s is the term for a discrete captured judgment policy. R_s^2 is the squared multiple correlation and yields the proportion of judgment variance that has been captured.

Cue Weights

Cue weights are critical in assessing the salience or degree of importance as the researcher is developing metrics from these weights. One method for assessing the cue sensitivity is to standardize the raw regression weights and convert cue scores to standard Z -scores. The modified judgment model would take the form of:

$$Z_{Y_s} = \beta_1 Z_{X_1} + \beta_2 Z_{X_2} + \beta_3 Z_{X_3} + \dots + \beta_n Z_{X_n} + \epsilon$$

$Z Y_s$

Prediction now concerns the standard score for each judgment; each β_i provides a less ambiguous indicator of cue importance (Cooksey 1996).

Results

Initial Survey Results

The hardest-to-see veneer log defects, according to the veneer log buyers we spoke with, are bird peck, T-shaped scars, pin knots, ingrown bark (especially in cherry), and insect-induced localized defects such as glass worm in ash and sugar streaks in maple. Most log buyers surveyed indicated that sugar maple is the most difficult species in which to predict veneer log quality. White oak also was cited by multiple sources as being particularly difficult to evaluate. Red oak, ash, and walnut are considered easy species in which to judge quality. Cherry, in some regards, is simpler to evaluate. However, pin knots and gum pockets are important defects in cherry that can be difficult to detect from an external examination. Also, some cherry logs have extremely flaky bark, which is much more difficult to read.

These generalized quality standards for veneer logs may be more or less important depending on the veneer market segment, species, and manufacturing system. For instance, logs above the butt log are acceptable when large trees with high crowns produce second logs of large diameter that do not have any branches. However, due to pin knots, the closeness to the surface of overgrown knots and smaller bole diameter, the majority of high quality veneer logs are butt logs – from 60 to 98 percent depending on species. Species that grow straighter with less taper, such as cherry and yellow-poplar, will yield more veneer logs from upper portions of the tree than will white oak, walnut, and hard maple.

Tapered or elliptically shaped logs can be a problem for some veneer products, but can be an asset if veneer with cathedral pattern is desired. Even severe butt taper is usually not a serious problem because it is removed with “butt reducers” during log preparation prior to peeling or slicing. A butt reducer grinds off buttresses and swollen bases and is usually accompanied by debarking. However, grain deviations associated with taper may be a problem if the veneer will be bent for furniture parts or woven into baskets.

Sweep is a defect that reduces the usable volume of a veneer log and affects the grain pattern of the veneer. The altered grain pattern can decrease the grade and value of the veneer. In addition, sometimes the change in grain direction characteristic of sweep causes buckling in certain types of high-end veneers. Crook is similar to sweep but usually is caused by deflection of the main stem caused by a major branch. Thus, in sectioning logs with crook prior to slicing, the position of the branch knot also must be considered.

External evaluation of logs includes not only log form, bark, and outer-wood evaluation, but also evaluation of log ends. Some log defects, such as gum in cherry, sugar streak in maple, and worm holes in

several species are seen only on the freshly cut ends of logs. Log buyers will spray log ends with water to enhance the visibility of the hardest to see defects (e.g., sugar streak and mineral stains/streaks). Growth ring consistency (or texture), density, and wood color also are judged by viewing log ends. The location of the pith (centered or off-centered) and evidence of tension wood (which can buckle and tear in the manufacturing process) are evaluated here too. Finally, the heartwood and sapwood content of logs is assessed by looking at the two log-end cross-sections.

Although centered heart and uniformly spaced rings are important for production of sliced veneer with symmetrical appearance, some cutting methods can bypass this requirement. Veneer manufacturers report a tendency for logs with off-centered piths to be more prone to splitting. Off-centered heart is addressed when the log is flitched (cut into sections) in the sawmill — the log is flitched so that the edge of the log sections goes through the pith. By doing this, the pith is contained in the part of the flitch that is not recoverable as veneer (called the backing board).

Non-uniform rings (or double texture) are less of a problem in the furniture-grade veneer market since this veneer is generally cut into smaller face sections than is door and panel veneer.

Sapwood is desirable in maple, ash, yellow-poplar, hackberry, sycamore, and sometimes in birch and hickory, whereas heartwood is desired in walnut and cherry. Sapwood color may be critical, such as in architectural panels and high-value furniture, or it may be less important when used in strips that will be dyed or given clear but darkening stains, such as basket-weaving strips. In white oak, red oak, hickory, and sometimes yellow-poplar, the heartwood-sapwood distinction is less important due to the color consistency between the two wood regions and less intense product color requirements.

Based on knowledge we gained from this series of visits with veneer log procurement and production personnel, it is evident that subtle appearance factors are critically important determinants of veneer value in both domestic and foreign high-end veneer markets (architectural, door, and panel). Therefore, these subtle factors are of keen importance to log buyers/graders when purchasing and sorting/selecting logs for processing to supply these high-end markets. The basic color (shade) of the wood is important for all species, but especially so for maple, walnut, ash, and white oak in today's veneer markets. Minor color variations and blemishes are considered defects in these three high-end veneer market sectors since these sectors require large veneer sections of consistent appearance. Thus the color variations caused by insect pests are a concern. These include gum pockets and rings in cherry, sugar flecks in maple, and glass worm in ash. Mineral streaks in oak and maple also are important since they too cause color variations that are defects in high-grade veneer. Next to color consistency, grain pattern consistency is almost as important. The texture of the growth-ring pattern must be uniform (i.e., consistent rate of growth) and the grain must be relatively straight.

Expected Results from SJT

Social judgment theory's statistical and conceptual approach provides a most informative method for assessing and analyzing several factors of the veneer log procurement process. As veneer procurement and production becomes more competitive, SJT allows for valuable insights. These insights include categorization and metrics for attributes and defects, ring count metrics for procurement and forest management, and useful information on the weights buyers place on other elements of the procurement process.

Expected Impact

Foresters will have better information on critical attributes affecting stand quality and value which will allow them to make forest management decisions that will promote optimal financial return to the forest landowner. In addition, timber sales consultants will be more aware of those criteria being used by buyers of high-quality timber when they make their pricing decisions; this will remove some of the ambiguity from the timber sales process so that the highest possible return to landowner is realized. Finally, the attribute importance ratings and metrics derived in this study can be used by individual companies to

produce a judgment system that will lead to more consistent procurement among company timber and log buyers.

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DETERMINING SAFE CLEARING LIMITS FOR SKID ROAD/TRAIL CONSTRUCTION

Chris B. LeDoux[†]

ABSTRACT.—Most of the timber harvested in the United States is extracted by ground-based skidders and crawler/dozer systems. Ground-based systems require a primary transportation network (skid trails/roads) throughout the area being harvested. Following cutting, logs are skidded or dragged along these surfaces to the landing, where they are loaded onto trucks and transported to processing plants. The degree of excavation/construction on these skid roads/trails varies with the steepness of the terrain and the size of logs/logging vehicles. Excavation/construction creates certain hazards that could damage property and threaten human lives. In one incident that led to a court case, excavation undercut the lateral support of a tree that was left on a vertical (vs. backsloped) cutbank. The tree later fell, severely injuring a logger. We provide a decision algorithm for determining practical clearing limits/guidelines for the proper construction of skid roads/trails.

Society is increasingly asking more of the nation's forests with respect to wood products (veneer, sawlogs, pulpwood, fuelwood, etc.) and recreational opportunities (hiking, biking, horseback riding, snowmobiling, etc.). Increased access to and use of our forest resources requires a network of haul roads and skid roads/trails (Skelton 1949; Meyer 1970). Haul roads generally are permanent and well designed, and receive heavy vehicle use. Most skid roads/trails are temporary and often poorly designed, but vehicle traffic on these roads is light once harvested timber has been extracted. However, skid roads/trails (hereafter called trails) continue to be used for recreation activities such as hiking, hunting, and horseback riding even after many of them have been gated or otherwise blocked off.

Most of the timber harvested in the United States is extracted by ground-based skidders and crawler/dozer systems. Ground-based systems require a primary transportation network throughout the area being harvested (Wackerman et al 1966; Pearce and Stenzel 1972). Following cutting, logs are skidded or dragged along these trails to the landing, where they are loaded onto trucks for transport to sawmills, paper mills, OSB plants, veneer mills, etc. (Conway 1976). The degree of construction/excavation required on these trails varies with the steepness of the terrain and the size of logs/logging vehicles (Wenger 1984). Excavation/construction can create certain hazards that if not eliminated can damage property and threaten human lives. In one incident that led to a court case (Melvin Willard White vs. Wenturine Brothers Lumber, Inc., ICI Explosives, U.S.A., and American Forestry Consultants, vs. John Bouch Logging, In the Court of Common Pleas of Westmoreland County, Pennsylvania, Civil Division No. G.D. 382-2000, 2003), excavation undercut the lateral support of a tree that was left on a vertical (vs. backsloped) cutbank. The tree later fell, severely injuring a logger. In this article we review current trail construction methods, review safety regulations, and provide practical clearing limits/safety guidelines for trail construction.

Current Construction Methods

The construction of trails on flat or gentle terrain requires little excavation. Trees, vegetation, or other obstacles are simply pushed out of the way with a bulldozer to create a trail that can be negotiated with a skidder, crawler tractor, farm tractor, or forwarder. Construction on steep terrain requires considerable excavation and movement of earth. On such terrain, the trail must be carved out of the hillside, which moves or otherwise displaces obstacles such as soil, rocks, and trees. Most of this material is deposited on the downhill side of the road and the bulk sometimes is buried in the fill or used as a barrier to hold the fill in place (Wenger 1984).

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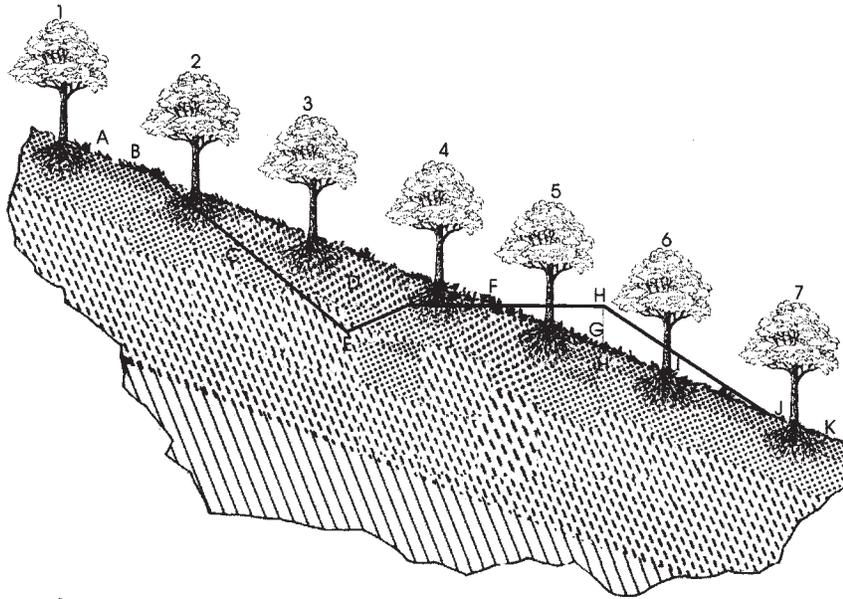


Figure 1.—Cut and fill cross-section of a skid road/trail on a moderate slope: (A): clearing limit; (B): top of backsloped cutbank; (C): cut slope; (D): cut section; (E): ditch; (F): centerline; (G): fill section; (H-H'); outer edge and height of fill; (I): fill slope; (J): toe of fill; (K): downhill clearing limit.

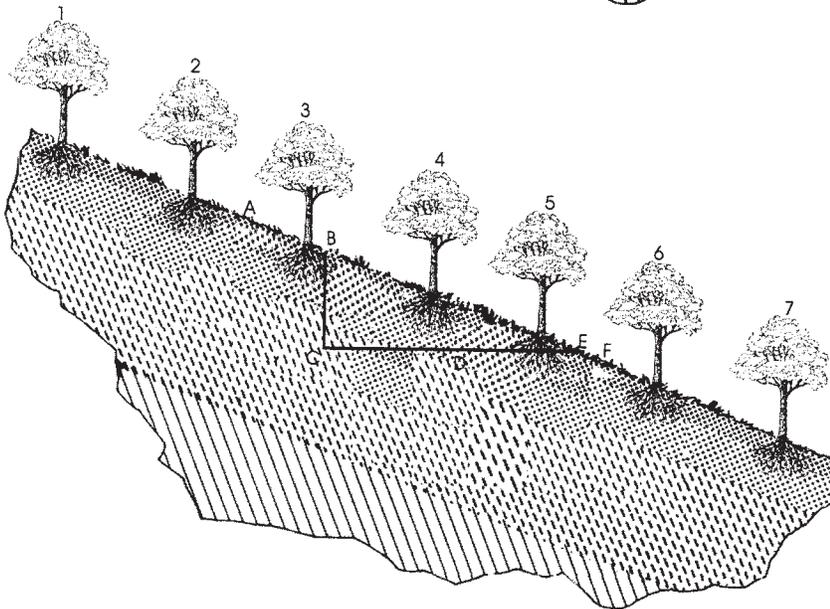


Figure 2.—Vertical cut cross-section of a skid road/trail on a moderate slope: (A): uphill clearing limit; (B): top of vertical cutbank; (C): bottom of vertical cut; (B-C): height of vertical cut; (D): centerline; (E): downhill edge of skid road/trail; (F): downhill clearing limit.

The cut and fill method (Fig. 1, D and G) is used occasionally to minimize the amount of excavation on the uphill side and the movement of soil. Ideally, the volume of earth excavated on the uphill side is used on the downhill side (the fill) to create the running surface of the trail. Balancing the cut and fill reduces the amount of excavation on the uphill side and minimizes the movement of earth, which, in turn, reduces the construction costs. The earth on the fill side must be compacted and free of woody material, e.g., such as stumps, root wads, and other vegetation. This process increases construction costs.

An alternative to cut and fill on low-volume, and low-use skid trails is using vertical cuts to carve the entire running surface of the trail from the uphill side (Fig. 2). The vertical cut method speeds construction, reduces costs, and eliminates the need for back sloping and fill compaction.

Whichever construction method is used, it is imperative that safe clearing limits be set (Fig. 3) as both practices create safety hazards, i.e., trees, vegetation, and other obstacles on the uphill side of the trail (Anonymous 1983; OSHA 2000, 2001). Trees with excavated or disturbed root systems are likely to fall onto the trail and must be removed. These stems can be grubbed out (including the root ball) or felled

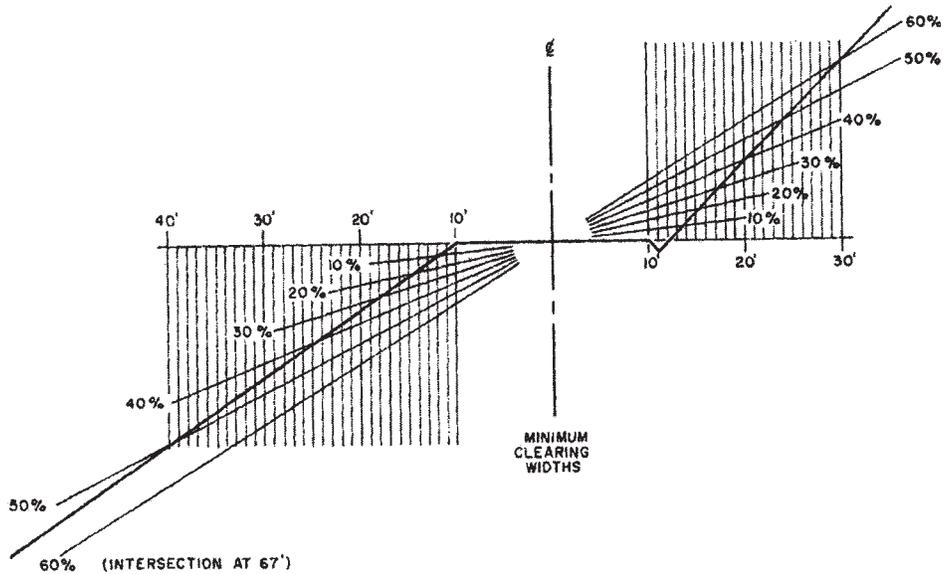


Figure 3.—Cut and fill cross-section of haul road on moderate slope showing uphill and downhill minimum clearing limits for alternative side slopes (from Wenger 1984).

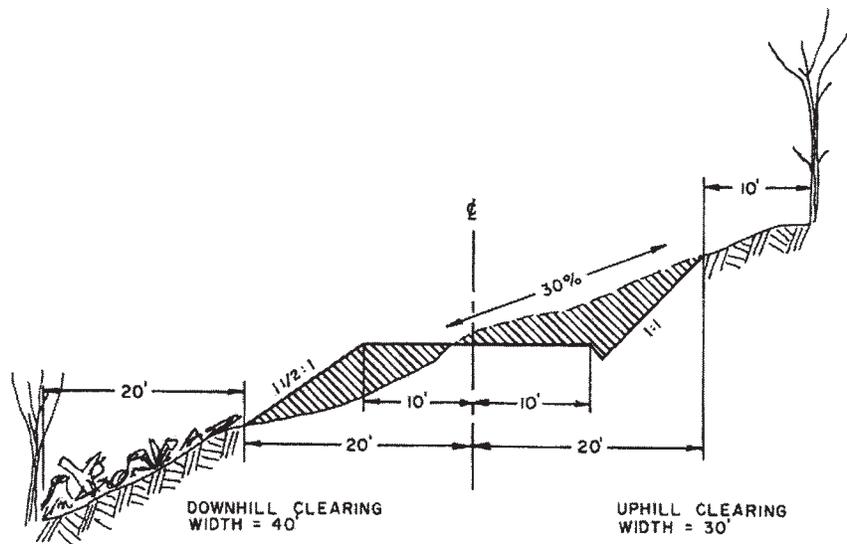


Figure 4.—Cross-section of haul road on moderate slope showing cut and fill areas and clearing limits for cut and fill banks (from Wegner 1984).

depending on their location relative to the uphill side of the cutbank. If circumstances do not allow timely removal, the tree(s) should be flagged and the OSHA Two Tree Length Rule observed until removal.

Practical and Safe Clearing Limits

Few guidelines on clearing limits for forest haul roads (Anonymous 1983; Wenger 1984; OSHA 2001, 2002) are applicable to trails. The general recommendation is that “all snags, danger trees, loose rocks, stumps or other unstable material shall be removed or cleared for a safe distance back from roadsides or roadside banks when they present a hazard to users of roadways”, and that “brush foliage or debris which would obstruct the view of a vehicle operator on roadway intersections or on sharp curves shall be

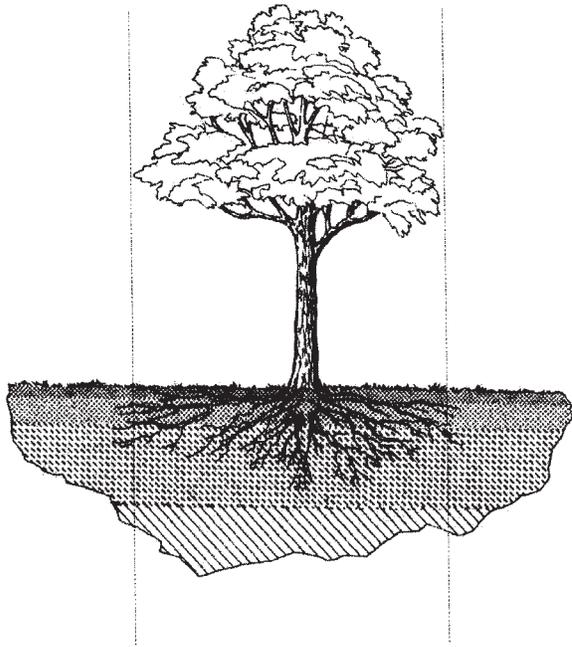


Figure 5.—Isometric scaling proportions of a tree's leaf and stem biomass area to the root biomass area.

cleared and all possible precautions shall be taken to relieve the hazard of such conditions” (Anonymous 1983).

A convenient way to determine minimum clearing limits for haul roads is to plot sideslopes on graph paper and then superimpose the road standards on them (Fig. 3). In most cases, clearing above the centerline should extend at least 10 feet beyond the cut stake position; clearing below the centerline should extend at least 20 feet below the fill stake to allow stumps and other debris to be deposited (Fig. 4). On steep to moderate sideslopes, trees with roots that were undercut during excavation and that are located adjacent to the upper clearing limit or vertical cut should be grubbed out or felled (Pearce and Stenzel 1972).

The following should be considered in judging whether to remove a tree or other obstacle from a cutbank during or following trail excavation:

- Sideslope (%). Steep slopes ($\geq 25\%$) are unstable and require more excavation on the uphill side and more fill on the downhill side. Very steep slopes require full bench roads.
- Soil, solid/loose. Loose rock, etc. moves easily.
- Distance from cutbank edge. The closer the tree/obstacle is to the cut edge, the more likely it will fall.
- Moisture/aspect. Once soils reach the plastic limit, they are nearly liquid in consistency.
- Size of tree/obstacle. The larger the tree/obstacle, the greater amount of mass is on the cutbank.
- Undercut roots/soil. These remove a tree's attachment to the ground, undermining stability.
- Species of tree. Certain species are shallow rooted while others have deep taproots.
- Tree health/vigor. A sound, healthy tree with a full crown always has a strong anchoring mechanism (root system).
- Clearing limits. Should be set and maintained where applicable.

With respect to tree health/soundness and determining clearing limits, Enquist and Niklas (2002) found that a tree's above-ground biomass relative to the leaf and stem area scales nearly isometrically to the root biomass area (Fig. 5). Thus, the outer diameter of the crown also defines the outer diameter of the majority of the root system. This concept is important in creating safe clearing limits for trails. Although some small feeder roots will extend beyond the crown diameter/drip edge (Gilman 1988; Tubbs 1977) and some of these feeder roots may become damaged or disturbed during excavation

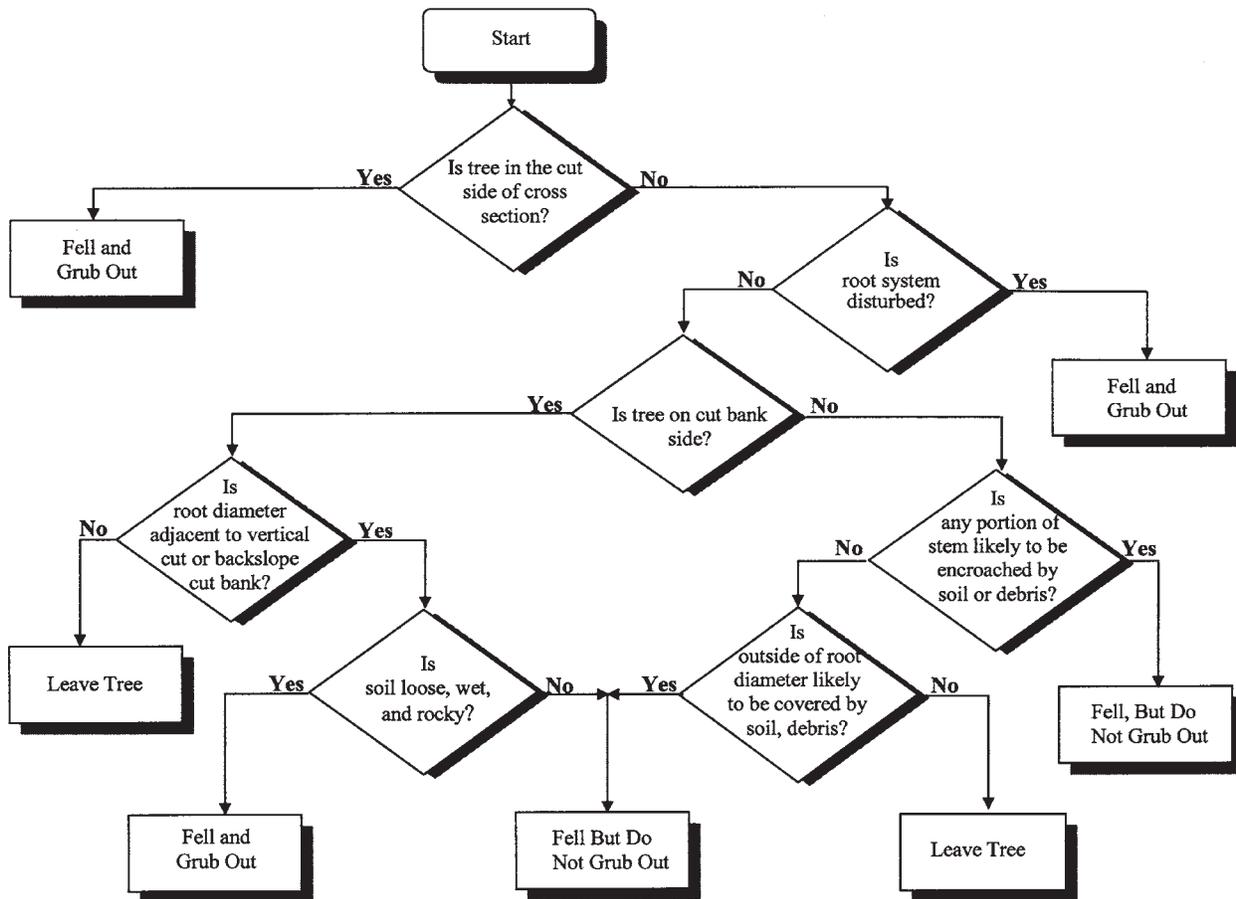


Figure 6.—Decision chart for determining practical and safe clearing limits for skid road/trails with backsloped or vertical cutbanks and fill slopes.

(Stout 1957), the lateral support of the tree will not be affected unless roots well within the crown diameter are damaged/disturbed.

Figure 6 is useful for setting practical and safe clearing limits for trail construction as it addresses decisions on removing trees within and adjacent to the cross section for trails with vertical or backsloped cutbanks. For a cut and fill cross section with the cutbank backsloped (Fig. 1), trees 2, 3 and 4 should be felled and the stump/root wad grubbed out. Trees 5 and 6 should be felled close to the ground and the stump/root wad left in place. Tree 7 should be felled as the toe of the fill will encroach on the root system (see Figure 5) and also might be covered by the stumps and materials from the uphill trees that were grubbed. Tree 1 is safe to leave since its entire root system has not been disturbed during the excavation or backsloping.

For a vertical cut cross-section (Fig. 2), trees 4 and 5 should be felled and the stump/root wad grubbed out. Tree 6 should be felled and the stump/root wad left in place. Tree 3 should be felled because its root system has been disturbed/excavated/exposed and the stump/root wad removed. Some applications may not require that the stump/root wad be removed. For example, the stump/root wad would not be grubbed if the forester/manager in charge determines that felling the tree eliminates the potential hazard and that site conditions (soil type, moisture, height of cutbank, etc) are sufficiently stable to keep the stump/root wad from falling into the trail. Tree 7 should be felled if the earth from the vertical cut excavation would bury any point of the stem or the outside root diameter. Tree 2 is safe as its entire root system is undisturbed (see Figure 5).

Considerations for Managers

The construction/excavation of skid roads/trails creates certain safety hazards that can be eliminated and/or mitigated by observing practical and safe clearing limits during and after trail construction/excavation. The decision chart included in this article should be helpful in determining guidelines that will result in a safer workplace. On a practical level, if there are doubts about a tree's stability, that tree should be removed.

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RELATIONSHIP BETWEEN LOGGING RESIDUE ACCUMULATIONS AND SITE CHARACTERISTICS IN SOUTHERN WEST VIRGINIA

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ABSTRACT.—In West Virginia, substantial volumes of logging byproducts, in the form of unwanted logs, limbwood, and treetops, remain on the ground following timber-harvesting operations. While these residues are ecologically important, their recovery and utilization following harvesting may have tremendous economic potential. We investigated logging residue accumulations after timber harvesting in fire-prone southern West Virginia using line-intersect methodologies. Seventy timber harvests that occurred during 2000-2001 were sampled for logging residue accumulations. The average diameter measured at the intersection with sampling transects for all residue was 7.3 inches. The average overall weight of wood residue left after timber harvest in the 14-county region was 10.4 tons/acre. Oak species had the highest residue accumulations followed by mixed hardwoods and maple species. We found little relationship between distance to landing, distance to market, and average slope with residue accumulations. Relationships between distance to markets and residue accumulations were analyzed for a number of potential residue buyers in the region. Because of their ecological importance, we do not recommend complete use of this resource. However, when left in the woods, logging residues are not converted into useable products, and ultimately, more jobs for the Appalachian region. More emphasis needs to be placed on the further utilization of this resource in West Virginia.

Introduction

Large amounts of logging residues or “slash”, in the form of unwanted logs, limbwood, and treetops, are abundant after timber harvesting in West Virginia (Grushecky et al. 1997). Logging residues contribute to the diversity and cover resources of a logged site (McComb 2003), whereas factors such as piece orientation, size, decay state, species and abundance of coarse woody debris dictate its importance to wildlife (Harmon et al. 1986). Logging residue, or coarse woody debris, is an important component of the habitat requirements for birds (Lohr et al. 2002), amphibians (Butts and McComb 2000), and small mammals (Loeb 1999). Future site productivity can also be impacted by the additions or removals of logging residues. Appalachian forest floor mass has been found to be greater immediately following cutting, and then falls to pre-cut levels within several years (Johnson et al. 1985, Mattson and Swank 1989). Johnson et al. (1985) determined that soil solution concentrations of NH_4^+ -N, NO_3^- -N and K were higher in cut areas harvested with a whole-tree system in Appalachia. In the southern Appalachians, long-term soil cation concentrations were not impacted by commercial sawlog harvests (Knoepp and Swank 1997).

Environmental impacts of logging residue removals must be balanced with economic benefits of their recovery. The utilization of logging residues may have tremendous economic potential in a state that relies heavily on its wood products industry. In 1997, the wood products industry accounted for over \$1 billion of the states total manufacturing output (U.S. Census Bureau 1997). This represents approximately 6 percent of the total manufacturing output in West Virginia, and does not include logging and forest management activities, both large components of the primary manufacturing industry. Within the past decade, several large wood products firms have established manufacturing facilities in West Virginia that use low-grade wood in their composite, or engineered wood products (Luppold et al. 1998). Before this, the market for low-quality roundwood in West Virginia was very limited. Currently, wood contained in logging residues is suitable for use in many engineered wood products, and potentially, for other value-added products in future markets. Efficient recovery of this material is imperative, especially since skidding and trucking have been identified as potential barriers against increased hardwood production in the area (Luppold et al. 1998).

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100 ft. transects) were located in each harvest for a total of 3000 ft. of transect per logged site. One of every four transects was sampled intensively to provide information related to residue characteristics. On intensive lines, measurements included the species and diameter at the transect intersection, as well as the small end diameter (to a minimum of 4 inches), large end diameter, piece length, and potential product (pulpwood versus sawtimber).

To calculate volumes and weights of logging residues, we used equations first developed by Warren and Olsen (1964) and modified by Van Wagner (1968). Van Wagner (1968) gave the following weight equation:

$$W = \frac{11.65S \sum d^2}{L}$$

where W is the weight of logging residue in tons/acre, S is the specific gravity of the logging residue, d is the diameter of logging residue at intersection with transects, and L is the length of the sampling transect. Van Wagner (1968, 1982) describes the development of logging residue equations in eloquent detail.

Average Slope of Harvested Site

Harvest boundaries for each of the 70 sites were obtained from the West Virginia State Division of Forestry notification records. Boundaries were digitized and converted into geo-referenced layers in a geographic information system (GIS) using ArcView 8.3 (ESRI, 380 New York Street Redlands, CA 92373-8100). Likewise, digital elevation models (DEM), in the form of an elevation grid, were incorporated into the GIS for each of the harvest sites. The ArcView 8.3 Spatial Analyst extension was used to calculate the mean slope percentage for each of the harvested tracts based on the clipped DEM for each harvest location, thus limiting slope estimations to the harvested areas.

Distance between Sample Point and Landing

Distances from residue sampling points to log landings were measured to determine the effect of skid distances on material recovery. Layers consisting of UTM coordinates for each residue sample point and landing locations were developed in the project GIS. A coordinate extension available for ArcView 8.3 was used to calculate the distance between each residue sampling point and the log landings. Distances calculated represent a straight line between the residue sampling locations and landings. This measurement represents skid turn distance well because in most cases, as the straight line distance between the landing and sample point locations increases, so does actual skidding distance.

Distance Between Harvest Sites and Engineered Wood Products Markets

Using the GIS, we calculated the total distance between the main log landing on each harvested site to three main users of woody residues in West Virginia; the Weyerhaeuser oriented strandboard (OSB) plant in Heaters, WV, the Georgia Pacific OSB facility in Mt. Hope, WV, and the TrusJoist Parallel Strand Lumber and Laminated Strand Lumber facility in Buckhannon, WV. We chose these facilities because they are an end market for the majority of low quality roundwood produced during timber harvests in West Virginia. For each harvested site, we attempted to calculate 2 alternative routes to each market. Routes were chosen based on the shortest distance suitable for tractor-trailers hauling residue. Routes were measured by digitizing roads between landings and markets using a distance-measuring tool in the GIS.

Results

We sampled 2100 transects in southern West Virginia during the summer of 2002 and measured a total of 5796 pieces of logging residue. This represents approximately 40 miles of logging residue transects. The most prevalent species of logging residue was yellow-poplar (*Liriodendron tulipifera*), representing 14.5 percent of the residue crossed by transects. Yellow-poplar was followed by red oak (*Quercus rubra*) and white oak (*Quercus alba*), which accounted for 13.3 percent and 10.2 percent of the number of residue pieces, respectively.

Table 1.—Average diameter of species at intersection with residue transects. Listed along with the average diameters and their respective standard deviations (in parentheses) are the average weight, average small end diameter, average large end diameter, and average length of logging residue sampled in southern West Virginia.

Species ^a	Mean Diameter	Weight (Tons/ac)	Small End Diameter	Large End Diameter	Length
Cherry	6.4 (2.7)	0.1 (0.2)	5.3 (2.4)	8.1 (3.1)	19.1 (11.9)
Maple	6.4 (2.6)	1.2 (1.0)	4.5 (1.3)	7.4 (2.7)	20.0 (11.9)
Misc. Hardwoods	7.3 (3.5)	2.5 (2.0)	5.0 (2.3)	9.3 (4.4)	21.1 (12.0)
Oak	7.7 (3.6)	5.0 (3.1)	4.8 (2.0)	9.6 (3.9)	20.1 (10.6)
Soft hardwoods	7.9 (3.7)	0.2 (0.4)	4.4 (1.2)	8.6 (3.4)	19.0 (10.0)
Softwoods	7.2 (3.1)	0.1 (0.3)	4.8 (1.5)	9.1 (3.8)	21.5 (14.2)
Yellow-poplar	7.2 (3.0)	1.2 (1.0)	5.2 (2.0)	9.1 (3.0)	20.3 (12.1)

^aCherry - black cherry (*Prunus serotina*); Maple - red maple (*Acer rubrum*), sugar maple (*Acer saccharum*); Oak - red oak (*Quercus rubra*), black oak (*Quercus velutina*), scarlet oak (*Quercus coccinea*), white oak (*Quercus alba*), chestnut oak (*Quercus prinus*), American chestnut (*Castanea dentata*); Soft Hardwood - aspen (*Populus spp.*), basswood (*Tilia americana*), cucumber (*Magnolia acuminata*), sycamore (*Platanus occidentalis*), tree-of-heaven (*Ailanthus altissima*); Softwood - eastern hemlock (*Tsuga canadensis*), pine (*Pinus spp.*); Yellow-poplar - yellow-poplar (*Liriodendron tulipifera*); Miscellaneous Hardwoods - American elm (*Ulmus americana*), black birch (*Betula lenta*), American beech (*Fagus grandifolia*), black gum (*Nyssa sylvatica*), black locust (*Robinia pseudoacacia*), yellow buckeye (*Aesculus octandra*), American dogwood (*Cornus florida*), fraser magnolia (*Magnolia fraseri*), hickory spp. (*Carya spp.*), sassafras (*Sassafras albidum*), sourwood (*Oxydendrum arboreum*), white ash (*Fraxinus americana*), yellow-birch (*Betula alleghaniensis*).

The average diameter measured at the intersection with sampling transects for all residue was 7.3 inches. The corresponding maximum diameter was 34 inches. To facilitate result interpretations, residue was broken down into 7 groups based on species attributes and marketability (table 1). The soft hardwood species group had the largest diameters at intersection (7.9 inches), but the second lowest frequency (77 pieces measured). Oak species had the second largest diameter at intersection (7.7 inches) and the largest frequency (2117 pieces measured) (table 1). The average overall weight of wood residue left after harvest in the 14-county region was 10.4 tons/acre. The minimum tonnage found among the 70 harvest sites visited was 2.3 tons/acre and the maximum was 20.3 tons/acre. The highest residue volumes were found in the central portion of our study area (fig. 1).

Oak was the most prevalent species group by weight, averaging 5.0 tons/acre over the study area (table 1). Miscellaneous hardwoods and maple species followed the oaks with 2.5 and 1.2 tons/acre respectively. Black cherry had the lowest average weight over the study area at 0.1 tons/acre remaining after harvest. The maximum amount of residue left of any one species was 15.1 tons/acre for oaks. Oaks, followed by miscellaneous hardwoods, had the largest accumulations of residue in the study area. The dominance of oak as logging residue in the study area is further emphasized when compared to the next three highest species in terms of total weight, miscellaneous hardwoods, maple species, and yellow-poplar (fig. 2).

On intensive transect lines; large and small-end diameters and total lengths were recorded so that residue piece size could be characterized. A total of 1305 pieces of logging residue were measured on intensive lines. The average large-end and small-end diameter of the pieces measured on intensive lines was 9.2 and 4.9 inches, respectively. The average length of all logging residue was 20.4 feet.

Oak species had the largest average large-end diameter of 9.6 inches. Miscellaneous hardwoods followed oak with an average large-end diameter of 9.3 inches. Yellow-poplar followed oaks and miscellaneous hardwoods with an average large-end diameter of 9.1 inches. Overall lengths were similar by species (table 1). Softwoods and miscellaneous hardwoods had the longest average lengths of 21.5 and 21.1 feet, respectively. These were followed by yellow-poplar residue that had an average length of 20.3 feet.

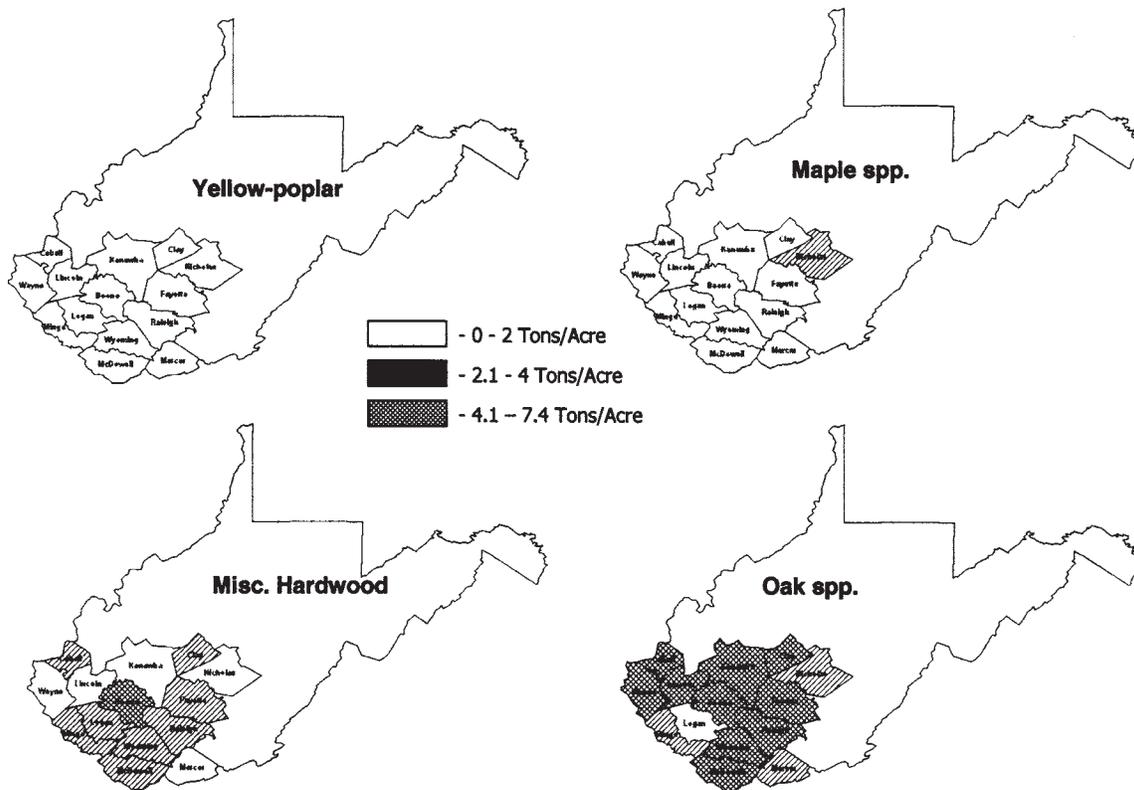


Figure 2.—Average logging residue weights for the top 4 species groups found during a 2002 inventory of wood residues in southern West Virginia.

Logging residue piece size was used to calculate the total amount of “marketable” residue available in the study area. This was based on determining the total volume of residue that met a 12-foot length criterion, which is a common specification in the engineered wood products industry in West Virginia. Once volume was calculated that met length specifications, total residue available was determined by multiplying the total weight available per acre by the average acres harvested in each county per year. Harvest data were obtained from notification forms supplied by the West Virginia State Division of Forestry. A total of 810,584 tons of residue were available during the study period in the 14-county region (table 2). When the residue loads were further broken down by species acceptable for the engineered wood products industries in West Virginia (Weyerhaeuser, Georgia Pacific, TrusJoist), approximately 332,649 tons of logging residue were available during the sampling period in the 14-county area.

The average slope of harvested areas sampled in southern West Virginia was 30.8 percent (± 9.3 percent). We did not find any evidence that slope had any influence on total residue accumulations remaining after harvest ($r=-0.09$, $P=0.43$). The average distance between residue sampling points and the log landings of harvests measured in southern West Virginia was 979.8 feet (± 490.5 feet). We did not find a relationship among distances from landings to the location of our sample points ($r=-0.13$, $P=0.28$).

The average distance to an engineered wood product facility from the harvest locations sampled in southern West Virginia was 108.9 miles (± 42.5 miles). The Truss Joist facility was farthest, on average, from the logged sites sampled at 153.2 miles (± 60.8 miles). The Georgia Pacific Oriented Strand Board plant was the closest, on average, to the logged sites sampled at 68.5 miles (± 31 miles). The Weyerhaeuser Oriented Strand Board plant averaged 105.2 miles (± 35.9 miles) from the logged areas sampled. We found no significant correlations among the distances from harvest landings on those sites sampled and the Engineered Wood Products facilities in West Virginia.

Table 2.—Total residue weight (tons/acre) found remaining after harvest in each of 14 counties sampled in southern West Virginia. Total residue available was calculated using the percentage of the total volume that met a 12 foot length specification. Acres harvested are average of years 1992-2001, obtained from West Virginia State Division of Forestry Logging Notification Forms.

County	Total Weight (Tons/Acre)	Percent Length Specification	Total Avail/Acre (Tons/Acre)	Acres Harvested	Total Tons/Year
Boone	12.7	0.87	11.0	9342	102,762
Cabell	11.7	0.75	8.8	1965	17,292
Clay	11	0.9	9.9	6447	63,825
Fayette	11	0.79	8.7	9169	79,770
Kanawha	11.3	0.84	9.5	13712	130,264
Lincoln	9	0.71	6.4	3220	20,608
Logan	7.1	0.7	5.0	6860	34,300
McDowell	10.8	0.89	9.6	9223	88,541
Mercer	7.8	0.83	6.5	2606	16,939
Mingo	10.1	0.88	8.9	4101	36,499
Nicholas	8.2	0.53	4.3	9363	40,261
Raleigh	11.7	0.83	9.7	9755	94,624
Wayne	10.5	0.78	8.2	2057	16,867
Wyoming	12	0.82	9.8	6942	68,032
Totals	144.9	11.12	116.3	94762	810,584

Discussion

The quantity of logging residues remaining after timber harvesting likely varies according to the intensity of the harvest operation and the available markets. There continues to be a substantial amount of woody residue remaining on site after timber harvesting in West Virginia. In southern West Virginia, Grushecky et al. (1997) found from 6.2 to 15 tons/acre of logging residues remaining after harvests that occurred in 1993-1994. With this research project, we found a similar amount of residue at 10.4 tons/acre in southern West Virginia. By species, logging residue breakdowns also were similar. In the 1993-1994 period, oaks were the prevalent residue, followed by mixed hardwoods and yellow-poplar (Grushecky et al. 1997).

Economic factors are likely the main determinant of the composition of logging residues remaining after harvest. The lower the value of the logging residues, the more likely it will be left on site. Efficient recovery of logging residues must occur at the time of harvest to remain profitable. Many of the factors that cause useable residue pieces to be left in the woods interact with one another. For example, skidding efficiency, site visibility, potential markets, and material quality all work together to generate the residue levels observed following timber harvesting in the region. Skidding and trucking have been cited as barriers to loggers wanting to increase production, especially in southern West Virginia (Luppold et al. 1998). We found little evidence that distance to markets, skid distances, and/or the average slope of harvested sites affected residue concentrations. Because of the high degree of variability in harvesting, markets, and site characteristics, there is a correspondingly high variation associated with residue weight estimates. Therefore, variation in residue loads hindered our efforts at discovering relationships among site characteristics and logging residue volumes. However, since harvests were chosen randomly for this project, we assume that the residue loads reported are a representative sample of all harvests in southern West Virginia.

Although no relationship was found among site factors and logging residue levels, the logging residue resource characteristics found through this study are important. If all species of logging residues are considered, there was enough logging residue to run an engineered wood production facility a total of 406 days, assuming the use of 2000 tons per day. Thus, this region alone could support an additional engineered wood products facility or a biomass conversion refinery, if they could make use of all residue species. This is especially true if the ring porous hardwoods such as oaks (*Quercus* spp.) could be better utilized. Oak is highly sought after for dimensional lumber, yet most engineered wood products

facilities in West Virginia prefer only small quantities of oak because of its undesirable physical properties for composite manufacturing. From a utilization standpoint, this is unfortunate as there is about 2.4 times the amount of residue at least 12 feet long when oak is included in average residue loading calculations. More emphasis needs to be placed on the use of lower quality, smaller oak in the form of topwood, pulpwood and unwanted logs. The feasibility of using this unwanted oak residue for value added products such as flooring, or components for the furniture and cabinet manufacturing industry, was addressed in the late 1970s and early 1980s (Reynolds and Gatchell 1979, Reynolds et al. 1983, Hansen and Araman 1985). These approaches or new technologies that allow for the co-firing of power plants or bio-refineries that use wood biomass need to be evaluated under current market conditions, especially since recent policies propose decreasing the United States dependence on foreign fuels.

When left in the woods, logging residues are not converted into useable products, and ultimately, jobs for West Virginians. However, logging residues remain an important component of Appalachian forest ecosystem. For this reason, we do not recommend complete utilization, but we should strive to use a greater portion of this valuable resource. While we typically think of the larger firms as creating the main demand for wood residue, other opportunities may exist for composite or solid fuel products and should be considered when developing marketing plans for these residues.

Acknowledgments

This research was supported, in part, by the USDA Forest Service, Northeastern Area State and Private Forestry award number 01-DG-11244225-305. The authors would like to thank students Steve Cooper and Michael Parsons, from Glenville State College, and Jason Grose from the WVU Division of Forestry, for their dedicated field work under demanding conditions. The authors would also like to thank the efforts of two anonymous reviewers. This manuscript is published with the approval of the Director, West Virginia Agriculture and Forestry Experiment Station as Scientific Article No. 2860.

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CHANGES IN FUEL LOADING AS THE RESULT OF REPEATED PRESCRIBED FIRES WITHIN THE OZARK FORESTS OF MISSOURI

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ABSTRACT.— Buildups of wildland fuels have created problems and concern across the nation. The central hardwoods region has little information published on the fuel loading occurring within stands of oak-hickory and oak-pine forest. Fuel was sampled on a 2,500-acre restoration project located in the Missouri Ozarks. Samples were taken before dormant season fires during the years 1998 through 2003. Samples were stratified by aspect and by number of fires during the 5-year period. Pretreatment fuel loading ranged from 6.6 tons/acre in intermittent drainages to 10.9 tons/acre on ridges (neutral slopes). After 5 years and either 2, 3 or 5 burns, total fuel loads were not changed greatly, but the composition of the fuel bed was altered in all samples. Woody fuels in the 1-, 10-, and 100-hour size classes increased as herbaceous fuels (litter) decreased. Restoring a fire regime to these natural communities may require years to reach the desired state of “naturalness” with corresponding changes in fuel load tonnage and composition.

Natural cover fires have been part of the Ozark landscape for centuries (Guyette 1982). With the success of fire suppression programs by the US Forest Service and Missouri Department of Conservation initiated in the 1930's, the frequency of fires on the landscape has decreased greatly (Westin 1992). The reduction or cessation of anthropogenic wildfires has resulted in altered plant communities with altered structures and species and a concurrent buildup in forest floor fuels (Beilman and Brenner 1951). Little is known about the fuel loading of forested lands within the central hardwoods, either in terms of existing fuel loads or the changes in these fuel loads due to prescribed burning. Here I report changes in fuel loading following repeated prescribed fire from 1998 through 2003 in the Missouri Ozarks.

Study Area

The study site is a 2,500-acre block of oak-hickory and oak-pine Ozark woodlands located in Shannon and Carter Counties, Missouri (fig. 1). The study site is in the central portion of the 5,657-acre Chilton Creek Management Area owned and managed by The Nature Conservancy (TNC) and comprises the upper Chilton Creek Basin. The study site lies within the Current River Hills subsection of the Ozark Highlands (Keys and Carpenter 1995), an area characterized by rugged, steeply dissected valleys and hollows, and narrow ridges. Soils are primarily Alfisols and Ultisols, well drained and developed over dolomite and sandstone karst bedrock.

The goal of management on the Chilton Creek Management Area is to conserve the native biota in a mosaic of high quality native habitats. The sole management treatment is prescribed fire.

The treatment area is divided into five units (fig. 1) to facilitate fire management and to evaluate the role of varying frequencies in restoring the native mosaic on the landscape. One unit, Kelly North, was burned annually. The other units were burned at random, one- to four-year fire return intervals (table 1). All units were burned during the first year of treatment, 1998.

This portion of the Ozarks has a long fire history, primarily of anthropogenic origin. The mean fire interval for this area was about 7.1 years before Euro-American settlement (1820) and 2.2 years between 1820 and the late 1920's (Guyette 1997). The wildfire record indicates no major fires on the study site since the drought years of the mid-1950's when a major fire burned across most of northern Carter County to the west bank of Current River east of the study area. There was no prescribed burning on Chilton Creek prior to initiation of this study.

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Chilton Creek Study Area

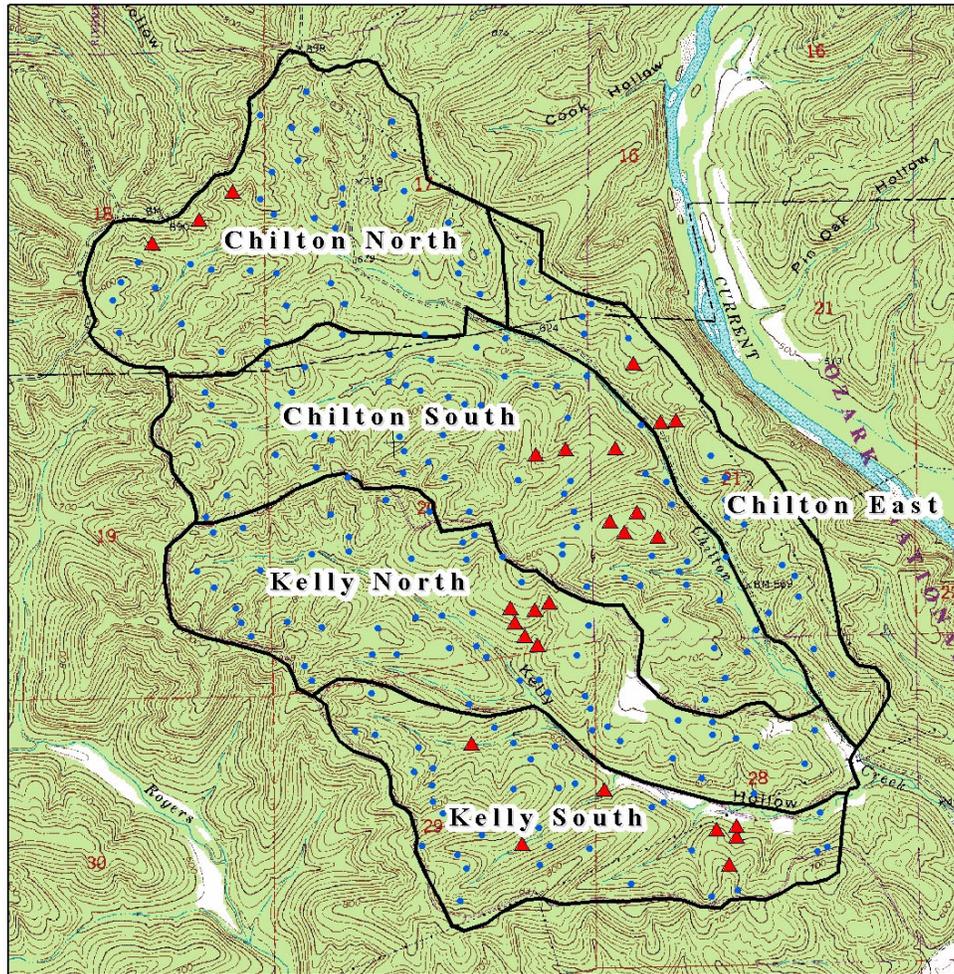


Figure 1.—Chilton Creek Study Area and the five fire management units (labeled). The Kelly North Unit was burned annually; the other units were burned at random, one-to four-year intervals (see Table 1). The twenty-six fuel plots were selected from the 250 permanent vegetation plots based on accessibility.

Table 1.—Units burned on the Chilton Creek Study Area, 1998-2002.

Fire Unit	1998	1999	2000	2001	2002
Kelly South	X		X	X	
Kelly North	X	X	X	X	X
Chilton South	X	X	X		
Chilton North	X				X
Chilton East	X				X

Methods

In 1996, 250 ½-acre plots were established to record changes in herbaceous and woody vegetation across the project in response to the fires. The study area was stratified by Ecological Land Type (ELT) based on slope, aspect, topographic position and geologic substrate (Miller 1981) prior to establishing plots. Twenty-six plots were selected for more intensive monitoring of fuels, fire behavior and differential mortality of small woody stems (Dey and Hartman 2003). Plot selection included all aspects (North, South, Ridge, Intermittent Drainage) and all burning frequencies (2, 3, or 5 burns). Due to the nature of the land treatment, rugged topography and safety issues, sites were selected in groups that could be accessed by ATV for equipment placement and removal and could be entered and exited safely by the observation crew during the prescribed burning activity. The 26 plot locations are shown on Figure 1.

Sampling was conducted on all 26 plots one month prior to the 1998 and 2003 prescribed burns. Two 50-foot down, dead woody (ddw) transects were established in cardinal directions with transect centers at the vegetation sampling plot center. Ddw transects segregated woody fuels into 1-, 10-, 100- and 1,000-hour timelag fuels. Ddw transects were sampled following the system described by Brown (1974).

Mean litter depth was recorded at 5-foot intervals along each transect. Litter and herbaceous vegetation were sampled within 1- by 2- foot quadrats at each end of each transect. Exact location of the quadrats was shifted at each sampling due to the reduction of litter within the quadrat when sampled. Litter/herbaceous samples were dried at 60° C for 24 hours to produce oven dry weights. This sampling scheme yielded a total of 52 transects, 104 herbaceous samples and 520 litter depth measurements per sampling year. Results were averaged and mean values per plot were analyzed.

Results

Results from the sampling indicate there may be changes in fuel loading with both aspect and frequency of burns, but variability within the samples was too great to draw any statistically valid conclusions. Therefore, the mean fuel weights of the sample plots on all units (n=26) were combined and analyzed and the results are shown here with examples of changes by fire frequency and aspect.

Initial mean fuel loads, recorded in 1998 before the first burn, ranged from a high of 10.9 (± 4.0) tons per acre on ridges to a low of 6.6 (± 2.4) tons per acre on intermittent drainages (table 2). North slopes had 8.6 (± 1.4) tons and south slopes 9.4 (± 1.6) tons per acre. Herbaceous fuel (primarily leaf litter) composed 37 percent of the total fuel load. Fuels that contributed most to fire intensity (herbaceous, 1-, and 10-hr fuels) comprised 49 percent of the preburn samples by weight.

After 5 years, mean total fuel loads ranged from 9.7 (± 2.3) tons per acre on south slopes to 6.4 (± 1.8) tons per acre on intermittent drainages. Ridge plots contained 7.7 (± 2.3) tons per acre and north aspect plots contained 7.0 (± 1.3) tons per acre. Herbaceous fuel (still primarily leaf litter) was reduced to 25 percent of the total fuel load. Light fuels (herbaceous, 1-, and 10-hr) were reduced slightly to 43 percent of the total.

Initial mean litter depth was 1.8 inches (table 2). This parameter was reduced to 1.6 inches by 2003. The structure of the litter was changed from a matted layer to a less dense, fluffy layer more conducive to combustion. Litter also became more irregular in distribution, as indicated by twice as many sample points having bare ground after the burns.

The only fuel component to be significantly reduced (95 percent Confidence Interval on the mean) from pre- to post-treatment was the herbaceous fuel load (fig. 2). Significant increases were detected in both the 1- and 10-hour fuel load components. Trends within the 100-hr fuels were higher and 1,000-hr fuel loads trended lower. The only aspect to indicate an increase in 1,000-hr fuels was South slopes, which typically burn hotter than other exposures. Increased fire intensity is expected to increase mortality on larger standing stems. The mean 1,000-hr fuel weight on south slopes increased from 4.1 (± 1.6) to 5.5 (± 2.0) tons/acre.

Table 2.—Chilton Creek fuels summary - pre and post burn

Aspect	Year	n	Fuel Weights						Total (se ¹)	Litter Depth - inches -
			1-hr (se ¹)	10-hr (se ¹)	100-hr (se ¹)	1000-hr (se ¹)	Herb (se ¹)	tons per acre		
South	1998	12	0.2 (0.02)	0.8 (0.11)	1.1 (0.32)	4.1 (1.61)	3.5 (0.19)	9.7 (1.68)	2.0	
North	1998	7	0.3 (0.04)	1.4 (0.24)	1.2 (0.48)	2.4 (1.35)	3.3 (0.19)	8.6 (1.37)	1.6	
Ridge	1998	4	0.2 (0.03)	0.6 (0.13)	0.5 (0.32)	5.8 (4.11)	3.8 (0.34)	10.9 (4.00)	2.0	
Int Dr	1998	3	0.2 (0.04)	0.5 (0.07)	0.0 (0.00)	3.0 (2.26)	3.0 (0.31)	6.6 (2.35)	1.3	
Preburn	1998	26	0.2 (0.02)	0.9 (0.09)	0.9 (0.21)	3.8 (1.05)	3.4 (0.12)	9.2 (1.07)	1.8	
South	2003	12	0.3 (0.03)	1.2 (0.16)	1.1 (0.46)	5.5 (1.98)	1.6 (0.09)	9.7 (2.26)	1.4	
North	2003	7	0.3 (0.02)	1.4 (0.26)	1.2 (0.71)	1.7 (0.73)	2.5 (0.18)	7.0 (1.29)	1.9	
Ridge	2003	4	0.4 (0.04)	1.8 (0.43)	0.3 (0.26)	3.0 (2.52)	2.4 (0.12)	7.7 (2.33)	1.6	
Int Dr	2003	3	0.4 (0.13)	1.0 (0.35)	2.6 (1.73)	0.2 (0.23)	2.1 (0.26)	6.4 (1.80)	1.6	
Postburns	2003	26	0.3 (0.02)	1.3 (0.13)	1.2 (0.35)	3.5 (1.03)	2.0 (0.09)	8.3 (1.17)	1.6	

¹se = standard error

²Int Dr = Intermittent Drainage

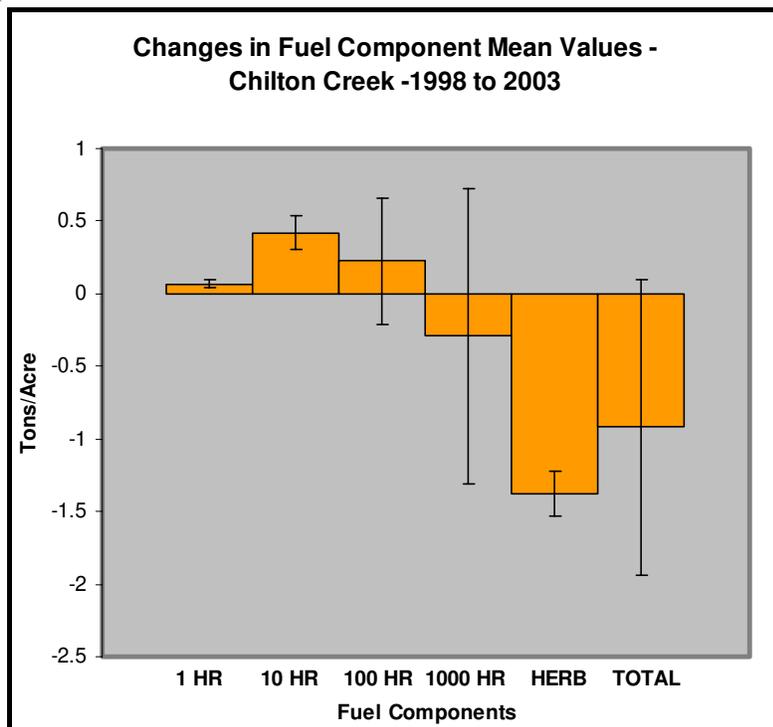


Figure 2.—Changes in Fuel Component Mean Values from 1998 to 2003 at the Chilton Creek Study Area. Error bars are 95 percent confidence intervals.

Discussion

Fuel loads prior to treatment were greater than those reported for unburned and unharvested Ozark woodlands by Kolacks and others (2003), 9.2 versus 7.2 tons per acre. Herbaceous litter plus 1-hr woody debris weights within 40-year-old, unburned stands sampled in April by Loomis (1975) yielded 5.3 tons per acre. Paulsell (1957) found in his January sampling of an oak-hickory flatwoods undisturbed for 15 years that “litter” (undefined) was 7.5 tons per acre in 1949. The same tract in 1957, still undisturbed, yielded 6.1 tons of leafy and herbaceous material and 0.6 tons of woody material, considerably greater than the 3.7 tons of herbaceous and 1-hr fuels found on this study and the 3.0 tons reported by Kolacks and others (2003). Inclusion of larger diameter woody fuels in the earlier calculations of “litter” could account for the increased weights.

The only statistically significant reduction in the fuel loadings was the reduced weight of herbaceous fuel on the plots. Herbaceous fuel, leaf litter in this case, is the most volatile fuel component and carries the fire until the woody fuels can be preheated sufficiently to ignite. In a landscape that is subjected to multiple fires after a long fire free interval, the finest fuels would be expected to be reduced with repeated burns.

In an unburned forested system with a developed midstory and woody understory, increased dead woody fuel loadings would be the expected result from mortality of the smaller woody stems the first years following initiation of fire management. The mean 1000-hr component across the study area was reduced numerically, but the 1-, 10- and 100-hr woody fuel components were increased by 25 percent, 39 percent, and 26 percent respectively (fig. 2). The same general trend was recorded by Paulsell (1957) following repeated burns on an annual and a 4-year fire return cycle. These smaller diameter woody fuels should add to the fire intensity of future burns and may enhance the effectiveness of the burns as the landscape returns to a fire induced state of openness.

The Chilton Creek Management Project is a long term effort by The Nature Conservancy and The Missouri Department of Conservation. Tracking of fuels and vegetative changes over time will document the continued progression as the landscape moves toward the desired future condition of a dynamic, fire regulated Ozark landscape.

Acknowledgments

This and companion vegetative fire effects studies (Dey and Hartman 2003, Sasseen 2003, Hartman and Heumann 2003) are the result of coordination and support by both The Nature Conservancy and the Missouri Department of Conservation. Documentation of long term fire effects will provide insight into fire management of habitats and biotic diversity within the Ozark Highlands of Missouri.

I also gratefully acknowledge the help and support of Dr. Russ Reidinger and his staff at Lincoln University, Jefferson City MO, for utilizing the fuel sample drying as an opportunity to train students in basic scientific method. Their efforts over a 6-year period provided a quality set of data.

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THE EFFECT OF THINNING AND PRESCRIBED FIRE ON FUEL LOADING IN THE CENTRAL HARDWOOD REGION OF MISSOURI

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ABSTRACT.—We collected fuel loading data in the Southeast Missouri Ozarks to determine if aspect (north and east facing slopes (protected), south and west facing slopes (exposed), and no slope (ridge)) has an effect on fuel loading in stands that received either thinning, prescribed fire, both prescribed fire and thinning, or no management (control). Aspect affected fuel loading in several categories for pre-treatment, post-thinning, and post burn-thinning treatments. The general pattern was a progression from exposed slopes, < ridges, < protected slopes. Thinning increased total fuel loading about 300 percent with 100 and 1000-hr solid fuels replacing litter as the heaviest component of the total. The prescribed burn did not significantly consume 100 and 1000-hour fuels. Burning alone and burning in stands that had recently been thinned resulted in a 50 and 25 percent reduction in total fuel loading, respectively. In both treatments a majority of consumption occurred in fuels < 3 inches with litter being nearly 100 percent consumed on all aspects in both burn treatments. Horizontal continuity was disrupted in both cases making reburn very unlikely. However, a fire could carry through the stands soon after the first leaf fall.

Very little information is available on fuel loading, let alone the effects of harvesting or fire on fuel loading, in the Central Hardwood Region of the United States. A recent literature search for such information yielded very few references that addressed the subject. Most studies were completed before the development of modern fuel sampling techniques and timelag classes (Brown 1974). Paulsell (1957), Scowcroft (1965), Crosby and Loomis (1974), and Loomis (1975) all evaluated fuel loading in the Missouri Ozarks before the development of modern techniques. Crosby and Loomis (1967) and Loomis and Crosby (1968, 1970) evaluated the effect of thinning on fuel loading and the contribution of hardwood fuels after aerial herbicide application in pine stands located in southeast Missouri, also before the development of modern techniques.

Anderson (1982) developed fuel loading models using modern techniques which are used by the fire behavior prediction model BEHAVE. These models including general fuel loading values for oak-hickory (*Quercus-Carya*) leaf litter and hardwood slash in the eastern United States. Ottmar and Vihnanek (1999) developed a stereo photo series for quantifying natural fuels in mixed oak types located in the central states. However, reported fuel loadings are only representative of the area that falls within the view of the camera and no stereo photos were developed for the portion of the Central Hardwood Region that extends west of the Mississippi River.

It is known that fuel loading and vegetative structure have been altered from pre-settlement conditions and that they are still changing. Most accounts of the Missouri Ozarks prior to settlement describe open woodlands with little to no underbrush (Ladd 1991, Nigh 1992). This open forest structure was the result of an anthropogenic fire regime, dominated by light surface fires, maintained by Native Americans. The mean fire-free interval (MFI) during the Native American depopulated (1580-1700) and repopulated (1701-1820) periods were 17.7 and 12.4 years, respectively (Guyette and Cutter 1997, Guyette and Dey 1997).

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Beginning in the early to mid 1800's, settlement began to have a substantial effect on the forest structure, fire regime, and fuel loading of the Missouri Ozarks. The MFI decreased to 3.7 years during the period of Euro-American settlement (1821-1940) (Guyette and Cutter 1997). A majority of the Ozarks was completely cutover in late 1800's and early 1900's to feed America's westward progression, and was conducted with little regard for regeneration (Cunningham and Hauser 1992, Nigh 1992, Guyette and Dey 1997, Guyette et al. 1999). Frequent fires and accumulations of slash, which resulted in intense fires, killed pine regeneration already in place and stimulated oak sprouting. This altered disturbance regime undoubtedly had an effect on pine regeneration and recruitment (Cunningham and Hauser 1992, Guyette and Dey 1997).

The composition of present day Ozark forests is much different from that of pre-settlement composition. The second-growth forests are more dense and contain less pine than historic stands with hardwoods completely replacing pine in many places (Cunningham and Hauser 1992, Nigh 1992, Guyette and Dey 1997, Nigh et al. 2000). A 66-percent reduction in relative abundance of pine from historic levels (circa 1900) has been noted in some areas along with a reduction in range from an estimated 6.6 million acres to only about 400,000 acres of pine and oak pine forest in 1976 (Cunningham and Hauser 1992, Guyette and Dey 1997). Fire suppression began in the 1930's increasing the calculated fire rotation length during the period 1970-1989 to about 326 years statewide (Westin 1992). In an adjacent area, the Arkansas Ozarks, the MFI was greater than 80 years for the period 1921-2000 (Guyette and Spetich 2003). There is evidence that fire exclusion, changing stand structures, and timber harvest have increased fuel loadings (Guyette 1999). Suppression created favorable conditions for the development of dense oak forest and allowed fuel loading to increase, unchecked by periodic fire and controlled only by decomposition.

Today federal and state agencies, as well as private organizations and individuals, are using thinning, harvesting, prescribed fire, or combinations of these treatments as management tools. In 2002 the Missouri Department of Conservation alone mechanically treated almost 5,000 acres with about 2,800 of those acres receiving pre-commercial thinning and intermediate, unevenage, and shelterwood harvest (Anonymous 2002). Also in 2002, the same agencies and organizations applied prescribed fire to more than 60,000 acres throughout Missouri. However, the effects of management activities on fuels are poorly understood. There is a need to better understand these effects since these treatments are often used for restoration of habitat and biodiversity.

We collected fuel loading data in southeast Missouri as part of a cooperative study funded by the Joint Fire Science Program. The purpose this study is to determine existing fuel loads and whether aspect (exposed, ridge, and protected) has an effect on fuel loading in stands that received thinning, prescribed fire, both thinning and prescribed fire, or no management (control). This study is the most ambitious of its kind in this area to date.

Study Area

The study area is located in the southeastern Missouri Ozarks near Ellington, MO on land managed by the Missouri Department of Conservation (Figure 1). In an effort to minimize variation caused by potential vegetative differences, study sites were all installed within the Black River Oak/Pine Woodland/Forest Hills Landtype Association utilizing the Missouri Ecological Classification System, which utilizes the US Forest Service National Hierarchy of Ecological Units for landscape classification (Meinert et al. 1997, Nigh and Shroeder 2002).

This LTA is characterized by strongly rolling to hilly lands with steep slopes with flat land found only in creek and river bottoms. Historically oak and oak-pine woodlands and forest comprised the area. These forest types still dominate, however, woodlands are second growth and have grown more closed due to fire suppression (Nigh and Shroeder 2002). The Black River Hills LTA is in the center of one of the largest blocks of forest in the Midwest which also supports a substantial timber industry (Nigh et al. 2000).

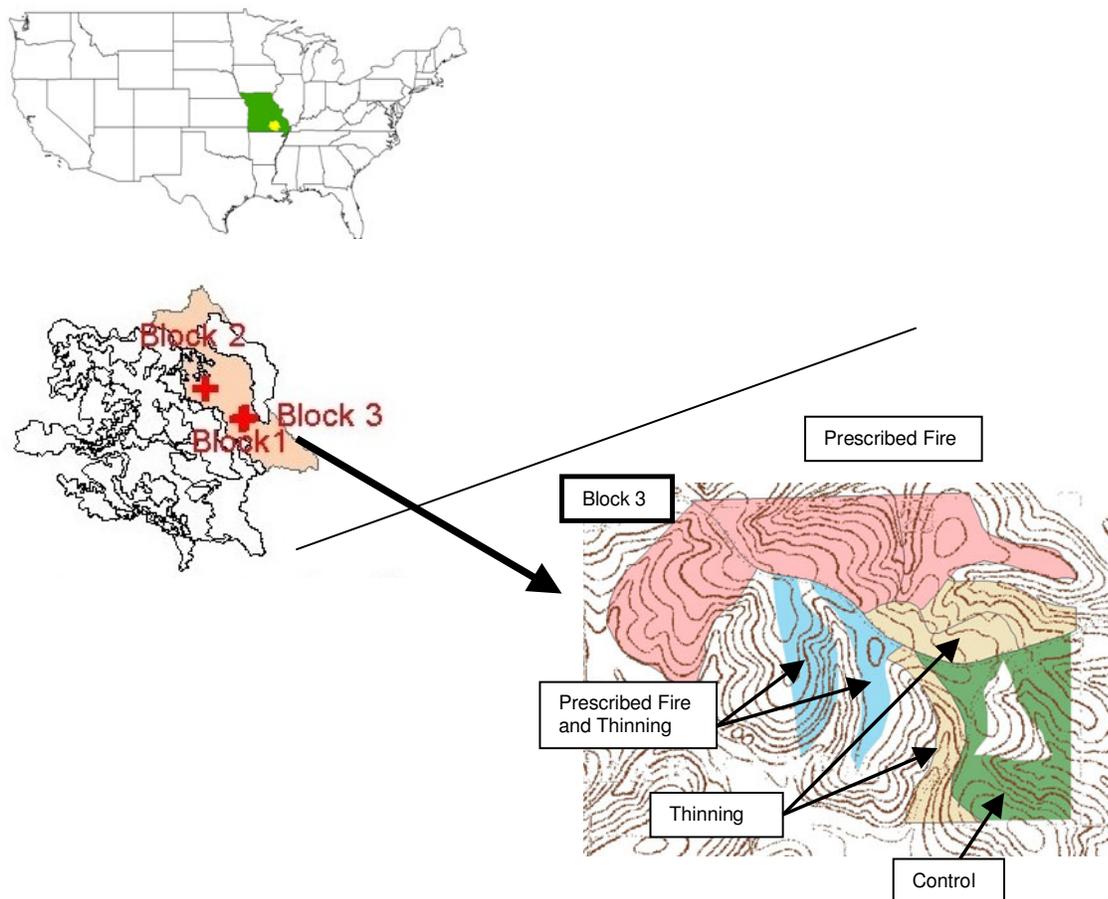


Figure 1.—Location of study sites within the United States and arrangements of treatments (block 3 used as an example).

Methods

Site Selection

Stands selected for the study had no management or documented fire for 30 years. All stands were fully stocked and composed primarily of oak-hickory and oak-pine forest types. The study was replicated across three complete blocks of 12 stands (3 aspect classes X 4 treatments) with each stand being an aspect/treatment unit. Aspect classes included exposed backslopes (135-315 degrees), ridge, and protected backslopes (315-135 degrees) (Nigh et al. 2000).

Treatments

Treatments were randomly assigned and included thinning, prescribed burning, both thinning and prescribed burning, and no treatment (control). Commercial thinning of the overstory occurred during the summer and early fall 2002. Thinning was accomplished using a mark-leave method. Preference was given to individuals having fire tolerance, good form, and canopy dominance. Leave trees were marked and a logger was allowed to harvest the remaining trees. Any remaining unmarked trees were slashed after the harvest was complete. Stocking was reduced to 60-percent. This stocking level is commonly used in intermediate cuttings, shelterwood systems, and savanna/woodland restoration (Johnson et al. 2002). Prescribed burns were completed in spring 2003 utilizing the ring fire method under a prescription commonly used in the region (Table 1).

Data Collection

Data were collected at 15 points within each stand prior to and after treatments were applied. These points were along a main transect placed at a random azimuth in each stand. The fifteen points were installed at

Table 1.—Weather parameter and fuel moisture prescription used for prescribed burning and observed values.

Attribute	Prescription	Observed Average	Observed Range
Temp (°F)	45-65	64	45-74
Mid Flame Wind (mph)	0-7	2.5	0-7.5
Rel. Humidity (%)	25-45	22.4	9-46
Fuel Moisture:			
1-hr.	5-10	5.4	5-6
10-hr.	8-15	9.8	8-12
100-hr.	12-18	13.7	13-18
1000-hr.	> 20	17.6	17-20

randomly-chosen 20-meter intervals along the main transect. Pre-thinning data, including controls, were collected winter/spring 2002. Post-thinning and pre-burn data collection occurred during winter/spring 2003. Post-burn and 2nd year control data collection was completed after the burns in spring 2003.

Fuels were inventoried from each of the 15 sampling points using a modified transect intercept method. Woody fuels were separated into four size classes: 0.0 to 0.25 in (1-hour), 0.25 to 1.0 inch (10-hour), 1.0 to 3.0 inch (100-hour), and greater than 3 in (1000-hour). 1000-hour fuels were further separated into rotten and solid categories. From each sample point, 1 and 10-hour fuels were inventoried along a 6 foot segment, 100-hour fuels along a 12 foot segment, and 1000-hour fuels along the entire 50 ft length of the transect. Fuel height, litter and duff depths were measured using a yard stick at 5 foot intervals along the fuel transect starting 1 foot from the origin (Brown 1974, Brown et al. 1982, Grabner 1996, Anonymous 2001). Litter and herbaceous samples were collected from a 2.0 ft² clip-plot located at the end of each fuel transect. Samples were then dried to a constant weight at 140° F (60° C) and reported on a dry-weight basis (Grabner 1996).

Data Analysis

For analysis, fuel data were organized into the following categories: litter, 1-hour, 10-hour, 100-hour, all fuel < ¼ inches (litter and 1-hour fuel), all fuel < 3 inches (litter and all woody fuel up to and including 100-hour), 1000-hour solid, 1000-hour rotten, total fuel load, fuel height, litter depth, and duff depth. Reliable fuel loading constants, specific gravities and squared average-quadratic-mean diameters, could not be found for species in the Central Hardwood Region. To compute fuel loadings in the 1, 10, and 100-hour time lag classes, we used fuel loading constants for northern red oak (*Quercus rubra* L.) taken from the National Park Service's Fire Management Handbook Software (USDI National Park Service). For 1000-hour rotten fuels, the average specific gravity for northern red oak decay classes 1-3 (Adams and Owens 2001) was used. We used these constants because the specific gravity of northern red oak is similar to black oak (*Q. velutina* Lam.), the most abundant species in our study area. Specific gravities for 1000-hour solid fuels were taken from the Wood Handbook (Forest Products Laboratory 1999).

Analysis of variance was used to determine if differences in fuel loading were related to treatment and aspect. Data were analyzed using the MIXED procedure in SAS. This procedure was used because it allows covariates to vary within a subject (Wolfinger and Chang 1995). A p-value of 0.05 or less was considered significant.

Results and Discussion

Pretreatment Fuel Loading

With exception to 1000-hour solid fuels, pretreatment fuel loading by timelag categories did not significantly vary between aspects (Kolaks et al. 2003). On average there was a progression of increasing total fuel load from exposed, to ridge, and finally to protected slopes. Differences worth noting (nearing

Table 2.—Pretreatment fuel loading and vertical structure.

	Fuel loading (tons/acre)							Vertical structure (in)		
	Litter	1-hour	10-hour	100-hour	1000-hour		Total	Fuel Height	Litter Depth	Duff Depth
E	3.0	0.5	0.3	1.0	0.80a	1.4	6.8	2.5	2.3	0.6
R	3.2	0.4	0.4	1.0	0.9a	1.8	7.9	3.0	2.7	0.7
P	2.9	0.4	0.3	1.6	2.1b	1.2	8.6	3.0	2.7	0.7

Aspects: E = exposed, R = ridge, P = protected
 Different letters within columns indicate significant (p<0.05) difference.

Table 3.—Fuel loading differences in tons/acre as result of commercial thinning.

	Fuel loading (tons/acre)							Vertical structure (in)				
	Litter	1-hour	10-hour	100-hour	< 1/4 in	< 3 in	1000-hour	Total	Fuel Height	Litter Depth	Duff Depth	
E	-0.7	0.4a	0.4	1.4a	-0.37*	1.4a*	13.1	1.5*	14.6	9.1a	-0.9	-0.20*
R	-0.2*	0.7	0.8	3.8b	0.42*	4.5b	13.3	1.2*	18.3	9.8	-1.1	0.1*
P	-0.7	0.7b	0.7	3.3b	-0.02*	4.5b	13.3	1.6*	19.6	12.8b	-1.0	-0.3*

Aspects: E = exposed, R = ridge, P = protected
 Different letters within columns indicate significant (p<0.05) difference.

*Differences not significant (p<0.05).

significance p=0.10-0.05) occurred between exposed slopes and ridges in 10-hour fuels, exposed and protected slopes in total fuel loading, between exposed and protected slopes as well as exposed slopes and ridges in fuel height, and exposed and protected slopes in litter depth (Table 2) (Kolaks et al. 2003).

Thinning

Changes Caused by Thinning. Commercial thinning reduced litter weight, all fuel < 1/4 in, litter depth, and duff depth (Table 3). All other categories showed an increase with 1000-hour solid fuels contributing the most followed by 100-hour fuels. However, the change in all fuel < 1/4 in, 1000-hour rotten, and duff depth on any aspect, as well as litter weight on ridges was not significantly different from pretreatment levels.

Increases were significantly higher on ridges and protected slopes than on exposed slopes in the 100-hour and all fuel < 3 in categories. Fuel height increases were also significantly higher on protected slopes as opposed to exposed slopes. Though not significant, the difference between exposed slopes and ridges in 1-hour fuels, exposed slopes and ridges in 10-hour fuels, and ridges and protected slopes in 100-hour fuels are worth noting (approaching significance (p=0.10-0.05)).

Reduction in litter weight, litter depth, and duff depth is most likely the result of the mechanical harvesting operation (rubber-tire skidder) that moved litter and duff laterally across the landscape. Many areas were completely devoid of litter and duff due to skid trails and landings. In addition, a more open overstory allows greater wind velocities at ground level that could remove or concentrate loose leaf-litter making it less likely to be sampled. Though not significant, the increase in duff depth on ridges could be attributed to uphill skidding during harvesting operations that deposited organic material from the surrounding exposed and protected slopes.

Post-Thinning Fuel Loading. There was an increase, albeit non-significant, in total fuel load from exposed slopes, < ridge, < protected slopes (Table 3). The end result of thinning was an increase in total fuel loading of about 300% (Table 4). 1000-hour solid fuel replaced litter as the heaviest component of total fuel loading. 10-hour fuel loading was significantly higher on ridges than on exposed slopes. Both ridges

Table 4.—Post-thin fuel loading and vertical structure.

	Fuel Loading (tons/acre)				1000-hour			Vertical Structure (in)		
	Litter	1-hour ¹	10-hour ¹	100-hour	Solid	Rotten	Total	Fuel Height	Litter Depth	Duff Depth
E	2.6	0.9	0.8a	2.4a	14.3	2.7	22.5	12.4	1.8	0.6
R	2.6	1.1	1.2b	4.7b	14.6	2.2	25.7	13.2	1.9	1.1
P	2.7	1.2	1.1	4.5b	17.3	2.3	28.4	16.3	1.9	0.7

Aspects: E = exposed, R = ridge, P = protected

¹Different letters within a column indicate significant (p<0.05) difference.

and protected slopes showed significantly higher fuel loading than exposed slopes in 100-hour and all fuel < 3 inches categories. Fuel height was also significantly higher on protected slopes as opposed to exposed. Nearing significance (p=0.10-0.05) was the differences in exposed and protected slopes for both 1 and 10-hour fuels. Differences among aspect trends were similar to those found in pre-treatment fuel loading; progression from exposed, < ridge, < protected.

Fuel descriptions for any of the 13 standard fuel models did not accurately describe our post-thinning fuel loadings (Anderson 1982) (Table 5). Post-thinning fuel load for all fuel < 1/4 in are most closely approximated by fuel model 9, hardwood leaf litter; 10, timber; and 12, medium logging slash. For all fuel < 3 in, fuel model 11, light logging slash, most closely describes ridges and protected slopes while fuel model 6, dormant brush and hardwood logging slash, best describes exposed slopes. Since significant difference exist between aspects in all fuel < 3 in, a different fuel model may need to be utilized on exposed slopes than on ridges and protected slopes. Also, since no single fuel model best describes overall post-thinning fuel loading, the selection of a fuel model based on fire behavior may be warranted (Ryan 1981, Anderson 1982).

Prescribed Burning

Consumption. For the most part, all burns were conducted within prescription. Average relative humidity (RH) was below prescription, however, ignition operations were completed before the RH dropped below the lower threshold (Table 1). Weather and 10-hour fuel moisture were taken by an on-site automated weather station. One, 100, and 1000-hour fuel moistures were taken from 2 automated weather stations that experienced similar weather patterns located in relatively close proximity (9 and 15 miles).

Average flame length was greater in the burn-thin treatment, 23 in, than in the burn only treatment, 18 in. However, the difference between treatments was not significant except when accumulations of slash were encountered. Flame length off of slash accumulations ranged from 5 to 50 ft with 14 ft being most common. Average flame lengths varied from 21 to 27 in on protected and exposed slopes, respectively, and 13 in on ridges. Observed rates-of-spread were higher on the slopes than on ridges.

Prescribed burning reduced fuel loading and vertical structure in all categories in both thinned and unthinned treatments (Table 6 and 7). Fuel consumption decreased as timelag size class increased (Table 6 and 7). Consumption did not significantly vary among aspects for either treatment. However, the data

Table 5.—Comparison of fuel loading in commercially thinned stands and loadings assumed by BEHAVE.

	Fuel loading (tons/acre)	
	< 1/4 in (1-hr)	< 3.0 in (100-hr)
Anderson (1982)		
Fuel Model 6	1.5	6.0
Fuel Model 9	2.9	3.5
Fuel Model 10	3.0	12.0
Fuel Model 11	1.5	11.5
Fuel Model 12	4.0	34.6
This Study		
Exposed	3.5	6.74a
Ridge	3.4	9.51b
Protected	3.9	9.43b

Different letters within a column indicate significant (p<0.05) difference.

Table 6.—Consumption of fuel for the burn-only treatment by aspect and fuel loading category.

Aspect ¹	Litter	Fuel Loading (tons/acre)					1000-hour		Vertical Structure (in)			
		1-hour	10-hour	100-hour*	< 1/4 in	< 3 in	Solid*	Rotten*	Total	Fuel Height	Litter Depth	Duff Depth
E	-2.9	-0.3	-0.2	-0.3	-3.2	-3.7	-0.1	-0.5	-3.9	-2.4	-2.1	-0.3
Percent	99	60	45	35	93	78	30	60	67	91	98	45
R	-3.0	-0.2*	-0.1	-0.1	-3.2	-3.6	-0.1	-0.2	-3.5	-2.8	-1.9	-0.3*
Percent	97	48	19	16	92	79	12	9	49	91	97	35
P	-2.5	-0.3	-0.1	-0.3	-2.8	-3.0	-0.1	-1.1	-3.5	-2.5	-2.0	-0.2*
Percent	96	27	16	19	89	62	37	41	47	90	96	27

¹Aspects: E=exposed, R=ridge, P=protected

*Consumption not significant (p<0.05) for individual value or entire category.

Table 7.—Consumption of fuel by aspect for the burn-thin treatment by fuel loading category and percent.

Aspect ¹	Litter	Fuel Loading (tons/ acre)					1000-hour		Vertical Structure (in)			
		1-hour	10-hour	100-hour	< 1/4 in	< 3 in	Solid	Rotten*	Total	Fuel Height	Litter Depth	Duff Depth
E	-2.8	-0.5	-0.4*	-.1*	-3.3	-4.6	-1.7	-1.0	-6.0	-7.0	-1.9	-0.30*
Percent	100	56	53	5	89	66	12	51	28	50	98	38
R	-2.7	-0.9	-1.0	-1.6	-3.6	-6.1	-0.7*	-.05	-7.20	-6.4	-2.0	-1.0
Percent	97	75	66	30	91	57	5	3	27	41	93	68
P	-3.0	-1.0	-0.6*	-1.1*	-3.9	-5.6	-1.3*	-1.0	-6.6	-8.0	-2.1	-0.40*
Percent	100	73	57	26	91	58	7	39	24	43	99	45

¹Aspects: E=exposed, R=ridge, P=protected

*Consumption not significant (p<0.05) for individual value or entire category.

suggest that a greater proportion of consumption occurred on exposed slopes in 1, 10, and 100-hour timelag classes than on ridges and protected slopes.

In all cases nearly 100 percent of litter was consumed. Litter was responsible for the consumption of about 90 percent of all fuel < 1/4 in and about 75 percent of all fuel < 3 in for un-thinned sites as well as 75 and 50 percent, respectively, for thinned sites. Scowcroft (1965) found somewhat similar results reporting that 90 to 85 and 80 to 70 percent of leaf litter was consumed on unthinned annually burned and periodically burned (every five years) sites, respectively.

The near 100 percent consumption of leaf litter along with the remaining percentages of fine fuel (all fuel < 3 in) eliminated surface fuel continuity, and a major influence on fire behavior (Davis 1959, Brown and Davis 1973, Anderson and Brown 1987) making the potential for immediate reburn virtually nonexistent. However, reburn can occur shortly after the first leaf-fall. Scowcroft (1965) reported that in the three years following a burn, 4 tons/acre of litter accumulated. He also reported an increased percentage of woody fuel on burned sites that fell from trees killed by the fire. With 50 percent of equilibrium litter accumulation being regained in 2.5 years, and 75 percent in five years after a fire (Guyette 1999), surface fuels and horizontal continuity return very quickly.

One hundred, 1000-hour solid, 1000-hour rotten, and duff depth consumption resulting from the burn was not significant for both treatments on most aspects (Tables 5 and 6). Burn-only treatment data suggest that a greater percentage of 1000-hour fuels were consumed on slopes of either aspect than on ridges. Higher intensity fires on the slopes likely caused this while fire behavior on the ridges was mostly only wind driven. However, this effect was not observed in the burn-thin treatment.

Table 8.—Post burn fuel loading by aspect for burn-only and burn-thin treatments.

¹ Aspect	Fuel Loading (tons/acre)						1000-hour			Vertical Structure (in)		
	Litter	1-hour	10-hour	100-hour < 1/4 in	< 3 in		Solid	Rotten	Total	Fuel Height	Litter Depth	Duff Depth
<u>Burn-only</u>												
E	0.03	0.2	0.2	0.6	.2	1.0	0.2	0.3	1.9	0.2	0.05a	0.4
R	0.08	0.2	0.2	0.5	.3	1.0	0.8	1.5	3.7	0.3	0.06	0.5
P	0.10	0.8	0.3	1.3	.4	1.9	0.1	1.6	4.0	0.3	0.08	0.6
<u>Burn-thin</u>												
E	0.01	0.4	0.4	1.5a	.4	2.3a	12.2	1.0	15.6	7.1a	0.04a	0.5
R	0.08	0.3	0.5	3.8b	.4	4.6b	13.3	1.7	19.5	9.4	0.14b	0.5
P	0.00	0.4	0.5	3.2b	.4	4.0b	15.6	1.6	21.3	10.8b	0.03a	0.5

¹Aspects: E = exposed, R = ridge, P = protected

Different letters within a column indicate significant (p<0.05) difference

*Differences between treatments not significant (p<0.05) for category

The lack of consumption indicates that under weather conditions conducive to prescribed burning larger woody debris and duff will not be substantially reduced. However, wildfire during high-risk conditions such as high temperatures, low relative humidity (< 20 percent), and low 1000-hr fuel moistures (< 15 percent) could yield different results. Our prescribed burn consumption results are contrasted by memorable fire seasons in Missouri, 1980 (Westin 1992) and 1999-2000, when 1000-hour fuels were readily consumed. This consumption was mostly likely the result of high wildfire risk conditions brought on by prolonged drought. Prescribed burning under these conditions could compromise objectives and holding efforts.

If prescribed fire is used as a site preparation tool for shortleaf pine regeneration several applications may be needed for sites that have not been previously burned. Burning only reduced the duff layer an average of 46 percent (Table 6 and 7). A layer of duff about ½ in thick (Table 8) remained after the burn on all aspects and in both treatments. This could possibly inhibit the establishment of pine seedlings from natural regeneration or direct seeding (Burns and Honkala 1990).

Overall burning reduced total fuel loading by about 50 percent on burn-only stands and 25 percent on burn-thin stands (Table 6 and 7). Seventy percent of all fuel < 3 in was consumed in burn-only stands and 60 percent in burn-thin stands compared to 50 percent consumption of fuel < 3.15 in reported by Clinton et al. (1998) in mixed white pine-hardwood stands of the southern Appalachians. In general, consumption in all categories except litter was less than what occurs in timbered stands of the western US (Kauffman and Martin 1989).

The Effect of Aspect on Post-Burn Fuel Loading

Burn Only. There were no post-burn significant differences between aspects in all categories of the burn-only treatment (Table 8).

Burn-Thin. Several categories, including 100-hour, all fuel < 3 in, fuel height, and litter depth exhibited post-burn differences between aspects in the burn-thin treatment (Table 8). Since 100-hour fuels are a component of all fuels < 3 in, it was not surprising that both categories were greater on protected slopes and ridges than on exposed slopes (Table 8). These differences are almost identical to those observed after thinning (Table 3). However, burning also produced a near significant (p=0.10-0.05) difference in total fuel loading, not observed pre-burn.

Fuel height was greater on protected slopes compared to exposed slopes. Although not indicated in post-thinning data, litter depth on ridges was greater than both protected and exposed slopes after the burn (Table 8). This greater depth was most likely the result of greater fragmentation of horizontal

continuity observed on ridges from increased mechanical harvesting traffic. Small islands of unburned fuel were common in high traffic areas and could have affected litter depth compared to areas where consumption and surface fuel continuity were more uniform.

There was no significant difference between the burn-only and burn-thin treatments in litter, 1-hour, all fuel < ¼ in, 1000-hour rotten, litter depth, and duff depth categories following the burn (Table 8). This is not surprising considering that many of the same categories were not significantly different from pre to post-thinning conditions (Table 3).

Conclusion

The results from this study indicate that aspect affects some categories of fuel loading in both unburned and unthinned stands as well as thinned stands. Thinning significantly alters 1, 10, 100-hour, and 1000-hour solid fuel loading while not affecting the others. In both thinned and unthinned stands there is a progression in fuel loading from exposed slopes, < ridge, < protected slopes. Significant changes in fuel loading due to position in the landscape could be more prevalent at smaller scales of ecological classification. Further research is also needed in developing constants for calculating fuel loadings in the Central Hardwood Region.

Consumption during prescribed fires was not significantly affected by aspect. Post-burn aspect differences were almost identical to pre-burn differences found primarily in heavy fuels not significantly affected by burning. With litter making up 50 percent or greater of all fuel < 3 in, prescribed burning will only temporarily reduce the threat of fire in the Central Hardwood Region. Since fine fuels return fairly quickly surface fuel continuity will soon be restored. One and 10-hour fuels are likely to increase after the burn do the contribution of branch wood from tree killed by the fire (Scowcroft 1965). This combined with the additional curing time had by 100 and 1000-hour fuels not consumed by the first fire, could result in high intensity fires in the near future.

Acknowledgments

Thanks to Mike Morris, Cliff Smith, Randy Jensen, David Whitener, Keith Crabtree, the Ellington Work Team, Ironton Work Team, Garwood Work Team, and other employees of the Missouri Department of Conservation who helped as well as Kenneth Davidson, Chris Hopfinger, Tim Parsons, Texas Nall, Ralph Yohai, Bill Dijak and Shawn Maijala of the USDA Forest Service North Central Research Station and the Mark Twain National Forest. Also thanks to Dr. Adnan Akyuz of the Missouri Climate Center for the use of a portable automated weather station and interpretation of weather data. Finally I would like to especially thank Sandra Clark, Nick Cahill, Michael Garcia, and Dr. Mark Ellerseick as well as my fellow students.

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TIMBER HARVESTING, PRESCRIBED FIRE AND VEGETATION DYNAMICS IN THE MISSOURI OZARKS

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ABSTRACT.—Little is known about the response of Missouri Ozark ground flora to silvicultural treatment. In this project, we examined herbaceous and woody species response to various thinning regimes and prescribed burns over several years at the Missouri Ozark Forest Ecosystem Project (MOFEP) and the Chilton Creek Management Area (CCMA). On MOFEP, ground flora and woody species regeneration data were collected across nine sites and three harvest treatments: 1) uneven-aged management, 2) even-aged management, and 3) no-harvest management. On CCMA, ground flora and woody species regeneration data were collected across burn units with different fire frequencies: 1) burned in 1998 and 2002, and 2) burned annually since 1998. Ecological Landtype Phase (ELTp) was used to stratify plots and for comparison and contrast among treatments. For woody regeneration up to 1 m, seedling and sapling development was greater on units that have been burned twice than units burned annually. For both fire frequencies and across ELTPs, white oak (*Quercus alba*) dominated regenerating species. White oak regeneration was also abundant in most of the areas on MOFEP, particularly the uneven-aged plots. Successional trajectories were developed to examine treatment and ELTp effect on species change over time. The trajectories appear to be influenced by treatment as well as ELTp. Burning homogenized the ground flora vegetation on the sites, and decreases variability among ELTPs, creating a consistent trajectory across ELTp. The effects of thinning and clearcutting, however, seem to be related to ELTp, therefore these ecological units can be used to understand and possibly predict the influence of specific management on vegetation.

Long-term studies of sufficient spatial extent are critical for understanding response of the vegetation communities associated with areas under active forest management. Furthermore, management for forested lands continues to develop as data reveal the consequences of various silvicultural approaches. Several large-scale studies have been conducted in attempts to characterize effects of forest management. For example studies in the Pacific Northwest (e.g., Halpern and Spies 1985, Franklin and others 2002), the Appalachians (Gilliam and others 1995, Meier and others 1995), and the Lake States (Metzger and Schultz 1984) underscore the need to include long time periods, realistic management objectives and large physical scale when examining effects of forest management activity.

Ecological land classification, a hierarchically structured, multi-factor approach for mapping ecological units at multiple scales, has been shown to be helpful in quantifying variation in ecological processes (Host and others 1987). As such, it may be valuable as a framework to examine the long term influences of management. This classification system characterizes the landscape by ecological variables including vegetation structure and composition. Management units 10's to 100's of acres (approximately 5-50 ha) in size can be defined by soil, aspect, geology and vegetation (Ecological Land Type phases [ELTPs]), and represent a typical management level unit. Although few studies have incorporated the use of Ecological Classification Systems or specifically ELTp as an approach to understanding the effects of management, evaluation of the ecological units and their ability to describe variation in ground flora structure and composition, and response to management will contribute to the overall utility of an Ecological Classification System (ECS).

Numerous research efforts have examined the effect of silvicultural practices on important timber species, woody regeneration, and to a more limited extent, ground flora species. Changes in herbaceous species composition can provide a sensitive measure of ecologically relevant changes in the environment

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(Daubenmire 1952, Phillippi and others 1998). Analysis in an ECS framework would disclose more about the responses at the classification unit level and may make results more understandable and useful to areas outside the study sites.

In order to develop a greater understanding of the role of forest management in forest dynamics of the Missouri central hardwoods forests, the Missouri Department of Conservation (MDC) initiated the Missouri Ozark Forest Ecosystem Project (MOFEP) in the southeast Missouri Ozarks in 1989. MOFEP is a 100-year project designed to study the effects of traditional silvicultural methods of forest management (unevenaged, evenaged and no-harvest [control]) on various ecosystem components (Brookshire and others 1997). While the effects of even-aged management on regeneration have been studied to some degree in the Missouri Ozarks, attempts to examine the effects of uneven-aged management on regeneration are relatively new. Uneven-aged methods are significantly varied, often with site factors determining which uneven-aged method and to what extent it is used (Smith and others 1997, Larsen and others 1999, Loewenstein and others 2001).

Nearby to the MOFEP study, the Nature Conservancy initiated a case study at the Chilton Creek Area to determine the effects of fire-based ecosystem management, with the management of the site aimed at restoring and sustaining the native biota of the site through prescribed burning (The Nature Conservancy 1996). Dendrochronological and cultural evidence suggests fire interval in the Missouri Ozarks varied substantially during the last several centuries (Guyette and others 2002). In the Missouri Ozarks the reduction in fire intensity and frequency followed extensive logging of forests, many of which were dominated by shortleaf pine (*Pinus echinata*). Despite a resource that once accounted for millions of acres, short leaf pine now accounts for fewer than 200,000 hectares. Implications for management of the hardwood resource in Missouri, obviously encompasses potential for understanding and managing the pine resource in the region, as well. The recognition that both oak and pine dominated Ozark forests may rely on fire has increased the used of prescribed fire in Missouri forests. Although there have been recent efforts of re-introducing fire, the effect of fire on plant species composition, regeneration, succession, diversity, habitat, or wood quality remains unknown. The Chilton Creek study makes use of MDC's Ecological Classification System to organize hierarchical management units based on soil, geology, landform and vegetation; these units presumably represent stand-sized systems that would respond similarly to management. Thus, CCMA provides an ideal opportunity to directly link classification units to response.

The objective of this project was to explore how prescribed fire, timber harvest and ELTPs influence ground flora composition, and explain forest dynamics. Although the harvesting treatments and burning treatments represent independent studies, we used ELTP as a sampling framework. We examine woody species response to treatment and overall richness, as well as examining how these change over time using ordination. ELTPs are used not only to stratify and compare silvicultural treatments, but also to examine the interaction of silvicultural treatments and ELTPs. We examined response of herbaceous and woody species for 1) specific ELTPs by treatment, and 2) within a treatment by ELTP. We contrast two long-term projects, which characterize typical forest management in the Missouri Ozarks. Although the study uses ELTP to stratify the plots, this project does not represent a test of the ECS *per se*, or the classification of ELTs and ELTPs

Study Area

Both study areas, MOFEP and CCMA, are located in the southeast portion of the Ozark Highlands. The physical boundaries of the Ozark Highlands are often open to debate, but in general, the region includes greater relief and steeper slopes than surrounding areas with exposed rock being older than outside the region (Rafferty 1980). Geologically, there is more dolomite, as opposed to limestone, and a prevalence of hard chert nodules (Rafferty 1980, Meinert and others 1997) when contrasted with adjacent areas.

MOFEP and CCMA are in the Current River Hills subsection, an area dissected by the Black, Current and Eleven Point rivers with local relief ranging from 61 to 183 meters (Nigh and Schroeder 2002). A subsection is an ecological classification level that can be > 1,000 acres (>400 ha). The Current River Hills subsection is mostly second growth forests of oak, oak-pine and mixed hardwood timber types. MOFEP

is composed of nine sites, each at least 240 ha and relatively free of manipulation for no less than 40 years. There are thirteen ELTPs on MOFEP with 43 soil map units. Five ELTPs, comprising a majority of the landscape of interest to land managers. These have been characterized by Nigh and others (2000) (numbers indicate ELTP designations):

- high, ultic shoulders (2.1)
- exposed, ultic, Roubidoux/upper Gasconade backslopes (3.1)
- protected, ultic, Roubidoux/upper Gasconade backslopes (4.1)
- exposed, alfic, lower Gasconade/Eminence backslopes (5.2)
- protected, alfic, lower Gasconade/Eminence backslopes (6.2).

Local relief at CCMA ranges from 24 to 103 m (80 to 340 ft) with elevation ranging from 152 to 299m (500 to 980 ft). There are fifteen ELTPs on CCMA, four of which are further characterized by this project:

- high, ultic shoulders (2.1)
- exposed, ultic, Roubidoux/upper Gasconade backslopes (3.1)
- protected, ultic, Roubidoux/upper Gasconade backslopes (4.1)
- exposed cherty and non-cherty variable depth to dolomite units, upper and lower Gasconade/Eminence, multiple landforms (7.12).

For this paper, we will focus on the ELTPs common to the two sites: 2.1, 3.1, 4.1 as well as 7.12.

Methods

On MOFEP, the nine sites are grouped into three replicated blocks. Each block (about 400 ha) contained one even-aged, one uneven-aged and one no-harvest (control) unit. All units were randomly allocated. Stratified random sampling was used to locate 70 to 76 vegetation plots, each 0.2 ha in size, in each of the nine sites for a total of 648 plots (Brookshire and others 1997). Each plot contains four 0.02 ha subplots, with four 1- m² quadrats within each subplot.

Treatments tested include plots located in clearcuts (CC) and intermediate harvest (I) as evenaged management (EAM); group openings (UG) and select harvesting (U) as unevenaged management (UAM) and one control (L) as no-harvest management (NHM). The first harvest began in 1996. Stands with site index ≥ 55 were managed with intermediate cutting. Harvest prescriptions follow Roach and Gingrich (1968), with approximately 10 to 12 percent of the site in clearcuts. Clearcutting and thinning affected 26% of the total area of the three even-aged sites with 15% (166 ha) and 11% (130 ha) respectively.

For uneven-aged management approximately 10 percent of each site was designated as “old growth” and the remaining 90 percent managed using group openings and select harvest (Shifley and Brookshire 2000). Approximately 57 percent of the total area of uneven-aged sites received single-tree selection harvests. There were between 153 and 267 group openings on each of the UAM sites, with group openings on south-facing slopes 21m in diameter (i.e. approximately one tree height), north-facing slopes 42m in diameter and ridge tops 32m in diameter (Law and Lorimer 1989). This allowed for a total of 10 to 19 ha per site in group openings.

The 2,268 ha Chilton Creek site is divided into five units for treatment, one of which is burned annually. The other four units are burned on a random 1-4 year return interval. The first prescribed burn occurred spring 1998, and the most recent data included in this analysis is from 2002. Each unit has then burned on a schedule based on random return intervals generated for each unit, independent of the other units. The result is a project area that has one annually burned unit and 0-4 units burned in any given year. A total of 250 plots were established randomly after stratification using ECS structure. Plot design and vegetation sampling follows MOFEP criteria for circular 0.2 ha plots (Brookshire and others 1997). For this paper, we will focus on plots that were annually burned and plots that have been burned twice in the five year period.

Ground flora vegetation sampling on both MOFEP and CCMA occurred from May to August. Within in each 1m² quadrat, all vascular species <1 m tall with live foliage were identified and percent cover to the nearest 1 percent estimated. Stems were counted for all tree seedlings less than 1m in height that were rooted in the quadrat. Woody species data were also collected from the 0.02 ha subplots and the 0.004 ha subplots. The circular subplots are nested within the 0.2 ha plot, with four of each subplot. In the 0.004 ha subplot, live trees at least 1m tall and less than 4 cm d.b.h. were tallied by species and size class.

Statistical Analysis

For MOFEP and CCMA, richness was calculated as the number of species per plot (16m²). Plot richness was evaluated with a combination of ninety-five percent confidence intervals and pairwise t-tests. Pairwise t-tests were used to test for treatment effect, year effect and ELT effect on richness. Upper and lower confidence intervals were calculated using SPSS version 11.0.

Temporal patterns were examined by plotting ELTp ordination scores obtained through Detrended Correspondence Analysis (DCA). Ordination of community samples through time reveals successional trajectories (Halpern 1988). Mean ELTp scores were plotted in two dimensional species space. Detrended correspondence analysis was chosen because of its computation efficiency with large data sets and its ecologically interpretable and intuitive results.

DCA scores were used to obtain measures of resistance and resilience, or the degree to which community composition changes and recovers after disturbance. Euclidean distance was used in two-dimensional ordination space to measure compositional dissimilarity between the initial and each post-disturbance ELTp sample. Resistance to disturbance was defined as inversely proportional to the maximum Euclidean distance between pre- and any year post-disturbance samples (Halpern 1988). Resilience, or the recovery of a community to pre-disturbance conditions, was defined as inversely proportional to the Euclidean distance between pre-disturbance and the final post-disturbance sample for each ELTp (Halpern 1988).

Successional trends were examined using trajectory analysis (McCune and Grace 2002). Vectors of change over time from pre-treatment to last year post-treatment were standardized by subtracting tail from head and tail. PC-ORD version 4.14 (McCune and Mefford 1999) was used to perform multivariate analyses.

Results

Regenerating species

CCMA – the effect of burning. Woody tree species regeneration was examined for five tree species: black oak (*Quercus velutina* Lam.), white oak (*Quercus alba* L.), post oak (*Quercus stellata* Wangenh.), scarlet oak *Quercus coccinea* Muenchh.), and shortleaf pine (*Pinus echinata* Miller). These species were sampled in various size classes from ground flora layer (<1m tall) to small sapling layer (≥1m tall and <3.81cm dbh) and large sapling layer (≥3.81cm to <11.43cm dbh). This paper will focus on the two larger size classes.

Across all ELTps, in the small sapling layer, annual burning resulted in a reduction of an average of 2600 stems per ha and 1316 stems per ha in the twice-burned plots. On annually burned and twice burned plots, the large sapling layer decreased by 229 and 183 stems per ha, respectively.

There was little change in shortleaf pine regeneration relative density in any size class from pre-treatment (1997) to post-treatment (2001) on CCMA (figs. 1a, b). The exception, however was a drastic decrease in twice burned ELTp 3.1 in the small sapling class. Overall abundance of post oak, scarlet oak, white oak and black oak showed little change as a result of prescribed burning. Scarlet oak exhibited decline in 2001 in the small sapling class, annually burned plots across all ELTps (fig. 1a). In the large sapling layer, this reduction occurred on annually burned 7.12s and on twice burned 2.1s and 3.1s .

There was no effect of prescribed burning on post oak regeneration. White oak relative density changed on specific ELTps, but overall was not significantly affected by burning. In the small sapling layer (fig. 2a) white oak had significantly reduced relative density on annually burned 7.12s. However in the larger sapling class (fig. 1b), relative density of white oak increased on twice burned 3.1s.

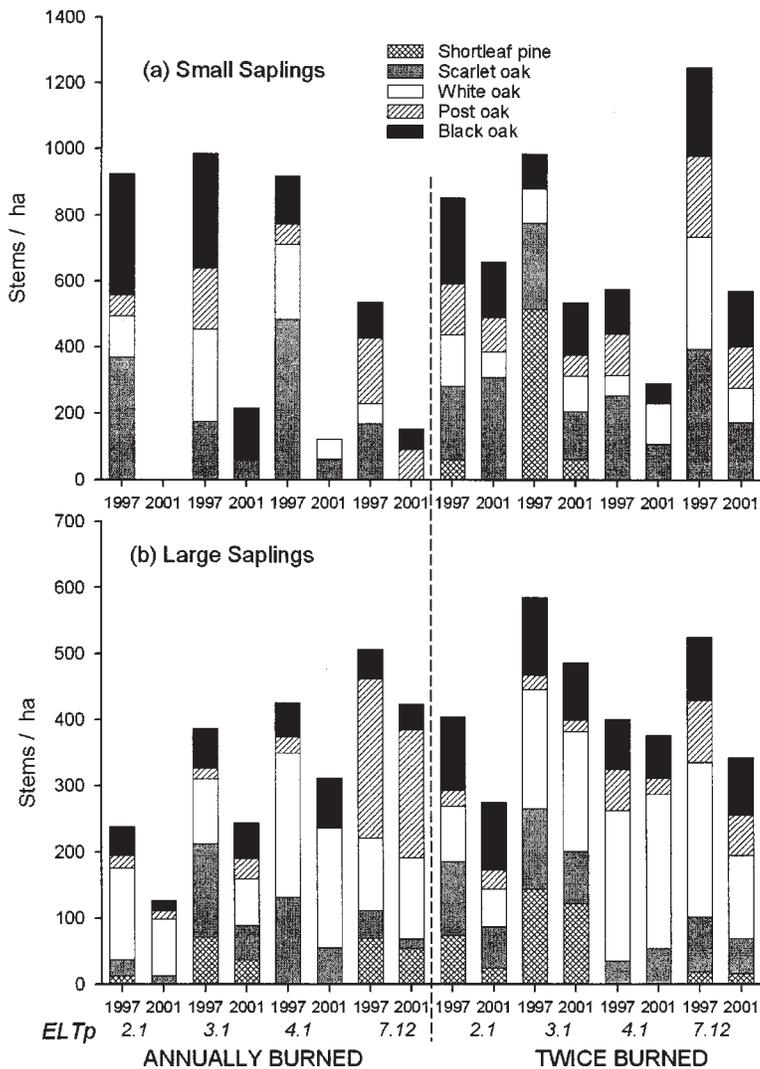


Figure 1a, b.—Mean number of stems per ha for (a) small sapling layer ≥ 1 m and < 3.81 cm dbh and (b) large sapling layer (≥ 3.81 cm and < 11.43 cm dbh) on CCMA pre-treatment (1997) and post treatment (2001) for shortleaf pine, black oak, white oak, post oak and scarlet oak.

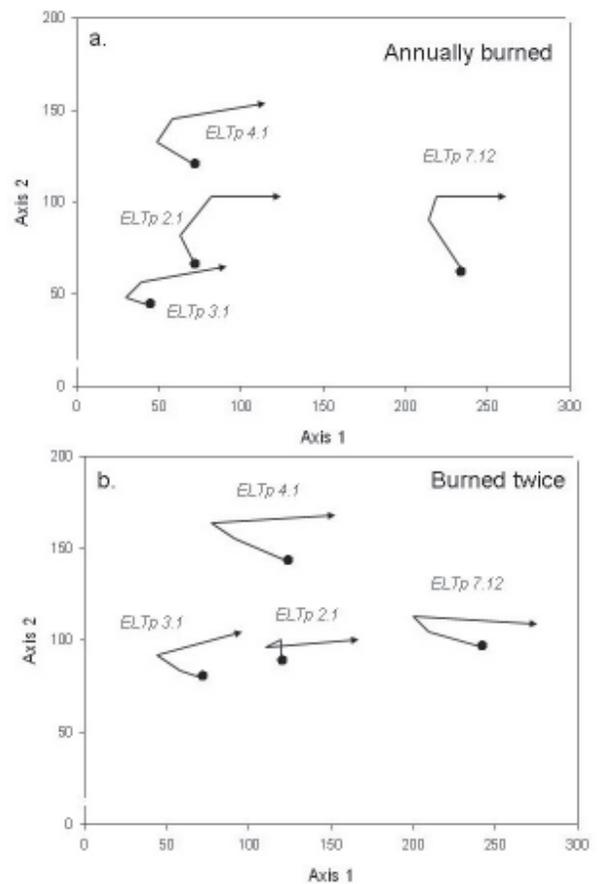


Figure 2a, b.—Detrended correspondence analysis (DCA) ordination through time representing understory communities of ELTPs on CCMA annually burned (since 1998) sites (a) and sites burned twice in a five year period (b). Symbol endpoints indicate pre-disturbance communities (1997); lines connect the same ELTP in subsequent years post-disturbance (1998, 2001, 2002); arrowheads coincide with the final sample and indicate the direction of change through time.

MOFEP—the effect of harvest. Generally, regardless of treatment, there were declines in the 5 species of interest, through the course of the study. With the exception of the clearcut sites, harvest treatments on MOFEP had little effect on the relative density of many species. Relative density of black oak did not change the sapling layers. Over the course of the study, white oak decreased generally except in clearcut ELTP 2.1 (table 1 and table 2). However, density in the small sapling class had not decreased from pre-treatment and had increased in clearcuts on ELTP 4.1 and group opening 2.1s (table 1). The large sapling class had significant increases in white oak relative density for all treatments and most ELTPs, with the exceptions of clearcuts on ELTP 3.1 and group opening on ELTP 4.1 (table 2). Shortleaf pine was common on certain ELTPs, namely 3.1 and 2.1 only in the small sapling class. Clearcutting appears to have reduced pine saplings.

In general, there were more saplings in harvested plots for select ELTPs than NHM. Indicative of the current overstory vegetation (Sasseen 2003), post oak was more abundant on control sites in the ground flora layer, than other treatments. Post oak is also the most common large sapling in clearcuts.

Table 1.—Stems per hectare of small sapling (>1 m tall and <3.81 cm DBH) on MOFEP during pretreatment (1994) and most recent sampling (2002) following treatment on select ELTps. Treatments include CC=clearcut (EAM), I=intermediate (EAM), L=No harvest (NHM), U=select (UAM), UG=group openings (UAM).

Treatment	ELTp	Year	Shortleaf pine	Scarlet oak	White oak	Post oak	Black oak
CC	2.1	1994	0	0	0	0	0
		2002	0	1482.6	1420.8	0	988.4
	3.1	1994	92.7	30.9	216.2	61.8	293.4
		2002	92.7	756.7	1776.0	339.8	1050.2
	4.1	1994	0	10.3	8.8	0	26.5
		2002	0	483.9	626.6	308.9	485.4
I	2.1	1994	0	0	92.7	0	30.9
		2002	0	0	154.4	0	154.4
	3.1	1994	61.8	46.3	20.6	0	61.8
		2002	0	61.8	103.0	61.8	144.1
	4.1	1994	0	123.6	0	0	0
		2002	0	61.8	329.5	0	61.8
L	2.1	1994	0	20.6	150.0	61.8	113.3
		2002	0	103.0	150.0	226.5	113.3
	3.1	1994	185.3	103.0	44.1	123.6	154.4
		2002	370.7	92.7	79.4	0	61.8
	4.1	1994	0	61.8	108.1	0	92.7
		2002	0	0	77.2	0	61.8
U	2.1	1994	30.9	37.1	101.1	0	69.5
		2002	61.8	74.1	213.4	0	193.0
	3.1	1994	0	51.5	88.3	41.2	154.4
		2002	0	72.1	432.4	267.7	144.1
	4.1	1994	0	20.6	113.3	0	0
		2002	0	61.8	123.6	0	144.1
UG	2.1	1994	0	92.7	46.3	0	0
		2002	0	30.9	339.8	0	61.8
	3.1	1994	0	82.4	77.2	247.1	123.6
		2002	1482.6	329.5	378.4	617.8	347.5
	4.1	1994	0	0	74.1	0	0
		2002	0	123.6	259.5	0	164.7

Composition

CCMA. Generally, plot level species richness was significantly different from pretreatment on annually burned plots over time (table 3). The exception is ELTp 7.12, which differed only immediately following first year of burning. Plots that were burned twice in ELTps 2.1, 3.1, and 4.1 differed significantly after burning, but the distinction in richness became less evident over time. In general, richness was greater over time in all ELTp and treatment combination.

DCA plot scores in species space for successive years were plotted as successional trajectories in two dimensional ordination space. Both treatments had similar patterns for all ELTps (fig. 2a, b). CCMA showed distinct separation of ELTps along both axes. ELTp 7.12 was distinctly distanced from the other three ELTps on axis one. Along axis 2, ELTp 4.1 (protected, upper backslope) was separate from other ELTps. For all the ELTps, however the trajectory was almost identical. Initially, ELTps changed along axis 1, but then the trajectory increased along axis 2, before finally reversing direction along axis 1 and returning almost to pre-treatment conditions. On CCMA, successional trajectories were not significantly different between treatments, but within a treatment annually burned 3.1s and 4.1s were significantly different.

DCA scores were also used to calculate relative resistance and resilience of each ELTp after treatment. CCMA ELTp 7.12 was the most resistant to disturbance and also the most resilient, i.e. it did not change as much after treatment as other ELTps and it returned to pre-treatment conditions rapidly (table 4).

Table 2.—Stems per hectare of large sapling (>3.81 cm DBH and <11.43 cm DBH) on MOFEP during pretreatment (1994) and most recent sampling (2002) following treatment on select ELTps. Treatments include CC=clearcut (EAM), I=intermediate (EAM), L=No harvest (NHM), U=select (UAM), UG=group openings (UAM).

Treatment	ELTp	Year	Shortleaf pine	Scarlet oak	White oak	Post oak	Black oak
CC	2.1	1994	0	49.4	432.4	0	37.1
		2002	0	210.0	506.6	0	185.3
	3.1	1994	144.1	58.7	163.7	12.4	101.9
		2002	78.2	216.2	129.7	6.2	98.8
	4.1	1994	0	38.8	102.4	18.5	22.7
		2002	0	81.2	95.3	0	61.8
I	2.1	1994	0	12.4	37.1	12.4	0
		2002	0	4.1	12.4	0	0
	3.1	1994	24.7	12.4	105.0	0	16.5
		2002	24.7	6.2	71.0	0	12.4
	4.1	1994	0	61.8	190.3	0	12.4
		2002	0	0	116.1	0	0
L	2.1	1994	18.5	16.5	119.8	64.9	22.2
		2002	6.2	8.2	113.7	68.0	17.3
	3.1	1994	40.2	51.9	111.2	46.3	35.0
		2002	46.3	49.4	117.4	34.0	20.6
	4.1	1994	0	0	103.8	0	0
		2002	0	12.4	82.8	24.7	0
U	2.1	1994	30.9	24.7	120.5	12.4	29.7
		2002	18.5	12.4	88.5	12.4	14.8
	3.1	1994	24.7	46.3	143.0	16.5	6.2
		2002	6.2	6.2	95.3	0	30.9
	4.1	1994	0	18.5	119.4	12.4	12.4
		2002	0	12.4	66.9	0	0
UG	2.1	1994	0	12.4	126.6	0	12.4
		2002	0	0	89.6	12.4	12.4
	3.1	1994	34.6	83.7	174.3	29.7	84.9
		2002	22.2	20.6	137.3	24.7	54.1
	4.1	1994	0	15.4	164.7	0	0
		2002	0	12.4	82.4	0	0

MOFEP. Differences in plot level richness for MOFEP were apparent in every ELTp with at least one treatment effect, but of the hundreds of possible combinations and potential contrasts in the MOFEP project, however only a few were significant (Table 5). ELTp 2.1 had the fewest significant differences, however there were fewer treatments in these ELTps. Treatment differences within ELTps were not consistent. Contrast with clearcuts provided a significant difference in each ELTp except 2.1. In many cases, the significance persisted and was evident throughout the years sampled, e.g. CC-L contrasts on ELTp 3.1 and 4.1. All treatments except control influenced species richness (Table 5).

The MOFEP trajectories were very different than CCMA trajectories for the same ELTps (example trajectory fig. 3). The ELTps separated in the same pattern for each treatment and the ELTps showed patterns of response. All ELTps in NHM are increasing on axis two while in clearcuts, all ELTps decreasing along axis two. In UAM and intermediate harvest, ELTps are showing a similar pattern. ELTps 3.1 and 4.1 are increasing along axis two, while ELTps 2.1 is decreasing along axis two.

Trajectory analysis showed clearcuts had significantly different successional trajectories from NHM for 3.1 and 4.1 Intermediate harvest had significantly different trajectories from NHM for all ELTps except 4.1, while select harvest was significantly different from NHM only for ELTp 4.1. However, intermediate and select harvest differed significantly for 2.1, but group openings trajectories differed significantly from NHM for ELTp 3.1.

Table 3.—P-values of CCMA treatment differences in mean richness by ELTp. Pairwise t-test of year effect on mean plot richness in annually burned and twice-burned plots by ELTp. Year contrast is post-treatment year specified minus pre-treatment year (1997). In 1998, annually burned and burned-twice areas had only been burned once.

ELTp	Contrast year	Annually Burned	Twice Burned
		Pr > t	Pr > t
2.1	1998	0.2674	0.0302
	2001	0.0318	0.933
	2002	0.009	0.1154
3.1	1998	0.7327	0.0379
	2001	0.3185	0.0017
	2002	0.2984	0.4005
4.1	1998	0.2084	0.0522
	2001	0.2797	0.0309
	2002	<0.0001	0.0943
7.12	1998	0.0045	0.1311
	2001	0.3194	0.8966
	2002	0.8211	0.5071

Table 4.— Relative resistance and resilience measures on CCMA, derived from Euclidean distances (ED) between pre- and post disturbance samples in a two-dimensional ordination space. Distances are in standard deviation units.

Treatment	ELTp	Maximum ED (SD Units)	Relative resistance (rank)	ED at final sampling (SD units)	Relative resilience (rank)
Annual	2.1	0.61	3	0.61	3
	3.1	0.52	1	0.52	1
	4.1	0.54	2	0.54	2
twice	2.1	0.34	1	0.34	1
	3.1	0.361	2	0.361	3
	4.1	0.363	3	0.358	2

ELTp 3.1 appeared to have overall high resistance and resilience, while ELTp 4.1 had low resistance and resilience (table 6). On MOFEP, ELTps in clearcuts had a similar resistance and resilience pattern to NHM, while intermediate (EAM), select harvest (UAM) and group openings (UAM) were similar (Table 6). Within clearcuts, ELTp 3.1 had the highest resistance and resilience. In NHM, ELTp 2.1 was the most resistant followed by 3.1 and 4.1. However, the resilience rankings were 4.1, 3.1 and 2.1.

Discussion

CCMA

The short term and immediate effect of fire is a considerable reduction in seedlings and saplings. According to the findings of this study, annual burning may be too frequent to sustain an overstory. Fire did not appear to promote shortleaf pine. Shortleaf pine did not show up in the small sapling class at all on annually burned plots and was reduced on plots that were burned twice. Fire frequency for promotion of shortleaf pine must still be resolved if there is an interest in restoring this once common forest type to the Missouri Ozarks.

Although not reported here, both fire frequencies resulted in increased forbs, legumes and woody shrub and tree species and a decrease in woody vines and graminoid species (Sasseen 2003), a response similar

Table 5.—MOFEP treatment differences in mean richness by ELTp. Harvest occurred 1996. Bold denotes significant difference from pre-treatment species richness. Only treatment comparisons with at least one significant year difference are shown. CC=clearcut (EAM), I=intermediate (EAM), L=No harvest (NHM), U=select (UAM), UG=group openings (UAM).

Treatment	ELTp	Year	Shortleaf pine	Scarlet oak	White oak	Post oak	Black oak
CC	2.1	1994	0	49.4	432.4	0	37.1
		2002	0	210.0	506.6	0	185.3
	3.1	1994	144.1	58.7	163.7	12.4	101.9
		2002	78.2	216.2	129.7	6.2	98.8
	4.1	1994	0	38.8	102.4	18.5	22.7
		2002	0	81.2	95.3	0	61.8
I	2.1	1994	0	12.4	37.1	12.4	0
		2002	0	4.1	12.4	0	0
	3.1	1994	24.7	12.4	105.0	0	16.5
		2002	24.7	6.2	71.0	0	12.4
	4.1	1994	0	61.8	190.3	0	12.4
		2002	0	0	116.1	0	0
L	2.1	1994	18.5	16.5	119.8	64.9	22.2
		2002	6.2	8.2	113.7	68.0	17.3
	3.1	1994	40.2	51.9	111.2	46.3	35.0
		2002	46.3	49.4	117.4	34.0	20.6
	4.1	1994	0	0	103.8	0	0
		2002	0	12.4	82.8	24.7	0
U	2.1	1994	30.9	24.7	120.5	12.4	29.7
		2002	18.5	12.4	88.5	12.4	14.8
	3.1	1994	24.7	46.3	143.0	16.5	6.2
		2002	6.2	6.2	95.3	0	30.9
	4.1	1994	0	18.5	119.4	12.4	12.4
		2002	0	12.4	66.9	0	0
UG	2.1	1994	0	12.4	126.6	0	12.4
		2002	0	0	89.6	12.4	12.4
	3.1	1994	34.6	83.7	174.3	29.7	84.9
		2002	22.2	20.6	137.3	24.7	54.1
	4.1	1994	0	15.4	164.7	0	0
		2002	0	12.4	82.4	0	0

to comparable studies (Arthur and others 1998, Paulsell 1957). However, Paulsell also found an increase in grasses after burning, which did not occur on CCMA. This discrepancy may be due to a shift in species from fire-sensitive grass and sedge species to fire-tolerant species. The fire-tolerant species of grasses and sedges may also need a more open canopy than is resulting from the initial burning on CCMA, which has been burned since 1998. Grass and sedge abundance may increase in future when fire tolerant species gain dominance.

There has been much evidence to suggest that oak dominance is dependent upon fire in oak-hickory forests (Blake and Schuette 2000, Brose and others 2001). Four years after burning on CCMA, relative density of oak species is decreasing. Black oak appears to be the least tolerant of fire, with decreases in relative density in all layers. However, the more fire tolerant, white oak also decreased in the sapling layers for most ELTPs, with the exception of a large increase in the annually burned 2.1. This may be due to sprouting of saplings since all five tree species were reduced on annually burned 2.1 in the small and large sapling layers. White oak relative density increased in the large sapling layer on annually burned 4.1 and twice-burned 3.1, suggesting future recruitment into the overstory, especially given the reduction in competition due to sapling mortality of other species such as black oak. Given differences in fire tolerance between the white oak and black oak group, there will likely be a shift in the future to more white oak and post oak on CCMA.

Table 6.—Relative resistance and resilience measures on MOFEP, derived from Euclidean distances (ED) between pre- and post disturbance samples in a two-dimensional DCA ordination space. Distances are in standard deviation units.

Treatment	ELTp	Maximum ED (SD units)	Relative Resistance (rank)	ED at final sampling (SD (units)	Relative Resilience (rank)
Control	2.1	0.2	1	0.18	3
	3.1	0.52	2	0.16	2
	4.1	0.79	3	0.06	1
Clearcut (EAM)	3.1	0.45	1	0.41	1
	4.1	0.62	2	0.5	2
Intermediate Harvest (EAM)	2.1	0.82	2	0.21	3
	3.1	0.5	1	0.11	1
	4.1	0.87	3	0.12	2
Select Harvest (UAM)	2.1	0.65	3	0.082	2
	3.1	0.243	1	0.07	1
	4.1	0.45	2	0.11	3
Group Opening (UAM)	2.1	0.54	2	0.04	2
	3.1	0.22	1	0.01	1
	4.1	0.63	3	0.28	3

Exploratory DCAs confirmed the importance of ELTPs based on environmental factors but also demonstrated the control of certain disturbances. DCAs showed a separation of ELTPs along axis 1, representing the influence of dolomite on the ground flora vegetation (fig. 3). Dolomite close to the surface results in soils with higher base percentages (35 to >50%) and typically supports a more diverse ground flora (Nigh 2000). Grabner (2002) found that unique assemblages of species resulted from the dolomite located at or near the soil surface. Variable depth to dolomite areas on CCMA (ELTp 7.12) had the highest richness and diversity of all ELTPs and showed distinct separation from other ELTPs on DCAs, indicating a distinct plant community. The effect of fire on the DCA and trajectory indicate that there can be a homogenizing effect and the fire frequency can determine consistent successional patterns. Halpern (1988) found that floras with greater community fidelity resulted in more distinct successional pathways. For CCMA, fire may create conditions suitable to a select suite of vegetation, and therefore trajectories can become more consistent over time.

The resistance and resilience measures correspond with successional theory equating diversity with stability (Lawton 1994). The annually burned sites had higher richness and diversity than the twice-burned and also had higher relative resistance and resilience scores. This substantiates the theory that high levels of diversity essentially buffer the effects of disturbance. ELTp 7.12, which had the highest richness and diversity, had the greatest resistance and resilience. It changes less than other ELTPs and can return to pre-disturbance conditions more quickly.

MOFEP

The increases in richness after harvest (and decrease on control) on MOFEP are similar to other studies and therefore expected (Beese and Bryant 1999, Halpern and Spies 1995, Collins and Pickett 1988). In this study on the MOFEP sites, the first post-harvest sampling season occurred three years after treatment and may have missed the initial response of vegetation to harvest. The immediate response to treatment, therefore, remains unknown.

Change in richness on MOFEP corresponded with intensity of treatment. Clearcuts had the highest richness and the greatest increases, followed by group openings, select harvest, intermediate harvest and

NHM. This pattern suggests a gradient of responses determined by the amount of light reaching the forest floor. When stratifying by ELTp, 4.1 consistently demonstrated changes in richness between pre- and post-treatment. ELTp 4.1 is protected in aspect, with slightly greater moisture levels, so the influx of light due to harvest may have a greater effect than exposed ELTPs.

White oak consistently decreased across treatments except for clearcut 2.1, while black and scarlet oak increased across harvested sites. The small and large sapling layers were dominated by white oak, with the exception of clearcuts, which had large scarlet oak components (table 2). White oaks were dominant in the sapling layers for all ELTPs and treatments including NHM. The oak forests of the Ozark Highlands are generally xeric landscapes where shade tolerant tree species are not major competitors.

Over time, tree species composition may shift to greater white oak density as light levels are reduced. White oak is moderately shade tolerant even in the sapling or seedling stage. Black oak is also moderately shade tolerant, but appears to have less shade tolerance than white oak at the seedling or sapling stage (Kabrick 2002). Black oak is also shorter-lived than white oak and may eventually be replaced by white oak. Scarlet oak, though comprising a significant portion of regeneration in clearcuts, is not expected to have significant in-growth into canopy layers due to its shade intolerance.

Exploratory DCAs on MOFEP data were not as consistent as CCMA. The axes represented a possible light gradient on axis 2 and an influence of dolomite gradient on axis 1, as they did for CCMA. However, plotted successional trajectories for MOFEP were very different from CCMA. There were large variances along axis 2, even for NHM, but the ELTPs separated in the same order along axis 1 across treatments. While there was great variance, successional trends did emerge. All ELTPs on NHM are moving towards a species composition associated with a closed-canopy. The changes along axis 2 for NHM suggest dynamic natural succession occurring with a trend towards more shade tolerant species.

All ELTPs in clearcuts are moving towards species associated with a more open canopy. While the stand as a whole undergoes the long process of stand initiation, the ground flora undergoes rapid floristic change. Gilliam and others (1995) found that the degree to which forest management alters species diversity and long-term succession patterns depends upon the level of “decoupling” of the relationship between understory and overstory. In the Appalachians, single-tree harvest decreased plant species diversity less than clearcutting. Meier and others (1995) found that clearcutting decreased the vernal-herb diversity in the Appalachians, because of modification of the forest floor and eliminating gap-phase succession. It is likely the processes in the Missouri Ozarks are distinct from most areas in the Eastern Deciduous forest.

MOFEP relative resistance and resilience scores were similar in range to the scores on CCMA, but with less consistent patterns. NHM had both the lowest and highest scores for resistance. Clearcuts were more similar to NHM than other harvest treatments in respect to ELTp resistance and resilience values. The microclimatic changes caused by natural successional conditions and/or intense harvest were less pronounced on ELTPs that were more xeric before treatment (i.e. ELTp 3.1).

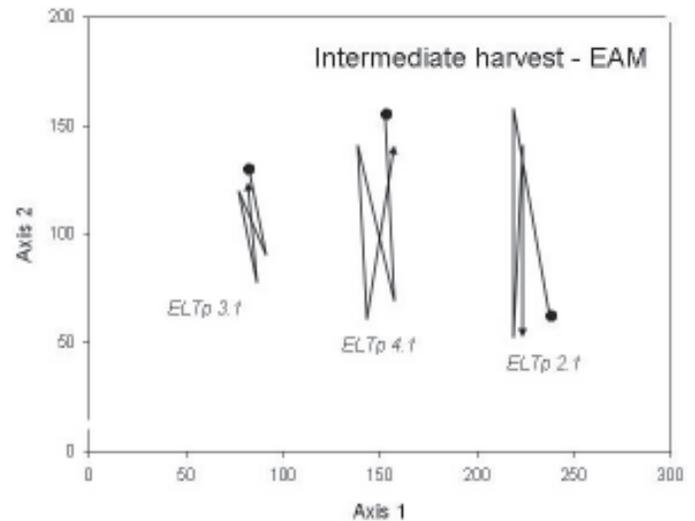


Figure 3.—Detrended correspondence analysis (DCA) ordination through time representing understory communities of ELTPs on MOFEP intermediate harvest sites (EAM). Symbol endpoints indicate predisturbance communities (1995); lines connect the same ELTp in subsequent years (post-disturbance 1999, 2000, 2001, 2002); arrowheads coincide with the final sample and indicate the direction of change through time.

ELTps in intermediate harvest, select harvest and group openings were similar in ELTp response to harvest treatments. ELTp 3.1 had the greatest resistance on these treatments, thereby supporting the diversity/stability theory. It had only moderate to low resilience however. The rankings for these three treatments were more similar to CCMA rankings, in terms of ELTps resistance to disturbance. ELTp 3.1 was consistently resistant and resilient and 4.1 was generally not resistant or resilient. Intermediate harvest, select harvest and group openings may not have been intense enough to cause lasting microclimate changes and may be more dependent on existing abiotic and biotic conditions to determine successional changes.

Successional trajectories in ordination space remained ordered along light (moisture) and geology gradients, suggesting the influences of initial species composition and local environment may persist through catastrophic disturbance. There were also differences between treatments that were very similar (i.e. intermediate and select harvest) that may have more to do with stochastic events such as immediate response of initial vegetation and buried seed or soil disturbance factors. Successional trajectories are determined through an interaction of both deterministic site factors and stochastic disturbance factors. In the short term successional trajectory apparent on MOFEP and CCMA, treatment effects may be most evident, but after initial response to disturbance deterministic site factors may dominate.

In general, the response at MOFEP is very different from the response at CCMA, but as expected since the treatments differ in intensity, frequency and magnitude. The predictability within an ELTp for understanding the effects of treatment may be greater for the harvested plots rather than the burned plots. Using prescribed fire may override the effects of site that could be controlling succession on the MOFEP sites. It is not possible to know if there will long-term differences in species richness, composition, abundance or successional trajectories as a result of management.

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MORTALITY, SURVIVAL, AND GROWTH OF INDIVIDUAL STEMS AFTER PRESCRIBED BURNING IN RECENT HARDWOOD CLEARCUTS

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ABSTRACT.— A study began in 1999 to examine the effect of prescribed fire following final overstory removal on the survival and growth of mixed hardwood regeneration. Two species and one species group were examined: black birch (*Betula lenta*), red maple (*Acer rubrum*), and mixed oak (*Quercus* spp.). Mortality differed by species and height class. Mortality was much higher for black birch (76 percent) than for the other species, regardless of height class. Oak survival was high for all height classes, averaging over ninety percent. Red maple survival was influenced by initial height class. Nearly half of the red maples less than three feet tall were killed by the fires while taller stems survived at a significantly higher rate. Interestingly, height of sprouting stems did not differ among the species in this study. Although sprout height was significantly correlated with initial height, few stems were taller than two feet the first year after the prescribed burn. Thus, prescribed burning reduced the size of larger maple sprouts to a height comparable to that of sprouting oak seedlings.

Introduction

Throughout the Central Hardwood Region, there is increasing interest in and use of prescribed fire to manage oak-dominated (*Quercus* spp.) forests (Yaussy 2000). This incorporation of fire into oak management is driven by increased recognition of the historical role fire played in perpetuating mixed-oak forests and the chronic, widespread difficulty of regenerating this forest type. As an oak regeneration tool, prescribed fire takes two roles. First, it is used in mature stands as a site preparation tool, paving the way for establishment of new oak seedlings by reducing litter depth, increasing light levels, decreasing density of competing understory vegetation, and controlling acorn insect pests. Its second use is in regenerating stands as a release treatment. In this role, fire selects for oak reproduction and against regeneration of other hardwood species based on differences in germination and early growth strategies, causing a shift in species composition between the two groups (Brose and Van Lear 1998; Brose and others 1999). Fire also accelerates early height growth of oak sprouts and improves their form. Currently, most use of fire as a release treatment to alter species composition and enhance oak's competitive status is in a 2-step shelterwood sequence; the shelterwood – burn technique. Unfortunately, there is little research of fire effects on hardwood regeneration after final overstory removal or following a clearcut (Augsburger and others 1987).

Fire in the aftermath of hardwood clearcuts is nothing new. Many historic photographs from the early 1900s show clearcuts in this period were followed by wildfires. The few existing studies suggest that burning of hardwood clearcuts could shift species composition towards oak dominance. Carvell and Tyron (1961) studied a mixed-hardwood sapling stand in West Virginia burned by a spring wildfire. Species composition shifted from cove hardwoods to mixed oak. Brown (1960) and Ward and Stephens (1989) documented the long-term development of mixed-oak forests in southern New England that originated from stand replacing wildfires. They found the stands that originated in the aftermath of severe wildfires had a significantly higher proportion of oak in the overstory than neighboring stands that had not burned. Wildfires in the early 1900s, in conjunction with other crucial forest events (near extirpation of white tail deer (*Odocoileus virginiana*) and loss of American chestnut (*Castanea dentata*), had profound effects on the successional trajectories of many eastern hardwood forests. From these studies, it appears that differences in species composition that start early in stand development, i.e., the stand initiation (seedling/sapling) stage, endure throughout that rotation.

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Table 1.—Initial distribution of regeneration density (stems/acre) by species group and height class.

	Species group	Regeneration height class				Total
		<1 ft	1-3 ft	3-6 ft	>=6 ft	
Star Lake, CT	Oak	225	475	70	68	838
	Maple	125	425	225	698	1,473
	Birch	800	1,475	1,300	930	4,505
	Other	6,150	3,850	2,815	698	13,513
	Total	7,300	6,225	4,410	2,393	20,328
Goodwin, CT	Oak	n/a	1,450	333	38	1,820
	Maple	n/a	675	923	788	2,385
	Birch	n/a	6,125	12,263	2,093	20,480
	Other	n/a	525	135	105	765
	Total	n/a	8,775	13,653	3,023	25,450

While the aforementioned studies provide ample evidence that prescribed fire can increase the proportion of oak, few studies have examined the mechanisms of this change. Some possible mechanisms include: a higher proportion of oak than other species survive a fire, i.e., the stems are not top-killed, a higher proportion of oak than competitors sprout after a fire, and the height growth of sprouting oaks is faster than that of other species. The objective of this small-scale study was to examine for evidence of these causal mechanisms.

Study Areas

Two study areas (Goodwin and Star Lake) were established in Connecticut in formerly fully-stocked, mature, mixed-oak stands that had been completely harvested by the 2-step shelterwood process. The harvesting of these two stands created seedling/sapling stands containing a mix of hardwood regeneration. Before the prescribed fire, reproduction was abundant, tall, and widespread at all sites but primarily comprised of black birch (*Betula lenta*) and red maple (*Acer rubrum*), especially at the Goodwin site (Table 1). The oak component (*Quercus rubra*, *Q. velutina*, *Q. alba*, and *Q. prinus*) was scarce relative to the other species and lagging behind in height. Sites had a moderate ericaceous shrub component consisting of blueberry (*Vaccinium* spp), huckleberry (*Gaylussacia* spp), and mountain laurel (*Kalmia latifolia*).

Methods

To determine preburn seedlings densities, 20 pairs of nested plots were systematically located at both sites to ensure uniform coverage of the stands. The interior of each pair of nested plots was a circular milacre and in it all hardwood regeneration < 4-ft tall was identified to species and tallied by 1-ft height classes. The outer plot was 1/300 acre and in it all hardwood regeneration >= 4-ft tall was identified to species and tallied by 2-ft height classes. Initial regeneration inventories were conducted before the prescribed fires.

Additional sampling was used to study the influence of prescribed fire on mortality and sprouting on individual stems. Oak, red maple, and black birch stems were selected across a range of size classes. Selected stems were tagged, flagged, and the initial height was measured in one foot height classes. A total of 576 individual stems were included in this study. The number of oak, red maple, and black birch was 175, 217, and 184 stems, respectively.

Stems were inspected for mortality and sprouting following the first growing season after each fire. Postburn seedling status was divided into three categories: live top, sprout, and mortality. Live tops were stems that survived the fire, i.e., were never top-killed. Sprouts were stems that had been top-killed but produced at least one new stem from the base. Mortality were stems that were top-killed and failed to sprout. Re-growth, the height of resprouted stems, was measured to the nearest 0.4 inch. Re-growth heights reported here were for two growing seasons at the Star Lake burn site and one growing season at the Goodwin burn site.

Table 2.—Environmental conditions and fire behavior for the prescribed fires.

	Star Lake, CT	Goodwin, CT
Date of Burn	May 5, 2000	April 19, 2002
Time of Burn	11 AM – 2 pm	12 – 2 pm
Burn Area (acres)	19	5
Aspect	North	East
Slope (percent)	0 – 10	0 – 5
Slope Position	Upper 1/3	Upper 1/3
Fuel Model	6	6
Air Temperature (° F)	n/a	66–72
Relative Humidity (percent)	44–60	48–66
Cloud Cover	Sunny	Cloudy
Mid-flame Wind Speed (mph)	3–5	8
Wind Direction	West	Southwest
Days since last Rain	2	2
Flame length (ft)	2–6	1–2
Rate of spread (chains/hr)	~30	4.3

Data were analyzed as a 2x3x3 factorial consisting of 2 sites, 3 species (birch, maple, and oak), and 3 size classes (<3 ft, 3-6 ft, ≥ 6 ft). Analysis of variance with Tukey's HSD test was used to test differences in sprout height among species groups, size classes, and sites with Bonferroni adjusted probabilities used when species group was found to be a significant factor. Differences were judged significant at $p \leq 0.05$. For the individual burns sites, only cells with at least four initial seedlings for each species group/initial height combination were included in the analysis. Only cells with at least twenty stems were included in the analysis of the combined prescribed burn sites.

Differences in seedling mortality and sprouting rates among species groups, study area, initial size class were tested using procedures in Neter et al. (1982, p 325-329). Again, differences were judged significant at $p \leq 0.05$ using Bonferroni adjusted probabilities. To avoid redundancy in data presentation, a table with sprouting rates is not provided. Because almost all stems were top-killed, sprouting rates can be determined by subtracting the mortality rate from 100%.

Fire Behavior

For all burns, fuels consisted of hardwood slash mixed with ericaceous shrubs (Table 2). Connecticut DEP-Division of Forestry personnel conducted the two burns in that state on May 5, 2000 and April 19, 2002. For both, drip torches were used to light a ring fire. This technique consists of igniting a backing fire along the downwind or uphill side to create a black line. Once a secure black line was established, flanking fires were lit along each side. Finally, once adequate black lines were secure on three sides, a head fire was ignited. Fire behavior at the Star Lake burn was recorded by video camera. Post-fire analysis indicated a moderately intense fire with average flame length of 4 feet and a rate of spread (ROS) of 30 chains/hour. Fire behavior at the Goodwin burn was monitored with thermocouples and data-loggers. They recorded that the fire was of low intensity (1-2 ft flame lengths) and slow moving (median ROS of 4 chains/hour).

Results

All of the black birches were top-killed by the prescribed fires. A small proportion of oaks (6 percent) and red maples (2 percent) survived without being top-killed. However, there were no significant differences among the species in the proportion that were top-killed in any of the size classes examined.

Mortality differed among species (Table 3). Mortality was higher for black birch than oak and red maple at both sites. For the combined sites, red maple mortality of stems less than three feet tall was higher than for oak, 45 percent and 8 percent, respectively. Mortality differed among size classes for oak and red maple, but not black birch. For both species, mortality was higher for stems that had been 3-6 ft tall prior to the burn than for > 6 ft tall stems.

Table 3.—Mortality (%) of regeneration following prescribed fire by species group and height class (feet) before the prescribed burn. Within each burn site, mortality values within a given height class (column) that are followed by different letters are significantly different at $p < 0.05$.

Species group	Height class before prescribed burn			Total	Height classes comparison		
	< 3 feet	3-6 feet	>=6 feet		Chi-Square	df	Prob
Star Lake, CT							
Oak	14% a	36% a	0% a	18% a	5.86	2	0.053
Maple	100% b	100% b	0% a	45% b	33.00	2	0.000
Birch	-	82% b	75% b	79% c	0.85	2	0.654
Goodwin, CT							
Oak	6% a	11% a	0% a	7% a	2.92	2	0.232
Maple	37% b	5% a	0% a	9% a	48.00	2	0.000
Birch	86% c	66% b	77% b	75% b	3.68	2	0.159
Combined sites							
Oak	8% a	16% a	0% a	9% a	7.68	2	0.021
Maple	45% b	24% a	0% b	15% a	58.02	2	0.000
Birch	88% c	70% b	77% b	76% b	2.79	2	0.247

This difference in mortality rates among species can be directly attributed to the prescribed burns and not normal mortality. On adjacent unburned control plots, survival of all species was much higher, ranging from 96 percent for oak to 98 percent for red maple.

Differences in sprouting rate were found between the two sites. Sprouting rates for maple and oak were higher at Goodwin than at Star Lake and the difference for maple, 90 percent at Goodwin and 55 percent at Star Lake, was significant. No differences in sprouting rate were detected for birch and oak although the trend for oak was towards lower proportion sprouting at Star Lake than Goodwin.

The proportion of stems that sprouted after the fires differed among species. Oak had the highest rate of sprouting. On average, 90 percent of oak rootstocks sent up a new stem. Conversely, black birch was the poorest sprouter as only 24 percent of top-killed stems produced a new stem. Maple was an intermediate sprouter. At Goodwin, 90 percent of the red maple sprouted, a proportion equal to oak, but at Star Lake only 55 percent of the red maple sprouted – significantly less than that of oak.

Pre-burn stem size had little apparent influence on sprouting rates for oak and black birch. Regardless of initial stem size, oak stems sprouted at between 62 – 100 percent and black birch sprouted at 13 – 30 percent. Only one significant difference was noted for these two species after Bonferroni correction. The sprouting rate for oaks 3-6 feet (86 percent) on the combined sites was lower than for oaks \geq 6 feet tall (100 percent) ($Z=2.55$, $p \leq 0.048$). Red maple did exhibit differences in sprouting rates among size classes. Red maples \geq 6 feet tall sprouted more often than stems < 3 feet, 100 percent and 53 percent, respectively ($Z=8.04$, $p \leq 0.001$). Red maples > 6 feet tall also sprouted more often than 3-6 feet tall, 76 percent ($Z=5.57$, $p \leq 0.001$).

No significant differences in the height of sprouting stems were found among species at either site (Table 4). However, height growth of sprouting stems was correlated with initial height for all three species at both sites ($F=19.6$, $d.f.=4$, $p < 0.001$). Generally, few stems were taller than two feet the first year after the prescribed burn.

Discussion

The results of this study suggest that prescribed burning can increase the amount of oak in the regeneration pool via differences in sprouting rate among species. Oak regeneration is more likely to sprout than other hardwood reproduction, especially as fire severity increases and burns occur in the growing season (Brose and Van Lear 1998). This is evident by comparing sites. The Star Lake fire occurred later in

Table 4. –Mean height (feet) of sprouting stems by species group and height class of sprouting regeneration. Within each burn site, mean values within a given row (species) that are followed by different letters are significantly different at $p < 0.05$.

Species group	Height class before prescribed burn			Total
	<3 feet	3-6 feet	≥ 6 feet	
Star Lake, CT				
Oak	2.2 A	2.6 AB	4.2 B	3.2
Maple	-	-	4.1	4.1
Birch	-	-	5.0 A	4.4
Goodwin, CT				
Oak	1.0 A	1.5 B	2.4 C	1.5
Maple	1.2 A	1.5 A	2.1 B	1.9
Birch	-	1.8 A	2.2 A	1.9
Combined*				
Oak	1.6 A	2.2 B	3.5 C	2.5
Maple	1.9 A	2.2 A	3.1 B	2.4
Birch	-	2.6 A	3.3 A	2.6

*Least square means are presented for the combined sites.

the spring and was a considerably more intense burn than the one at Goodwin based on flame length and ROS. Consequently, mortality (no sprouting) was higher for all species there than at Goodwin (Table 3). Birch and maple mortality ranged from 37 – 100 percent in the smallest size class (< 3 ft). Conversely, oak mortality for the smallest size class was only 14 percent at the hotter Star Lake burn. The same relationship existed between birch and oak in the largest size classes (≥ 6 ft). Few, if any, oaks failed to sprout while mortality averaged 24 percent for birch. Clearly, the birch component at both sites has been considerably reduced while 82 -93 percent of the oak stems are still intact.

While not examined in this study, the differences in mortality rates among species at Star Lake are likely due to differences in germination and early growth strategies (Brose and Van Lear 2004). Acorns have hypogeal germination, i.e., cotyledons remain in the shell and serve as a belowground energy source for seedling development. Black birch and red maple seeds have epigeal germination, i.e., cotyledons emerge and rise above the shell to form the first photosynthetic leaves. This difference in germination strategy places the root collar of oak seedlings, and the accompanying dormant buds, lower in the soil profile than that of black birch and red maple.

This basic difference in germination strategy is accentuated by wildlife. Acorns are routinely buried an inch or more into the forest floor by birds and small mammals, while seeds from black birch and red maple typically are not cached. Thus, an oak seedling generally will have a deeper root collar than a black birch or red maple seedling because of seed burial and hypogeal germination.

Another important silvical difference between oak and black birch/red maple reproduction is the developmental rate of the root system. Upon germinating, oaks send a strong radicle deep into the soil to establish a taproot and emphasize root development over stem growth (Kelty 1988, Kolb and others 1990). Black birch and red maple take the opposite approach; root development is sacrificed to promote rapid stem growth. Thus, oak regeneration usually is shorter than its competitors, but have larger root systems. It is these two silvical characteristics, hypogeal germination and emphasis on root development, and seed burial by wildlife that allows oak regeneration to be favored over reproduction of their competitors in a periodic fire regime.

The second mechanism by which prescribed burning can alter species composition is that oaks may have adaptations that allow a higher proportion of stems to survive a fire with live tops. In this study, some oaks survived the fires while many of their competitors did not. The intact tops of these surviving stems should confer a competitive advantage to them over sprouting seedlings as no height growth was lost to

the fire and, thus, does not have to be recovered. In addition, an intact top does not necessitate the use of stored starch reserves in the root system to produce a new sprout.

Our results do not suggest the height growth of sprouting stems is a mechanism that will cause a long-term change in species composition. No significant differences in height growth were found among species. However, these results are for the first two years only and may not be conclusive in demonstrating a significant difference among species. The major impact of prescribed burning appears to be an equalizing of heights among the species. While there was a wide range in pre-burn heights, and corresponding wide range in crown widths, most seedlings were of a similar size after the prescribed burns.

The equalization of seedling size suggests a final mechanism, albeit a speculative one, that may benefit oak in a periodic fire regime. In a post-fire environment oak seedlings are released temporarily from competition with larger red maple saplings. The sprouting oaks that were less than three feet tall prior to the burn were still shorter than sprouting maples that had been ≥ 6 ft tall before the burn. However, after the burns the average height difference between the species was now less than two feet. As a result of this equalizing of size, most oaks were growing in direct sunlight and were not shaded by neighboring, taller red maple stems. Height growth and root development of oak regeneration growing in full sun is generally better than for oaks growing in shade and for maple (Smith 1983). Therefore, it is likely that a proportion of those oak seedlings that had been growing in close proximity to taller red maples prior to the burn will now be able to maintain height growth rates comparable to red maple and form part of the upper canopy at crown closure. We will continue to monitor these plots to determine the long-term impact of prescribed burning in seedling stands on stand structure.

Acknowledgments

We would like to thank Division of Forestry, Connecticut Department of Environmental Protection for their assistance and cooperation in conducting the prescribed burns and J.P. Barsky for assisting in data collection. This research was partly funded by the Joint Fire Sciences Program and the USDA Forest Service Northeastern Research Station.

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EFFECTS OF FIRE AND THINNING ON GROWTH, MYCORRHIZAL COLONIZATION, AND LEAF ANATOMY OF BLACK OAK AND RED MAPLE SEEDLINGS

Carolyn J. McQuattie, Joanne Rebbeck, and Daniel A. Yaussy[†]

ABSTRACT.—Prescribed fire and/or thinning may improve oak seedling reproduction in forests and limit competitors such as red maple due to altered light or soil-moisture conditions. Because both leaf and root development may be affected by these disturbances, differences in biomass, mycorrhizal colonization, and leaf anatomy between seedlings of black oak (*Quercus velutina*, an ectomycorrhizal species) and red maple (*Acer rubrum*, an endomycorrhizal species) were described and quantified. In spring 2001, four treatments were initiated at Zaleski State Forest, Vinton County, OH: undisturbed control (C); basal area thinned by 29 percent (T); site exposed to prescribed burn (B); or thinned plus burned (T+B). In June and August 2001, four seedlings (two oak, two maple) from six plots per treatment were excavated from three subsites with different moisture levels (two mesic, two intermediate, two xeric; 48 seedlings per species per collection date). Roots were prepared chemically for mycorrhizal analysis; leaf sections were embedded in epoxy resin and examined microscopically. Biomass measurements of all seedling parts were quantified. Endomycorrhizal colonization of maple roots was not affected by treatments. Oak short roots were predominantly ectomycorrhizal; endomycorrhizal structures were observed primarily in June (T+B). In oak leaves, blade thickness and starch grains in chloroplasts (August) were greater in T and T+B than in C or B. Maple leaves, while thinner than oak, had thicker leaf blades in all disturbance treatments compared to C, but less starch in chloroplasts (August). By August, treatments positively affected leaf parameters in oak (leaf area, mass, specific leaf mass) and maple (leaf number, area, mass) probably due to increased light and/or altered soil conditions. Oak seedlings grown on sites with thinning and burning as management tools may gain a competitive advantage by forming both mycorrhizal types early in the season (i.e., additional nutrient uptake) and through increased starch production in leaves late in the season that would improve seedling growth and carbon transfer to roots.

Since the early 20th century, the composition and structure of Appalachian mixed-oak forests has been shifting from oak-dominant to mesophytic dominant species. Many attribute this shift to changes in cutting practices and to an aggressive suppression of fire beginning in the 1920s. Currently, the mid- and understory of these forests contain a high proportion of shade-tolerant mesic species, e.g., red maple (Lorimer 1984), that exhibit low resistance to fire (Abrams 1992) and fewer of the less shade-tolerant oak, e.g., black oak. Written accounts at the time of European settlement (ca. 1800) in southern Ohio suggest that Native Americans used fire to manage forest composition and structure (Hutchinson and others 2003). Oak is drought-tolerant and fire-adapted, with a large root system that resprouts after repeated burnings (Abrams 1996, Brose and others 2001). Oak regeneration on xeric sites is successful, but failures are common on high-quality mesic sites (Abrams and Downs 1990). Fire may reduce understory vegetation and thereby reduce competition; overstory thinning would result in increased light for oak.

Mycorrhizae are symbiotic associations of tree fine roots and certain soil fungi. Oak species are classified as ectomycorrhizal (possessing an external fungal mantle and Hartig net hyphae between cortical cells) while maples are classified as endomycorrhizal (with vesicles, arbuscules, and/or hyphal coils in cortical cells) (Smith and Read 1997). Both types of mycorrhizae generally benefit seedling establishment or growth through increased nutrient uptake and improved water relations in the roots (Allen 1991). The effects of fire on rhizosphere processes such as mycorrhizal associations are scarce and generally limited to coniferous ecosystems (Mah and others 2001).

Improved light conditions in the forest due to disturbances such as thinning or burning generally increase photosynthesis in leaves; photosynthates due to increased carbon fixation can be observed

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microscopically as starch grains in leaf chloroplasts. Reich and Abrams (1990) reported that fire enhanced photosynthesis of *Quercus ellipsoidalis* seedlings. Similarly, Musselman and Gatherum (1969) observed that photosynthesis increased by 50 percent in red oak seedlings grown at 31 percent of full sunlight compared to those grown at 8 percent. Photosynthate allocation to mycorrhizae that is used for fungal growth has been considered a “cost” to the plant (Jones and others 1991). However, light-enhanced photosynthesis may support both mycorrhizal colonization (with no additional cost) and increased seedling growth to improve oak competitiveness and survival. Characterizing seedling biomass, mycorrhizal colonization of fine roots, and leaf anatomy (leaf thickness, starch in leaves) after thinning and burning of forest sites will help us better understand seedling growth and physiology following such treatments. In this study we compare and quantify first-year effects of burning and thinning on growth parameters, mycorrhizal colonization, and foliar anatomy in red maple and black oak seedlings.

Study Area

Zaleski State Forest is a study area within the Ohio Hills site of the U.S. Joint Fire Science National Fire and Fire Surrogates project. Located within the unglaciated Allegheny Plateau, the site has a highly dissected topography with elevations ranging from 200 to 300 m and slopes of 10 to 40 percent. The forest is dominated by strongly acidic (pH=3.6) loamy soils (ultisol: Hapludults) with a C:N ratio of about 20 (Ralph E.J. Boerner, 2003, Ohio State University, pers. commun.). The study area is about 80 ha and is divided into four 20-ha treatment units corresponding to undisturbed control (C), basal area thinned by 29 percent (T), site exposed to prescribed burn (B), and site thinned and burned (T+B). Within each treatment area, ten 50- by 20-m plots were established across a range of moisture conditions (three xeric, four intermediate, and three mesic) determined with an integrated moisture index (IMI) (Iverson and others 1997). In all, 40 vegetation plots were generated.

On the T and T+B sites, the average initial basal area was 27.1 m²ha⁻¹. A commercial thinning from below to 18.6 m²ha⁻¹ basal area was conducted during the winter of 2000; low-intensity surface burns were conducted on 4 April 2001. Air temperature was measured at 25 cm from forest floor with stainless steel temperature probes and logged every 2 seconds with buried data loggers. Maximum air temperatures during the burns ranged from 42.2 to 414.6°C in the T+B unit and 63.8 to 397°C in the B unit.¹

Methods

Seedling Collection

In June and August 2001, two black oak and two red maple seedlings randomly selected from two mesic, two intermediate, and two xeric plots from each of the four treatments (C, T, B, T+B) were excavated (48 seedlings per species per collection). Seedlings were predominantly 1.0 cohort or older. Root systems with accompanying soil were placed in sealed plastic bags in coolers and transported the same day to the USDA Forest Service laboratory at Delaware, OH. Seedling root systems were washed free of soil before processing.

Seedling Growth and Biomass

Seedling height, taproot length, and stem and root-collar diameters were measured. Roots were separated into tap and first-order laterals, and leaves and first-order laterals were counted. Leaf area was estimated using a calibrated LI-3100 Area Meter.² Short roots from each seedling were placed in vials containing FAA (formalin-acetic acid-alcohol) for mycorrhizal assessment. Representative short roots and leaf-blade segments (2 mm²) were placed in 3-percent glutaraldehyde to process for microscopical

¹Iverson, L.R.; Prasad, A.M.; Hutchinson, T.F.; Rebbeck, J.; Yaussy, D. Fire and thinning in an Ohio oak forest: grid-point analysis of the fire behavior, environmental conditions, and tree regeneration across a topographic moisture gradient. In preparation.

²The use of trade, firm, or corporation names in this paper is for the information and convenience of the reader. Such use does not constitute an official endorsement or approval by the U.S. Department of Agriculture or the Forest Service of any product or service to the exclusion of others that may be suitable.

analysis. Following growth measurements and tissue sampling, tissue of each seedling was separated into leaves, stem, and roots, oven-dried at 70°C to constant mass (g dry weight), and weighed. The root/shoot ratio and specific leaf mass (g m⁻²) of each seedling were calculated.

Mycorrhizal Analysis

Red maple. Fine roots were cleared in 10-percent KOH and stained with trypan blue (Phillips and Hayman 1970). Endomycorrhizal structures (hyphal coils, vesicles, arbuscules) were observed under a stereomicroscope, and colonization of maple roots systems was calculated using the grid-line intersect method (Giovannetti and Mosse 1980) using 1-cm intersects marked in a petri dish. Forty-three intersects per root system were evaluated by recording the presence or absence of mycorrhizal structures.

Black oak. Root tips from each root system were inspected microscopically for a fungal mantle or Hartig net (ectomycorrhizal structures). Oak short roots with no obvious fungal mantle were cleared and stained (as for maple roots) to determine whether endomycorrhizal structures were present. Quantification of oak root mycorrhizal colonization was conducted by direct counts of entire root systems (Grand and Harvey 1982) and rating each tip as ectomycorrhizal, endomycorrhizal, or nonmycorrhizal. The proportion of short roots that were mycorrhizal was then expressed as percent ectomycorrhizal tips only or percent ectomycorrhizal plus endomycorrhizal tips.

Leaf Microscopy

Leaf-blade segments from each treatment combination fixed in 3-percent glutaraldehyde were post-fixed in 2-percent osmium tetroxide, dehydrated in a graded ethanol series, and infiltrated with PolyBed-Araldite epoxy resin. Resin-embedded leaf cross sections were photographed under an Olympus BH light microscope, and leaf thickness at three positions per blade was measured for three leaf blades per treatment. Leaf segments were ultrathin-sectioned and examined with a JEOL JEM-1010 transmission electron microscope; leaf mesophyll cells were photographed and two starch grains in chloroplasts from five mesophyll cells in each of three blades per treatment were measured (length x width of 30 grains per treatment).

Statistical Analysis

The effects of treatment, IMI, and their interactions on percent mycorrhizal colonization, leaf thickness, taproot length, number of first order laterals, shoot height, seedling diameter, root-collar diameter, leaf number, area, and biomass, stem biomass, root dry biomass, root/shoot ratio, and specific leaf mass were tested as a two factor ANOVA using SAS General Linear Model analysis (SAS Institute 1999). Effects were considered significant if $p \leq 0.05$.

Results

Seedling Growth and Biomass

Growth and biomass data from red maple and black oak seedlings collected in June (2 months after prescribed fire) showed no significant differences due to treatment or IMI (soil moisture) (data not shown). Although not always statistically significant at $p \leq 0.05$, black oak and red maple seedlings harvested from T, B, and T+B plots in August were more massive than the C seedlings (table 1). In red maple, leaf area per seedling was greater by 106, 180, and 247 percent on B, T, and T+B plots, respectively, compared to controls ($p = 0.02$). Leaf mass ($p = 0.03$), and leaf number ($p = 0.03$) was substantially greater in all treatments relative to controls. Root-collar diameter of maple seedlings from xeric plots were 1.6 times larger than that of seedlings from intermediate or mesic plots ($p = 0.01$). No other significant effects associated with IMI were detected. Borderline significant treatment effects were detected for root collar diameter ($p = 0.06$) of maples. In black oak, total leaf area per seedling was greater by 39, 175, and 140 percent in T, B and T+B plots, respectively ($p = 0.014$), compared to C. Leaf mass did not differ between seedlings from C and T plots, but did increase by 170 percent in B and T+B oak seedlings compared to controls ($p = 0.007$). Specific leaf mass was nearly 20 percent greater in oak seedlings from T and T+B plots relative to controls ($p = 0.05$). Root mass showed borderline significance ($p = 0.06$), as control roots had 53 to 75 percent less mass than roots in all other treatments. IMI had no significant effect on oak growth or biomass measurements, and no significant interaction between IMI and treatments was detected for either species.

Table 1.—Mean (\pm std error)^a of the growth and biomass of black oak and red maple seedlings four months after prescribed fires and/or overstory thinning at Zaleski State Forest, Vinton County, OH

Growth parameter	Treatment			
	Control	Burn	Thin	Thin+Burn
Black oak				
Shoot height, <i>cm</i>	18.24 \pm 2.36	15.60 \pm 2.40	14.03 \pm 2.36	14.03 \pm 2.36
Stem diameter, <i>mm</i>	2.65 \pm 0.19	3.10 \pm 0.20	2.68 \pm 0.19	2.92 \pm 0.19
Root collar diameter, <i>mm</i>	3.02 \pm 0.35	5.59 \pm 0.35	4.52 \pm 0.35	4.57 \pm 0.35
Leaf number	4.25 \pm 0.47	4.92 \pm 0.47	3.58 \pm 0.47	4.91 \pm 0.47
Leaf area, <i>cm</i> ²	53.52 \pm 26.29 a	146.88 \pm 26.29 b	74.28 \pm 26.29 a	128.37 \pm 26.29 b
Leaf mass, <i>g</i>	0.27 \pm 0.10 a	0.72 \pm 0.10 b	0.46 \pm 0.10 a	0.75 \pm 0.10 b
Specific leaf mass, <i>g m</i> ⁻²	4.96 \pm 0.24 a	5.08 \pm 0.24 a	5.96 \pm 0.24 b	5.86 \pm 0.24 b
Stem mass, <i>g</i>	0.51 \pm 0.17	0.72 \pm 0.17	0.62 \pm 0.17	0.60 \pm 0.17
Root mass, <i>g</i>	0.83 \pm 0.33	3.24 \pm 0.33	2.39 \pm 0.33	1.75 \pm 0.33
Taproot length, <i>cm</i>	23.03 \pm 1.83	22.96 \pm 1.83	25.45 \pm 1.83	19.70 \pm 1.83
First-order lateral roots, <i>no.</i>	29.00 \pm 4.19	32.92 \pm 4.19	27.83 \pm 4.19	25.75 \pm 4.19
Root:shoot ratio	1.12 \pm 0.44	2.57 \pm 0.44	2.13 \pm 0.44	1.42 \pm 0.44
Plant mass, <i>g</i>	1.61 \pm 0.53	4.74 \pm 0.53	3.48 \pm 0.53	3.20 \pm 0.53
Red Maple				
Shoot height, <i>cm</i>	9.89 \pm 0.88	8.32 \pm 0.88	12.38 \pm 0.88	11.20 \pm 0.88
Stem diameter, <i>mm</i>	1.55 \pm 0.19	1.67 \pm 0.19	1.84 \pm 0.19	2.06 \pm 0.19
Root collar diameter, <i>mm</i>	1.53 \pm 0.29	2.47 \pm 0.29	2.25 \pm 0.29	2.90 \pm 0.29
Leaf number	4.08 \pm 0.62 a	7.08 \pm 0.62 b	8.75 \pm 0.62 b	11.33 \pm 0.62 c
Leaf area, <i>cm</i> ²	18.78 \pm 6.82 a	38.64 \pm 6.82 ab	52.72 \pm 6.82 bc	65.24 \pm 6.82 c
Leaf mass, <i>g</i>	0.08 \pm 0.03 a	0.16 \pm 0.03 a	0.23 \pm 0.03 ab	0.27 \pm 0.030 b
Specific leaf mass, <i>g m</i> ⁻²	4.57 \pm 0.03	4.10 \pm 0.03	4.38 \pm 0.03	4.10 \pm 0.03
Stem mass, <i>g</i>	0.09 \pm 0.03	0.14 \pm 0.03	0.21 \pm 0.03	0.15 \pm 0.03
Root mass, <i>g</i>	0.15 \pm 0.21	0.48 \pm 0.21	0.78 \pm 0.21	0.58 \pm 0.21
Taproot length, <i>cm</i>	16.97 \pm 2.70	19.08 \pm 2.70	22.14 \pm 2.70	18.83 \pm 2.70
First-order lateral roots, <i>no.</i>	23.83 \pm 2.78	24.58 \pm 2.78	29.25 \pm 2.78	23.50 \pm 2.78
Root:shoot ratio	0.86 \pm 0.46	1.74 \pm 0.46	1.44 \pm 0.46	1.48 \pm 0.46
Plant mass, <i>g</i>	0.32 \pm 0.27	0.78 \pm 0.27	1.21 \pm 0.27	0.99 \pm 0.27

^aLeast square means \pm 1 standard error presented; means followed by a different letter (across a row) significantly different at $p \leq 0.05$.

Mycorrhizal Assessment of Roots

Red Maple. Endomycorrhizal colonization was extensive in maple roots for all treatments (table 2). Percent colonization was significantly greater in the T+B treatment in August compared to T or B alone, but it was not significantly different from the control. The greatest percent increase in infection from June to August (14.2) was in T+B, possibly due to increased light and less competing vegetation. Hyphal coils in cortical cells were the most common endomycorrhizal structure, though vesicles and arbuscules also were present.

Black Oak. Ectomycorrhizal short roots displayed numerous morphotypes (examples: brown smooth, brown fuzzy, black), but these were not identified by species. Roots appearing nonmycorrhizal that were cleared and stained with trypan blue often contained hyphal coils and vesicles (endomycorrhizal structures) that were indistinguishable from those observed in red maple roots. Percent mycorrhizal colonization (table

Table 2.—Mean percent (± 1 std error)^a ectomycorrhizal (EM) and endomycorrhizal (VA) colonization in red maple and black oak seedling roots from undisturbed soil (Control) or exposed to treatments (Thin; Burn; Thin+Burn)

Treatment	Red maple		Black oak			
	June (VA)	August (VA)	June (EM)	June (EM+VA)	August (EM)	August (EM+VA)
Control	61.6 \pm 3.8	69.2 \pm 3.2 ab	69.1 \pm 4.6 a	72.0 \pm 3.9 a	54.4 \pm 4.6 ab	54.4 \pm 4.2a
Thin	59.3 \pm 3.8	62.8 \pm 3.2 b	44.8 \pm 4.6 b	46.2 \pm 3.9 b	60.9 \pm 4.6 a	61.2 \pm 4.2a
Burn	69.1 \pm 4.1	64.4 \pm 3.2 b	53.4 \pm 4.6 b	54.0 \pm 3.9 b	50.2 \pm 4.6 ab	50.9 \pm 4.2a
Thin+Burn	61.8 \pm 3.8	76.0 \pm 3.2 a	48.3 \pm 4.6 b	67.9 \pm 3.9 a	46.6 \pm 4.6 b	50.5 \pm 4.2a

^aMeans followed by different letters within a column significantly different at $p \leq 0.05$.

Table 3.—Mean (± 1 std error)^a leaf thickness (μm) of red maple and black oak seedlings grown in control soil or exposed to treatments

Treatment	Red maple		Black oak	
	June	August	June	August
Control	108.0 \pm 3.3 a	114.8 \pm 3.2 a	155.2 \pm 3.8 bc	129.3 \pm 4.2 c
Thin	120.0 \pm 3.3 b	126.9 \pm 3.2 b	162.0 \pm 3.8 ac	141.8 \pm 4.2 b
Burn	118.4 \pm 3.3 b	130.5 \pm 3.2 b	148.3 \pm 3.8 b	128.0 \pm 4.2 c
Thin+Burn	118.8 \pm 3.3 b	131.6 \pm 3.2 b	169.3 \pm 3.8 a	155.6 \pm 4.2 a

^aMeans followed by different letters within a column significantly different at $p \leq 0.05$.

2) was quantified for ectomycorrhizal (EM) tips only and for the combination of EM plus endomycorrhizal (VA) tips. In June, the percentage of EM tips was highest in control roots, i.e., undisturbed soil had greater early season EM colonization. However, with the addition of VA tips, percent colonization in T+B increased to the level of the control. By August, there was no treatment difference in percent colonization when EM+VA tips were quantified, nor was there an effect attributable to treatment in EM tips alone (table 2) because no treatment differed significantly from the control. It is interesting that ectomycorrhizal colonization between June and August increased in the T treatment (44.8 to 60.9 percent) while the percentage in other treatments decreased. Soil moisture (IMI) did not significantly affect mycorrhizal colonization.

Leaf Anatomy

Leaf thickness was greater in black oak than red maple for each treatment (table 3). Red maple leaves were thicker in August than in June for each comparable treatment, whereas black oak leaves were thinner in August compared with those in June. Black oak leaves were thicker in T and T+B treatments (table 3) due to thicker mesophyll layers (spongy mesophyll plus palisade cells). In August, the percentage of leaf thickness due to mesophyll layers ($n = 12$ measurements per treatment) were: C, 72.8; B, 73.1; T, 80.8; T+B, 78.0. All disturbance treatments resulted in thicker red maple leaves (table 3) compared to controls due to less air space in mesophyll area in the latter (i.e., more tightly packed cells and thinner mesophyll).

Soil moisture index (IMI) did not affect thickness of red maple leaves (table 4). Leaves from black oak seedlings growing in the xeric site (June and August) and intermediate site (August) were thicker than those from the mesic site (table 4). Black oak leaves from the mesic site had a slightly thinner mesophyll cell layer and elongated (rather than rounded) upper epidermal cells that accounted for the overall thinner leaves.

In June, starch grains were found in chloroplasts from all leaves of both tree species. By August, starch grains in chloroplasts of black oak leaves were larger and more abundant in the T and T+B treatments (fig. 1) than in the C or B treatment (fig. 2). Measurement (LxW) of starch grains quantified the larger size in the T and T+B for black oak and in the T treatment for red maple (table 5). Differences in frequency of starch in mesophyll cells in August also were observed: 47 percent of maple mesophyll cells versus 29 percent of oak cells examined had no starch grains present. Although 91 percent of leaf mesophyll cells

Table 4.—Effect of soil moisture (IMI, integrated moisture index^a) on mean thickness (\pm 1 std error)^b ($\frac{1}{4}$ m) of red maple and black oak leaves

IMI	Red maple		Black oak	
	June	August	June	August
Xeric	112.2 \pm 2.9	123.6 \pm 2.8	170.9 \pm 3.4 a	145.5 \pm 3.7 a
Intermediate	117.0 \pm 2.9	129.7 \pm 2.8	157.6 \pm 3.4 b	143.0 \pm 3.7 a
Mesic	119.7 \pm 2.9	124.6 \pm 2.8	147.6 \pm 3.4 c	127.6 \pm 3.7 b

^aIverson and others 1997.

^bMeans followed by different letters within a column significantly different at $p \leq 0.05$.

from oak seedlings grown in the T or T+B contained abundant starch grains, maple leaves from these same treatments had about equal numbers of cells with and without starch grains. In C and B treatments, there were no starch grains in 69 and 41 percent, respectively, of mesophyll cells from leaves of sampled oak seedlings. In red maple leaves from the same treatments, 63 and 37 percent of examined mesophyll cells had no starch grains.

Discussion

Forest disturbance due to burning, thinning or both increased foliar area and mass in red maple and black oak seedlings, probably by increasing the amount of light in the understory. Light increased from 8.7 percent open sky in C plots to 17.8 in T, 14.5 in T+B, and 11.8 in B plots (Todd Hutchinson, 2001, Northeastern Research Station, pers. commun.).

Thicker leaf blades in black oak (T and T+B) were due to increased mesophyll thickness. In other studies, increased leaf-blade thickness in *Quercus petraea* (sessile oak) (Igboanugo 1992) and in black oak (Ashton and Berlyn 1992) was attributed to increased mesophyll thickness. Of the three oak species they examined, black oak was the most drought-tolerant, had the greatest anatomical plasticity, and the highest photosynthetic rates under different light conditions (Ashton and Berlyn 1992). They also reported that thicker anatomical dimensions promote higher water-use efficiency and lower evapotranspirational demands under higher light conditions. This led Ashton and Berlyn to conclude that black oak is sufficiently flexible to adapt to heterogeneous environments and to compete in drought-prone environments.

The abundant starch grains observed in leaf chloroplasts of black oak in August probably is due to high photosynthesis, as reported for black oak (Ashton and Berlyn 1992). Late-season starch production may benefit shoot growth (observed in August) and, although not directly measured in our study, may result in greater transfer of carbohydrate to roots (belowground biomass investment). Growth stimulation of black oak by inoculation with ectomycorrhizal fungi was demonstrated by Daughtridge and others (1986), and additional root carbohydrate might favor mycorrhizal colonization.

The extensive endomycorrhizal colonization of red maple in our study (59 to 76 percent) was similar to or higher than that reported for sugar maple by Cooke and others (1992), Klironomos (1995), and DeBellis and others (2002). Much of the herbaceous vegetation in the understory is classified as endomycorrhizal (Brundrett and Kendrick 1988); therefore, a high level of soil inoculum may account for the high colonization levels in our study. A lack of treatment effect due to thinning on endomycorrhizal colonization also was observed by DeBellis and others (2002); selective cutting had

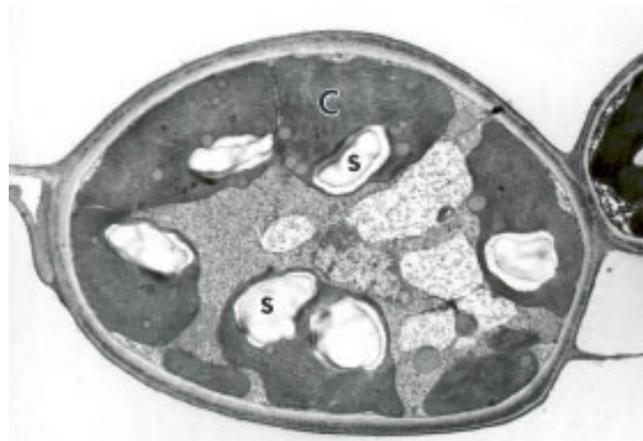


Figure 1.—Spongy mesophyll cell from black oak seedling, B+T treatment, August collection. Large starch grains (S) were observed in chloroplasts (C).

Table 5.—Mean (± 1 std error) starch grain dimensions (when present¹) within leaf mesophyll cells of red maple and black oak seedlings (samples collected in August 2001, about 4 months after a dormant-season prescribed fire)

Species	Treatment	Length		Width
		μm		
Black oak	Control	1.03 \pm 0.08	0.45 \pm 0.06	
	Thin	1.64 \pm 0.08	0.84 \pm 0.09	
	Burn	1.30 \pm 0.10	0.61 \pm 0.07	
	Thin+Burn	2.09 \pm 0.06	0.77 \pm 0.05	
Red Maple	Control	1.43 \pm 0.18	0.36 \pm 0.02	
	Thin	1.84 \pm 0.12	0.47 \pm 0.03	
	Burn	1.31 \pm 0.08	0.38 \pm 0.04	
	Thin+Burn	1.26 \pm 0.07	0.51 \pm 0.07	

¹Five mesophyll cells per leaf were randomly chosen for measurement of starch grains.

no negative effect on mycorrhizal community structure because of the rapid regeneration of mycorrhizal hosts and minor levels of soil disruption. Klironomos (1995) reported that soil type can affect variability in endomycorrhizal structures; hyphal coils, the most common endomycorrhizal structure observed in our study, also was the most common structure found in sugar maple roots by Klironomos in podzolic (acidic) soil.

Oak species generally are considered as ectomycorrhizal (Dickie and others 2001); however, we observed both ectomycorrhizal and endomycorrhizal associations on black oak roots. Endomycorrhizal associations on predominantly ectomycorrhizal oak roots have been reported (Grand 1969, Watson and others 1990, Dickie and others 2001), especially in extremely wet or dry soils (Watson and others 1990).

Endomycorrhizal infection of predominantly ectomycorrhizal host plants might be increased in the presence of abundant endomycorrhizal hosts (Smith and others 1998, Dickie and others 2001), as occurred in the understory at Zaleski State Forest. The percentage of endomycorrhizal colonization of black oak roots decreased in August. Ectomycorrhizal fungi might be able to displace endomycorrhizal fungi (Lodge and Wentworth 1990, Chen and others 2000), and this likely occurred in our late-season sampling across all treatments. Additional research is needed to determine whether there is a functional significance (e.g., increased nutrient uptake) with respect to the dual mycorrhizal colonization of black oak roots. In *Salix repens* (a dual mycorrhizal plant), the endomycorrhizal association was beneficial over the short term (increased P uptake), though the ectomycorrhizal association benefited growth over the long term (van der Heijden 2001).

Thinning and/or burning favorably affected oak seedling leaf area and mass, leaf thickness, the presence of leaf starch in August, and possibly ectomycorrhizal colonization, probably due to an increase in the amount of light reaching the forest floor. The flexibility of oak roots in mycorrhiza formation (colonization by both endo- and ectomycorrhizae) might explain a portion of its success in resource-limiting or dry sites. Continuing to monitor stand management using thinning and burning

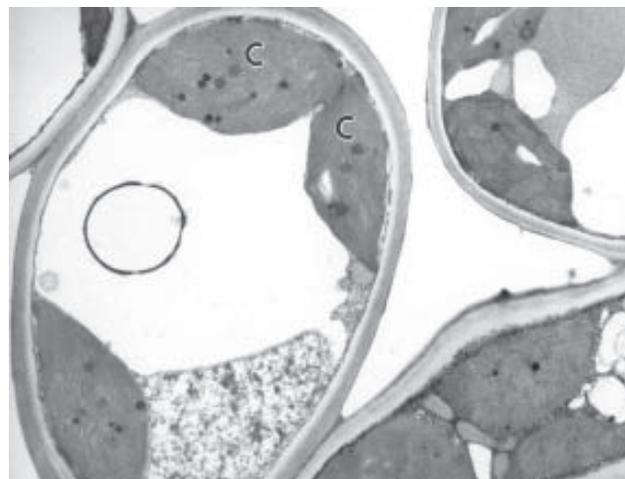


Figure 2.—Spongy mesophyll cells from black oak seedling, B treatment, August collection. Chloroplasts (C) had few starch grains.

prescriptions will provide additional data on oak competitiveness over the long term in mixed hardwood forests.

Acknowledgments

This research was funded by the National Fire Plan. Although the authors received no direct funding from the U.S. Joint Fire Science Program (JFSP) for this research, it could not have been accomplished without JFSP support of existing Fire and Fire Surrogate project sites.

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OAK STUMP SPROUT HEALTH AND SURVIVAL FOLLOWING THINNING AND PRESCRIBED BURNING IN SOUTHERN OHIO

Robert P. Long, Joanne Rebbeck, and Zachary P. Traylor[†]

Oak stump sprouts are an important component of regenerating oak stands in central hardwood forests. Research initiated at three locations (REMA, Tar Hollow, and Zaleski) in southern Ohio in 2001 is testing whether thinning (T), prescribed burning (B), and the combination of thinning and burning (T+B) will produce stands with sufficient advanced oak regeneration to ensure an oak component in future stands. Stands were thinned from below (29 percent reduction in basal area) 4 months prior to prescribed burning in spring 2001. Oak stump sprouts were surveyed starting in 2001 (n=117), on T and T+B treatments at each site to determine stump diameter, number of sprouts, dominant sprout height, dominant sprout basal diameter, mortal status, and causal agents, if any, affecting dominant sprout health. In spring 2002, additional stump sprouts were added to the survey (total n=318), and all were evaluated in spring and late summer 2003. Oaks were combined as white oaks (n=224), *Quercus alba*, and chestnut oak, *Q. montana*; and red oaks (n=94), scarlet oak, *Q. coccinea*, and black oak, *Q. velutina*.

Mean stump diameter for the white and red oaks did not vary between treatments ($P > 0.70$); however red oaks mean stump diameter was slightly larger, 36.3 cm compared with white oaks mean stump diameter, 33.7 cm. Mean number of sprouts per stump was not affected by treatments for red or white oaks ($P > 0.07$). White oaks produced an average of 17.1 sprouts per stump while red oaks produced an average of 13.4 sprouts per stump. Dominant sprout height and diameter growth were evaluated with a repeated measures analysis of variance. There were no treatment differences ($P > 0.19$) for height or diameter growth of red oaks from 2001 through 2003. However, for white oaks, dominant sprout height was significantly ($P \leq 0.02$) greater in T+B treatments, 199 cm, compared with T treatments, 176 cm, in 2003. Dominant stem basal diameter growth for white oaks was also significantly ($P \leq 0.02$) greater in the T+B treatments compared with T treatments in both 2002 and 2003. Sprout mortality from 2001 to spring 2002 was 22% and independent of treatment. Mortality was only 4.9% by spring 2003 and an additional 2.6% by August 2003 with no relationship to treatment. Five categories describing the most prominent causal agents affecting sprout health were identified: deer browsing, pathogens, insect injury, powdery mildew fungi, and insect injury + browsing. Frequency of agents affecting sprout health was unrelated to treatments. Powdery mildew fungi infected white oaks in 2001 and 2002, but frequency of infection was unrelated to treatments. Frequency of injury to sprouts from deer browsing decreased from a high of 26% in spring 2002 to 3% by August 2003 as sprouts grew past heights where they are readily browsed.

Acknowledgments

We thank David Hosack, Kristy Tucker, Brad Tucker, and Bill Borovicka for assistance in data collection and processing. This is a contribution of the National Fires and Fires Surrogate Project (FFS), funded by the U.S. Joint Fire Science Program.

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SURVIVAL OF HARDWOOD SEEDLINGS AND SAPLINGS ONE YEAR FOLLOWING A PRESCRIBED FIRE AND OVERSTORY THINNING IN SOUTHEASTERN OHIO

Joanne Rebbeck, Robert P. Long, and Daniel A. Yaussy[†]

Prescribed fires in combination with thinning are being investigated as silvicultural tools to improve oak regeneration in mixed-oak forests of southeastern Ohio. Stands were thinned from below (29% reduction in basal area) 4 months prior to spring 2001 burns at the Raccoon Ecological Management Area, at the Tar Hollow and Zaleski State Forests. Each site was divided into four treatments of 20 ha each: control (C), burn only (B), thin only (T), or thin plus burn (TB). Within each treatment area, ten 50- by 20-m plots were established across a range of moisture conditions (three xeric, four intermediate, and three mesic) determined with an integrated moisture index (IMI) (Iverson and others 1997). Within each plot, the health and mortal status of 10 seedlings (three size classes: small, <10 cm tall; medium, 10-50 cm tall; and large, 50.1-139 cm tall) and 10 saplings (three size classes: small, >140 cm tall and <3 cm DBH; medium, 3-6 cm DBH; and large, 6.1-9.9 cm DBH) were evaluated every June and August since 2000. The 1080 seedlings monitored were either oak, red (*Quercus rubra* and *Q. velutina*) and white (*Quercus alba* and *Q. prinus*), or hickory (*Carya* spp.); while the 1080 saplings were either maple (*Acer rubrum* L. and *A. saccharum* Marsh.), blackgum (*Nyssa sylvatica* Marsh.) or hickory.

At 14 months postburn, sapling mortality was greatest in TB units, while small oak seedling mortality was greatest in B units. In B units, small white oak seedling mortality was slightly higher (67%) than red oaks (50%). However, mortality of medium-sized red oak seedlings (28%) and white oaks (37%) did not vary between B and TB plots. Among saplings, small and medium maples (red and sugar) had the greatest amount of mortality, 27 and 22% in TB plots, respectively. In contrast, mortality of large maple saplings (3.1%) was similar to equivalent-sized blackgum and hickory (4%). Small hickories were the most prolific sapling resprouters (86%) and large blackgum the least (9%). Maple sapling resprouting ranged from 26-50%. The percentage of medium and large saplings with declining crown vigor (50% or more dieback) ranged from 24-79% in the B and TB units. Large blackgum and maple (31%) saplings appeared to have fewer individuals with declining crown vigor compared with medium (72%) and large (67%) hickories.

These results demonstrate that thin-barked red and sugar maples are more susceptible to fire than thicker-barked blackgum and hickory only when stems are small (<6 cm DBH). This suggests that a single low-intensity dormant season prescribed fire is not sufficient to remove larger non-desirable species such as maple and that more aggressive measures such as repeated higher intensity fires combined with herbicide treatments maybe needed.

Acknowledgments

We thank David Hosack, Zachary Traylor, Mary Ann Tate, Kristy Tucker, Brad Tucker, Ben Crane, Robert Ford, Justin Wells, Jeff Matthews, and Lisa Pesich for assistance in data collection and processing. This is a contribution of the National Fires and Fires Surrogate Project (FFS), funded by the U.S. Joint Fire Science Program.

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NITROGEN LEACHING BELOW RIPARIAN AUTUMN OLIVE STANDS IN THE DORMANT SEASON

Jennie M. Church, Karl W. J. Williard, Sara G. Baer, John W. Groninger, James J. Zaczek[†]

ABSTRACT.—Our research objective was to determine if excess nitrate (NO_3^-) and ammonium (NH_4^+) were leaching below the rooting zones of autumn olive stands during the dormant season. Autumn olive is a nitrogen fixer, through a symbiotic relationship with actinomycetes of the genus *Frankia*. It is an exotic woody shrub that was promoted for wildlife habitat but has become naturalized and is difficult to eradicate. Suction lysimeters were installed in plots of autumn olive and adjacent open field plots in three southern Illinois riparian sites. Soil water samples were collected every two weeks from January 2003 to March 2003 and analyzed for nitrate and ammonium. Results showed significantly greater nitrate leaching during the dormant season under autumn olive than in open fields representative of early successional species composition. Mean nitrate-N concentrations were 16.84 mg L^{-1} under autumn olive compared to 1.01 mg L^{-1} under open field conditions. Ammonium was not significantly different between the two plot types. In watersheds where autumn olive is a common riparian plant, the observed significant nitrate leaching could have important water quality and site productivity implications, suggesting the need for vegetation management.

Autumn olive (*Elaeagnus umbellata* Thunb.) is a nitrogen fixing woody shrub native to Japan, China and Korea (Rehder 1940). As a nitrogen fixer, it reduces atmospheric nitrogen (N_2) to ammonia (NH_3) for use by plants, through a symbiotic relationship with the actinomycete (a filamentous bacteria) *Frankia*. This symbiotic relationship provides a source of energy to the microorganism and a supply of nitrogen to the autumn olive plant.

Autumn olive was first cultivated in the United States in the 1830s (Redher 1940). It is an open-growing woody shrub that can reach a height of ~6 meters and a spread of ~7 meters and has been promoted for a wide range of uses including food and cover for wildlife, nectar for bees, and erosion control (Allen and Steiner 1972, Hayes 1976). Interplanted autumn olive was also found to increase height and volume of the hardwood species black walnut (*Juglans nigra*) (Funk et al. 1979, Geyer and Rink 1998). Autumn olive was planted extensively in Illinois in the 1960s and 1970s and has become naturalized across much of the eastern United States. This exotic species has spread from its original planting sites, with old-field and disturbed sites as optimal habitat for reproduction (Nestleroad et al. 1987). Its rapid spread is attributed in part to seed dispersal by birds that are attracted to the fruit (Nestleroad et al. 1987). Attempts to control the plant by herbicide have been made with limited success (Edgin and Ebinger 2001).

In Ontario, Canada, Catling et al. (1997) found the plant went from unknown as a wild plant, to a rapidly spreading weed within 10 years. The shrub has the potential to establish on natural sandy openings such as prairie, savanna, barren, dune and shore communities, which are in need of protection in Canada (Catling et al. 1997).

Based on our knowledge of the literature, N fixation rates by autumn olive have not been quantified. However, autumn olive is an actinorhizal nitrogen fixer like red alder (*Alnus rubra* Borg.). Red alder has been planted in the Pacific Northwest in conifer stands, especially Douglas-fir (*Pseudotsuga menziesii*) and shown to increase biomass and root systems of the Douglas-fir (Brozek 1990). Red alder N fixation rates have been found to range from 50 to $200 \text{ kg ha}^{-1} \text{ y}^{-1}$ (Cole et al. 1978, Binkley et al. 1992).

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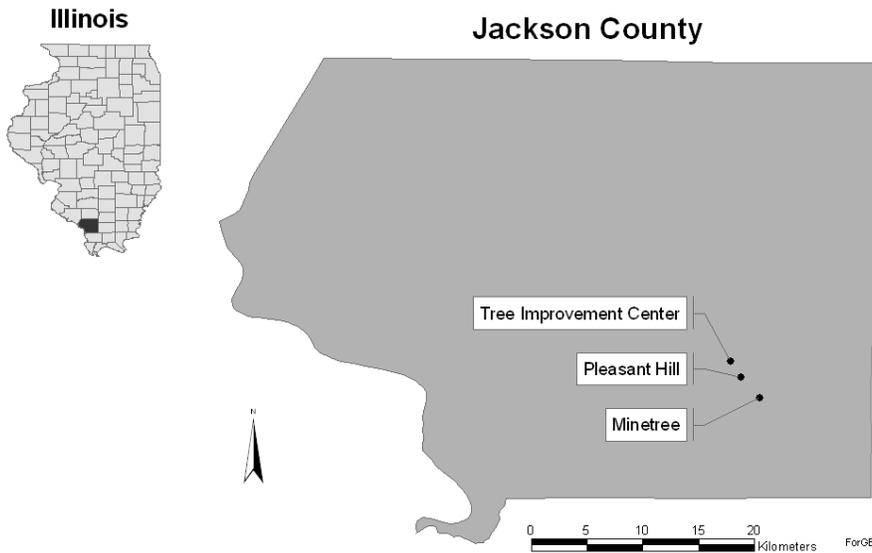


Figure 1.—Study location in Jackson County, Illinois.

Nitrogen fixing species can influence soil acidity and leaching of nutrients. Van Miegroet and Cole (1984) found increased nitrification that occurred under a red alder stand was a source of soil acidification, even more so than atmospheric deposition. Base cations can leach through the system with mobile NO_3^- as their companion ion. The release of cations from the soil decreases the nutrient availability of the site and may have implications for long-term site productivity. The proton generation generally results in increased acidification in the upper most soil horizons (Van Miegroet et al. 1988).

Studies have found that N-fixing plants like red alder can contribute excess NO_3^- to the soil and ground water. These excess nutrients can leach into streams (Van Miegroet and Cole 1984). Nutrient pollution negatively impacts water quality and biota in the stream and is the cause of major environmental problems like hypoxia in the Gulf of Mexico (Rabalais 2002). The rapid spread of autumn olive and its ability to fix nitrogen may have important downstream water quality implications.

This research compared the amount of nitrogen in soil water under autumn olive and old-field sites. The objective was to investigate whether autumn olive, as a nitrogen-fixing plant, was producing excess nitrogen that was leaching below the rooting zone.

Study Sites

The study was conducted on three riparian sites on land owned by Southern Illinois University at Carbondale in Jackson Co., IL. (fig. 1). The sites were chosen due to their high density of autumn olive shrubs and the occurrence of old field habitats. Soil properties differed among, but not within, sites (table 1).

The Minetree Road (MT) riparian site was a pasture until 1990, and is not currently in agricultural use. It was dominated by autumn olive, black locust (*Robinia pseudoacacia*), briars (*Rubus spp.*) and various grasses. The plots were located upslope from an intermittent stream and did not experience any flooding during the study period.

The Pleasant Hill Road (PH) riparian site contained autumn olive and mixed hardwood species and was once grazed. It is presently fenced from an adjoining pasture. The riparian plots were adjacent to a stream that was disconnected from its floodplain through severe downcutting; no riparian flooding occurred during the study period.

The Tree Improvement Center (TIC) riparian site was originally owned by the U.S. Forest Service and contains experimental stands of black walnut and numerous other hardwood species. The plots were located upslope from an intermittent stream and were not flooded during the study period.

Table 1.—Study site characteristics.

	Site		
Location	Minetree Rd (MT)	Pleasant Hill (PH)	Tree Improvement Center (TIC)
	Lat. 37° 41' N	Lat. 37° 41' N	Lat. 37° 42' N
	Long. 89° 14' W	Long. 89° 15' W	Long. 89° 16' W
Soil Classification	<i>Oxyaquic Fragiudalf</i>	<i>Typic Fluvaquent</i>	<i>Oxyaquic Fragiudalf</i>
Soil Texture Class	Loam	Silt loam	Silt loam
Soil Organic Matter	3.0%	2.1%	2.2%
Autumn Olive Plots			
Biomass	11.22 kg m ⁻² ±(2.64)	10.01 kg m ⁻² ±(4.46)	5.67 kg m ⁻² ±(0.50)
Average Age (yrs)	13	11	7

Each site contained two plots in the same landscape position, one dominated by autumn olive and the comparison plot in an open field condition. The open field condition was defined as no autumn olive plants present, and dominated by herbaceous vegetation, primarily grasses. Autumn olive biomass (table 1) was predicted from stem diameter at 5 cm above ground level (Thakur et al. 1993). The plots measured 8.4 meters by 4.8 meters. A buffer width of 0.61 meters was incorporated on the inside of the plot boundary, and the remaining area was divided into 18 equal subplots (1.2 meters by 1.2 meters), 6 of which were randomly chosen for lysimeter installation.

Methods

The lysimeters were composed of PVC pipe with a porous ceramic cup and measured 61 centimeters in length. They were installed at a soil depth above the permanent water table, as indicated by the occurrence of mottling (Mitch and Gosselink 1993). The lysimeters were capped with rubber stoppers and put under 60 centibars of tension to draw soil water in through the porous cup.

In October 2002, the lysimeters were installed following the procedure given by Soil Moisture Equipment Company (Santa Monica, CA). Great caution was used to ensure no surface organic matter was in the backfill and that the backfill was of the same horizon in the soil profile. A 2-3 cm layer of Bentonite clay was placed in the bottom of the wells at the TIC and MT sites to prevent groundwater upwelling. It was not used at the PH site because no mottling was observed at that site. Bentonite clay was also placed around the lysimeter at the surface at all sites to prevent direct infiltration of surface water around the sides of the lysimeter.

When the lysimeters were installed, they were capped and immediately put under tension at 60 centibars. They were flushed three times in November and December 2002, since nitrification generally increases following soil disturbance (Paul and Clark 1996). Flushing consisted of evacuating and discarding the soil water and restoring tension. Tree shelters were placed around the lysimeters at the three sites in January and February 2003, to prevent deer damage.

Bi-monthly samples were collected from January 2003 until March 2003 for a total of six sample dates and 182 samples. The soil water samples were transferred from the lysimeters via suction into a flask and then poured into acid washed 125 ml plastic sample bottles, which were kept on ice during transport to the laboratory for analysis. The flask was washed in the field with de-ionized water between samples. All lysimeters were returned to 60 centibars of tension.

In the laboratory, samples were analyzed for ammonium (NH₄⁺) (mg L⁻¹) on a Hach 4000v Spectrophotometer (Loveland, CO) using the Nesslerization method (APHA 1992). The samples were analyzed for nitrate (NO₃⁻) (mg L⁻¹) on a Dionex Ion Chromatograph (Sunnyvale, CA).

The soil water concentrations from the subplots within each plot were averaged prior to analysis. Nitrate-N concentrations were analyzed as randomized complete block design with repeated measures

Table 2.—Mean soil water nitrate-N and ammonium concentrations for each site by date.

Date	Mean Nitrate-N (mg L ⁻¹)		Mean Ammonium-N (mg L ⁻¹)	
	Autumn Olive	Field	Autumn Olive	Field
		Minetree Site		
January 4	* ¹	*	*	*
January 21	12.80	0.22	0.09	0.19
February 4	14.68	1.93	0.25	1.66
February 18	12.07	0.00	0.14	2.17
March 4	34.35	0.10	0.17	2.13
March 20	13.04	0.05	0.10	1.47
		Pleasant Hill Site		
January 4	32.62	2.62	0.34	0.14
January 21	18.57	*	0.19	*
February 4	36.24	1.17	0.24	0.09
February 18	31.95	0.86	0.15	0.15
March 4	27.28	4.17	0.26	0.17
March 20	30.71	1.72	0.09	0.07
		Tree Improvement Center Site		
January 4	2.14	0.10	0.23	0.09
January 21	4.40	0.12	0.11	0.17
February 4	2.47	0.08	0.07	0.03
February 18	1.26	0.08	0.14	0.05
March 4	8.32	0.07	0.14	0.19
March 20	5.17	0.04	0.13	0.03

¹No samples taken due to deer damage

over time within each experimental unit. The data were analyzed using the mixed model procedure in SAS (SAS Institute 1999). Nitrate data were log transformed to obtain normality. Ammonium data were not examined statistically due to the large number of zero values.

Results and Discussion

The mean soil water nitrate-N concentration was significantly (16.7 times) greater in the autumn olive plots than the open field plots (fig. 2). The results are consistent with studies of red alder stands, where excess nitrogen leaching was observed via soil water collection (Cole et al. 1978, Van Miegroet and Cole 1984). Cole et al. (1978) found 2.2 kg ha⁻¹ yr⁻¹ of nitrogen lost through leaching in red alder stands compared to 0.6 kg ha⁻¹ of nitrogen lost through leaching in Douglas-fir ecosystems. Van Miegroet and Cole (1984) found that there was approximately 50 kg ha⁻¹ yr⁻¹ of nitrate-N leached beyond the 40 cm soil depth in red alder stands.

Soil water nitrate-N concentrations in the autumn olive plots at the MT and PH sites were consistently above the 10 mg L⁻¹ drinking water standard set by the USEPA. Nitrate-N values at MT ranged from 12.07 to 34.35 mg L⁻¹ and the values at PH ranged from 18.57 to 36.24 mg L⁻¹. The TIC site autumn olive plot values were always lower than 10 mg L⁻¹ (table 2).

At the PH field plots, soil water nitrate-N concentrations were generally higher than the MT and TIC field plots. Currently, the PH site contains a livestock-grazed pasture. Significant nitrate leaching can occur from urine and fecal spots in grazed pastures in the humid United States (Stout et al. 1997). The MT site has not been used for livestock pasture in at least 13 years, and the TIC site has not been cultivated or pastured for at least 40 years, when the center was established.

Date of sampling was not a significant effect in our model ($p = 0.09$). There was a strong main effect of vegetation type on nitrate-N concentrations below autumn olive stands across all sampling dates (fig. 3). There is evidence that nitrogen fixation varies seasonally, with the highest rates occurring when soil

temperatures reach 25°C, normally during the growing season (Zitzer et al. 1989). The nitrate-N that leached during the dormant season may have originated from residual nitrogen fixed during the previous growing season or reduced, but still active, N fixation during the dormant season.

On average, ammonium-N concentrations were lower than nitrate-N concentrations in both vegetation types (table 1). Much of the ammonium generated by N fixation was likely immobilized by heterotrophic microbes and plants or converted to nitrate by nitrifying bacteria (Paul and Clark 1996). Also, ammonium does not readily leach due to its positive charge and incorporation into clay lattices (Paul and Clark 1996).

Conclusions

Invasive plants are known to degrade ecosystems throughout the United States by excluding native vegetation. Our results demonstrate an additional negative ecosystem consequence of invasive species, excess nitrate leaching, and should provide further justification for the institution of policies to address this issue. Nitrate export from autumn olive infested sites could contribute to eutrophication of downstream water bodies especially in saline environments (estuaries) (Rabalais et al. 2002). Eutrophication and subsequent hypoxia in the Gulf of Mexico is a critical issue facing the agricultural and marine fisheries communities in the eastern and southern United States.

Acknowledgments

This research was funded by the U.S. Department of Agriculture (USDA) McIntire Stennis Cooperative Forestry Program, the state of Illinois through the Illinois Council on Food and Agricultural Research (C-FAR) Water Quality Strategic Research Initiative, and the Department of Forestry at Southern Illinois University Carbondale. We thank Mike Bill, Christopher Blattel, Roger Haschemeyer, Benjamin Snyder and Chad Yocum for their assistance in the field.

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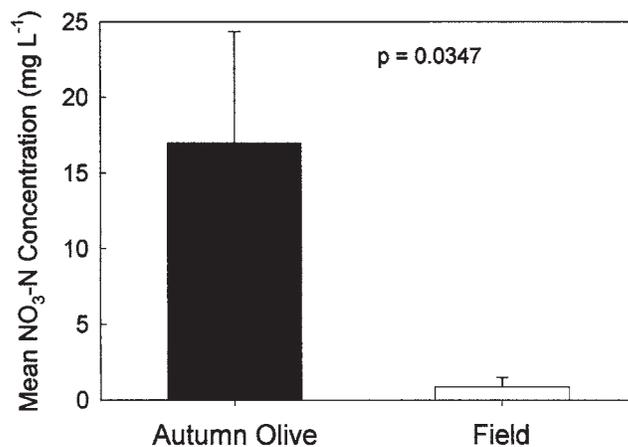


Figure 2.—Mean soil water nitrate-N concentration in autumn olive and open field plots across three riparian sites in southern Illinois (January 4, 2003-March 20, 2003). (Error bars represent ± 1 standard error)

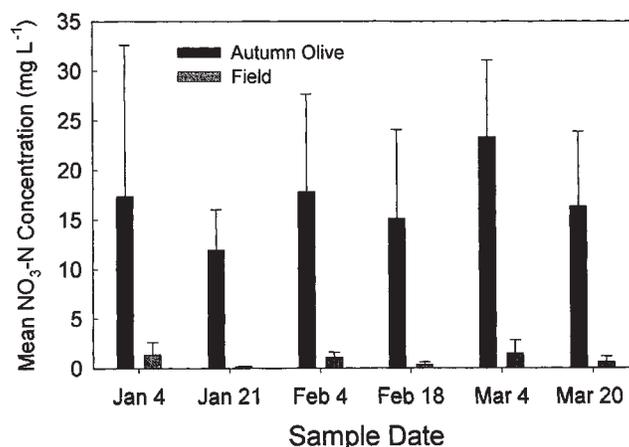


Figure 3.—Temporal variability of mean soil water nitrate-N concentrations in autumn olive and open field plots across three riparian sites in southern Illinois. (Error bars represent ± 1 standard error)

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CHANGES IN SOIL ORGANIC CARBON CONCENTRATION AND QUANTITY FOLLOWING SELECTION HARVESTING

Robert L. Ficklin, John P. Dwyer and R. David Hammer[†]

ABSTRACT.—Timber harvesting operations are known to influence the physical properties of forest soils, especially soil density. However, changes in soil chemistry also may occur as a result of perturbation. Two selection harvests were monitored— one using skidders and one using draft animals— to compare changes in soil organic carbon (SOC) 9 to 12 months following harvest operations. Bulk density changes also were measured, so that changes in SOC mass could be quantified. Minimal soil compaction in primary and secondary skid trails for both harvesting systems was observed, but the extent of soil compaction and scarification differed by a factor of three between systems. Furthermore, compaction from soil sampling confounded precise quantification of skidding compaction. Carbon concentrations were not different between skid trails and control areas on either site when surface and subsurface horizons were considered, but carbon content was higher in the subsurface horizons of the animal-skidded site. SOC quantities increased negligibly in the skid trails of both sites, which was likely due to decomposition of fresh logging slash in contact with scarified soil. Higher bulk densities and lower rock contents in skid trails compared with controls also may have contributed to the increased carbon content per unit volume, since coarse fragments accounted for 20 to 40% of sample volume on both sites.

Introduction

Maximizing private and societal benefits from forest resources often involves timber management. However, some forest management policies limit or prohibit harvesting in sensitive or recreational areas. While the use of low productivity harvesting systems, such as animal skidding, is not appropriate for large-scale timber production, there is a niche market in many states for such operations. Furthermore, if animal skidding does reduce the negative soil impacts associated with logging, it may be possible and preferable to permit logging in forest tracts not well suited for mechanical logging.

Public concern for perceived environmental degradation and forest mismanagement often has been based on the visual or aesthetic changes to forest ecosystems; however, unseen changes due to disturbance may be of greater significance for the long-term sustained productivity of the ecosystem. Traditional forest harvest techniques have the potential to reduce site productivity through nutrient loss and altered soil properties (Reisinger and others 1992). Off-site environmental degradation following timber harvesting may result from nutrient and sediment loading of perennial drainage networks. Soil erosion occurs for several years after harvesting, so the effects of timber harvesting in deciduous forests are best summarized in reviews of long-term research (Likens and others 1977, Swank and Crosley 1988).

To address public concerns about timber harvesting, public land management agencies such as the USDA Forest Service and the Missouri Department of Conservation have started to emphasize the use of selection harvesting techniques instead of clearcuts. Selection cutting offers promise for reducing the losses of soil and soil nutrients, but the dynamics of soil organic carbon [SOC] cycling are unknown under selection cutting regimes. By examining the effects of disturbance on SOC, it will be possible to better model the relationship of ecosystem perturbation with site productivity and global climate change (Anderson 1992, Schlesinger 1990).

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Soil organic matter and soil compaction are perhaps the most important determinants of future forest productivity. Understanding the interrelationships of these key parameters is essential to preserving ecosystem integrity (Powers and others 1990). Data on the cycling of carbon under selection harvesting schemes are needed to provide information on the magnitude, distribution and dynamics of the SOC pool. Quantifying soil organic matter in forested ecosystems is difficult because of the number of factors affecting its distribution.

Temporal and spatial heterogeneity within natural systems can be large (Wilding and Drees 1983). Separating spatial from temporal variability is difficult and is dependent upon scale, techniques and perspectives (Anderson 1972, Hammer and others 1987). Furthermore, the quantification of SOC pools is confounded by difficulties in the measurement of bulk density, especially in soils with high coarse fragment contents (Kern 1994). While skeletal or near-skeletal soils present problems for bulk density determination, there also is some evidence that the concentration of SOC may be higher in soils with high coarse fragment contents (Schaetzl 1991).

A factor that confounds our understanding of harvest impacts on SOC is that most published studies of soil organic matter report results as concentrations instead of quantities. Brandt (1993) conducted an exhaustive review of literature and concluded that very little of the SOC literature was conducted with sufficient rigor or reported in usable units for estimating the magnitude of soil carbon pools. Understanding impacts of various sources of carbon upon global climate requires more precise knowledge of pool sizes or quantities. In this study, SOC quantities were determined for both surface and subsurface mineral soils. Additionally, the sampling scheme permitted extrapolation to broader landscapes on the basis of the percentages of areas represented by different geomorphic surfaces of similar soils and landscapes in the region.

The primary objective of this study was to quantify the change in SOC following selection timber harvests. Based on the characterization of soil samples from two upland Missouri Ozark sites, changes in the amount of SOC was determined, and the data are now part of a larger database for modeling changes in the global climate. Specific objectives were: 1) to determine if the quantity of SOC and the soil bulk density changed following selection harvesting, and 2) to determine if the use of alternative harvesting techniques reduced or eliminated changes in SOC and bulk density that would have negative effects on future forest productivity.

Study Areas

This study was installed on two upland oak/pine forest sites in Wayne and Reynolds counties of southern Missouri. Soil survey data are not available for these counties; however, the soils in both areas were formed in limestone residuum. Summit positions consisted of soils formed primarily in Roubidoux geologic strata minerals, and midslope and footslope position soils were formed in Gasconade deposit parent materials. A thin loessial veneer was noted on both areas, and footslope positions had a thicker silty textured surface horizon as a result of the movement of the original loess deposits from summit and shoulder positions to footslope positions. The soils on both sites were rocky, and chert (SiO₂) fragments were ubiquitous both on the soil surface and in the subsurface.

Both sites were under an uneven-aged management regime, which incorporated both group and individual tree selection harvests for natural stand regeneration. The group selection areas provided 70' openings in the canopy to encourage regeneration of shortleaf pine (*Pinus echinata* Mill.), and individual-tree selection was used between the group openings for stand improvement. The sampling areas within each of the timber sales were approximately 20 acres in size which was large enough to establish plots across multiple landscape positions. One of the study sites was logged using conventional rubber-tired cable skidders, and the other site was logged using mules to skid logs, since logging with draft animals is purported to be a low-impact alternative harvesting system. The logs skidded using mule power were ground skidded without a sully to lift the front end of the log. Conversely, most of the logs skidded by the cable skidder did have the lead end of the log elevated during skidding.

Methods

To address potential differences in site and stand conditions between the two study/harvest areas, pre- and post-harvest forest inventory measurements were taken using six 0.8-acre rectangular plots that encompassed all of the major landforms within the sites. The inventory data were used to determine the intensity of harvest and residual stocking on each study site. These measurements were essential for interpreting treatment responses on both the animal and rubber-tired skidded sites, since both types of operations cannot exist in the same stand. A Wilcoxon rank test was performed to compare the medians of plot measurements between the two study sites. These fixed area plots also were used to calculate the percent of soil area disturbed during harvesting.

Most soil properties are not normally distributed (Young and others 1993), so Hammer (1991) and Daniels and Hammer (1992) have recommended that soil sampling be stratified using geomorphic surfaces as sampling “treatments”, with intensive sampling within each area to better allow determination of site variability. The geomorphic features delineated in this study include summit, footslope, southwest midslope and southeast midslope landscape positions.

Another limitation to our understanding of SOC distribution and dynamics is the misconception that most SOC is in the surface soil horizons, which often leads to sampling by depth increment rather than by genetic horizons (Hammer and others 1995). By comparing soil nutrient values within horizons that formed similarly, sampling variability is reduced compared to depth sampling methods; therefore, this study compared changes in soil properties by horizon instead of by depth.

To provide the best comparison of SOC changes in skid trails due to harvesting activities, the data reported in this paper are from skid trails and control (undisturbed) areas adjacent to skid trails within the harvested areas. The post-harvest samples were taken before a growing season passed, so changes in control versus skid trail SOC due to changes in stand density, new herbaceous growth, and altered hydrology did not have time to occur. Pre-harvest soil samples were taken initially as a baseline against which post-harvest samples were to be compared. However, delays in harvesting on both study sites made pre- and post-harvest comparisons of SOC problematic. The pre-harvest samples were processed, but these data are not presented in this manuscript due to the potential effect of delayed harvesting on SOC comparisons.

Following the completion of harvest activities in 1996, soil cores were taken from both sites using a two-person power auger similar to the device used by the USDA Forest Service for initial measurements in the Long-Term Soil Productivity (LTSP) study (Ponder and Mikkelsen 1995, Ponder and Alley 1997). Eight 10-cm diameter cores were taken to a depth of 30 cm within two skid trail types (primary and secondary) within the four aforementioned landscape/slope positions. Four cores were extracted from undisturbed areas adjacent to the skid trails within each of the positions. This study design provided 16 control cores and 64 treatment cores from both primary and secondary skid trails per study site, so a total of 160 cores were taken from the two study sites. The GLM procedure of SAS was used to accommodate the unequal sample size between control and skid trail (treatment) samples. The resulting linear statistical model contained the effects of site, landscape/slope position (plot) within site, skid trail type, soil horizon, and all possible interactions of these factors.

Since determination of bulk density was a prerequisite for the quantification of SOC, validation of the core sampling technique was deemed prudent. Unfortunately, a paired t-test comparison of soil core volumes and core hole volumes performed on a subsample of the pre-harvest cores revealed that the coring technique was compacting the soil samples ($p=0.0001$); therefore, adjustments to horizon thicknesses for both sites were made based on the ratio of the depth of the hole to the length of the core. All post-harvest core-hole volumes were adjusted in this manner. While this hole:core ratio adjustment improved estimates of bulk density, the underlying assumption that each horizon was compacted proportionally to its thickness is suspect. Subsequent research on the power-auger sampling technique has shown that subsurface horizons are more susceptible to compaction from power auger sampling than are surface horizons (Ficklin 2002).

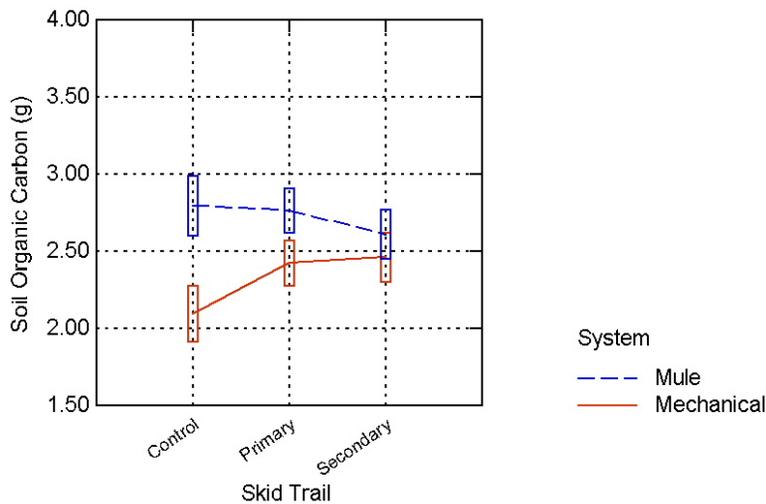


Figure 1.—Mean SOC quantities with standard error boxes for cores extracted from skid trails and control areas by system.

Each core was described in the laboratory by physical and color characteristics, and horizons were delineated for further analysis of the parameters of interest: bulk density, coarse fragment content and carbon concentration. Adjustments to bulk density were made for coarse fragment content in addition to the adjustments made for the compaction from sampling. SOC concentration was determined by combustion in a Leco C-144 carbon analyzer, and two replications were made for each horizon to assure the repeatability of carbon measured. In the final analyses, the average of the two concentrations was used for the comparisons of carbon concentrations, and the quantity of carbon in each horizon was determined based on the mean SOC concentration and the soil mass without rock.

Results

Test results indicated that residual trees per acre ($p=0.630$), residual sawtimber volumes ($p=0.749$) and residual basal areas ($p=0.420$) were not significantly different between sites. Based on these observations, we concluded that stand conditions would not influence the extent of soil disturbance. However, more trees were cut on the animal-skidded tract than on the rubber-tired skidder tract ($p=0.005$), and a greater amount of basal area was cut on the animal-skidded site ($p=0.037$). Since harvesting more trees requires more turns in the stand, it was assumed that the animal-skidded site would be predisposed to greater soil disturbance than the rubber-tired skidder site. Interpretation of results must be made within this context.

The area of skid trails within the 4.8 acres of plots on the animal-skidded site was 4,400 ft² for primary trails and 2,000 ft² for secondary trails- 2.1% and 1.0% of the total harvest area, respectively. The rubber-tired skidder site had 10,700 ft² of primary skid trails and 9,600 ft² of secondary skid trails- 5.1% and 4.6% of the total harvest area, respectively. Given the difference in the extent of skidding disturbance, inferences about the impact of harvesting on changes in soil density and SOC require consideration of both the magnitude of change and the proportion of the stand affected by the soil density and SOC changes.

When both surface and subsurface horizons were examined, the quantity of SOC was higher for skid trails and control areas on the animal-skidded tract than on the mechanically harvested tract ($p=0.035$) (Figure 1), but the concentration of SOC and the bulk density of soil did not differ ($p=0.323$ and $p=0.853$, respectively) (Figures 2 and 3). The trend in SOC concentration across skid trails and control areas was slightly lower on the mechanically skidded site. However, a statistically significant difference between systems was not observed for the horizon*skid trail interaction ($p=0.193$). No differences between systems were observed for any treatment combination within the surface A horizons, but the quantity of SOC in Bt subsurface horizons differed ($p=0.0001$) with the mule skidded site having 40% more SOC in primary and secondary skid trails. The concentration of SOC in the subsurface did not vary between systems ($p=0.212$). Bulk density did not differ between systems for both surface or subsurface horizons, and skid trail and system*skid trail factors also did not differ. When only control areas outside skid trails were examined, mean SOC quantities were similar for surface and subsurface horizons; however, antecedent

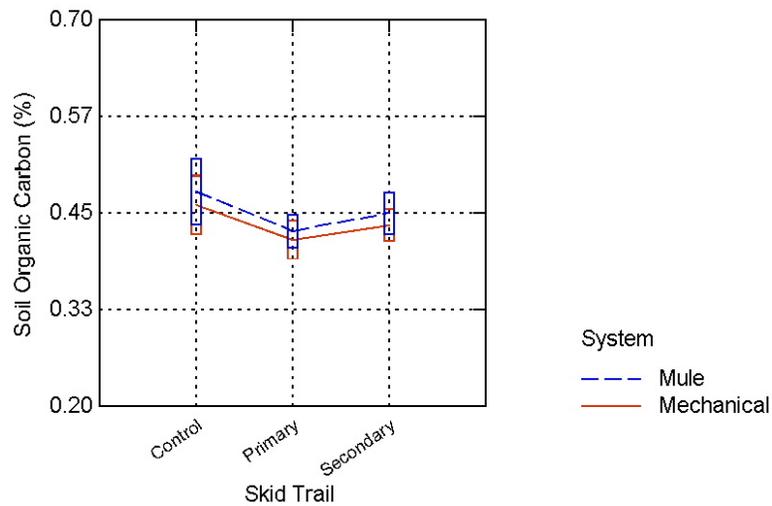


Figure 2.—Mean SOC concentrations with standard error boxes for cores extracted from skid trails and control areas by system.

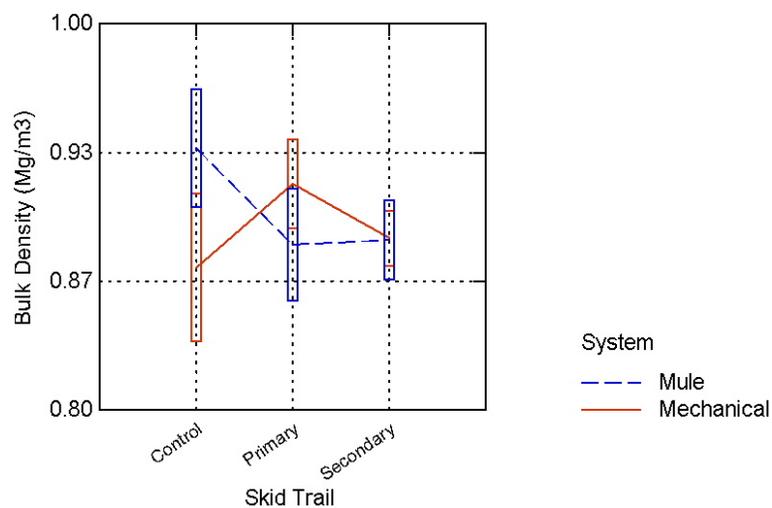


Figure 3.—Mean bulk densities with standard error boxes for cores extracted from skid trails and control areas by system.

subsurface SOC quantities and concentrations were higher on the mule skidding study site ($p=0.046$ and $p=0.022$, respectively).

Discussion

The patterns and processes of natural soil heterogeneity, particularly at the landscape or watershed scale, make treatment replication difficult or impossible. Therefore, cause-and-effect relationships often are inferred from rigorous, systematic treatment sampling (Milliken and Johnson 1989), with attention to the formative processes under which the soil was created (Daniels and Hammer 1992). Soil carbon has been shown to fluctuate significantly temporally and spatially by topographic position in response to changes in parent material, biotic respiration, temperature, moisture and pH (Hammer and others 1987, Hanson and others 1993). Therefore, it was essential to assess the significance of treatment responses by controlling as many of these potentially confounding factors as possible. Soil density and compressibility between the control areas of both study sites were not controllable, but patterns of SOC concentrations across trail types did facilitate the monitoring of SOC quantity changes.

The increase in the quantity of carbon in the skid trails on the mechanized skidder site was not a direct result of increased soil density from compaction. Rather, a combination of compaction, rock contents in control versus skid trail areas and a minimal decrease in SOC concentration best explain the measured trends. The absence of a measurable decrease in SOC is attributed to the short time lapse between harvesting and soil measurements. Debris from skidded logs still was present in both primary and

secondary skid trails, and the decomposition of this debris would be expected to mitigate SOC losses due to perturbation in the short term. The removal of rock fragments from the skid trails due to skidding also contributed to the observed increase in carbon content compared to the undisturbed control areas.

Bulk density determination in rocky soils has long been known to be problematic, so accurate estimation of soil compaction from skidding was difficult. With the adjustments to sample volume measurements, soil density responses to skidding disturbance were improved and trends across control and skid trail areas were discernable, but considerable “noise” still persisted. Contrary to expectations, carbon content did not decrease significantly in the skid trails of either site. This observation is expected to be a transient effect due to the decomposition of slash debris within the skid trails. Similarly, carbon concentrations did not change or decrease measurably in the skid trails of either site, so the observed increases in carbon content were likely due to both increased soil density and decreased near-surface rock content in the skid trails. As a result of displacing rocks from the skid trails during skidding, the mass of soil per unit of sample volume increased. Higher carbon contents in the subsurface horizons of control samples within the animal-skidded site make comparisons of carbon change to a 30 cm depth difficult. Additional measurements to quantify soil carbon changes over a period of years instead of months are needed to determine true trends in SOC quantities following harvesting operations with either mechanized or animal-skidding systems. The plots used in this study still are marked, so follow-up measurements would be possible.

The comparison of animal skidding with rubber-tired skidding revealed some advantages to using mules to minimize the impact of logging operations on Ozark forest sites. The advantages observed were not related to the magnitude of disturbance. Rather, the extent of disturbance was reduced by nearly one-third with the use of animal skidding-- even when a greater number of stems, a greater basal area, and a higher volume were removed on the animal-skidded site which resulted in more log-turns. An additional benefit observed on the animal-skidded site was a reduction in the extent and severity of damage to residual trees (Ficklin and others 1997). This study does not provide evidence that animal skidding improves the retention of SOC in forest soils following harvesting, but the data on short term changes in SOC dynamics following soil compaction and scarification do provide evidence of benefits from mule skidding. Given the reduced area of soil scarification, it is reasonable to hypothesize that rapid fluxes of CO₂ from forest soils following selection harvests would be reduced when animal skidding techniques are employed. As long as a niche market exists for low productivity harvest systems employing animal skidding, forest landowners have an option for timber management even when fiber production is not a primary management objective. If future studies support the hypothesis that reduced site scarification also reduces CO₂ efflux, then the prospects for forest landowners to earn more carbon credits with the use of animal skidding would be an incentive to adopt such low intensity harvesting operations.

Acknowledgments

The authors thank the Missouri Department of Conservation for both financial and in-kind support for this research. We also would like to thank the University of Missouri Research Board for providing supporting funds for this project. The authors thank Dr. Michael G. Shelton and other reviewers for their constructive and insightful comments.

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IMPACTS OF HISTORIC FIRE, STAND AGE, AND NITROGEN SATURATION STATUS ON SOIL NITROGEN CYCLING IN MID-APPALACHIAN FORESTED WATERSHEDS

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ABSTRACT.—Our research objective was to determine if past land disturbance due to fire, stand age, or reported nitrogen saturation status have any influence on soil nitrogen cycling in forested watersheds. Seven forested watersheds in northeastern West Virginia were chosen for the study: a burned watershed, and logged watershed (past land disturbance); an old-growth watershed, and two mature 60-90 year old watersheds (stand age); and a nitrogen saturated watershed, and nitrogen limited watershed (N saturation status). Monthly net N mineralization and nitrification rates were measured along 3 transects on each watershed in June, July, and August 1998 by employing buried bags in 0-10 cm of mineral soil. The old-growth watershed and two nearby mature watersheds had similar net N mineralization and nitrification rates. The burned watershed had lower net N mineralization rates than a nearby logged watershed, indicating the importance of past land disturbance. Watershed 4 on the Fernow Experimental Forest, WV, which has been cited as the best example of a nitrogen saturated watershed in the northeastern United States, had significantly higher net N mineralization and nitrification rates than nearby adjacent Fernow watershed 10. The observed difference in nitrate-N export in streamflow between Fernow 4 (5.12 kg ha⁻¹ yr⁻¹) and Fernow 10 (1.22 kg ha⁻¹ yr⁻¹) could be due to significantly higher soil nitrate production rates.

Northern temperate forests long have been considered nitrogen limited systems based on the relative absence of nitrogen leaching from forested watersheds. However, long-term monitoring of experimental forested watersheds throughout the eastern United States has shown significant increases in nitrate export over the past several decades (Johnson and Lindberg 1992, Stoddard 1994, Peterjohn et al. 1996). The theory of nitrogen saturation has been formulated to explain increasing nitrate leaching. A nitrogen saturated ecosystem is defined as one in which chronic atmospheric nitrogen deposition results in nitrogen supplies that exceed plant and microbe demands, and thus, nitrogen is available for export in streamflow (Agren and Bosatta 1988, Aber et al. 1989, Aber et al. 1998, Aber et al. 2003). However, not all forested watersheds in the eastern United States show nitrogen saturation symptoms. In fact, substantial variation (500%) in nitrate exports has been reported for forested watersheds within the mid Appalachian region (DeWalle and Pionke 1996).

Variability in soil nitrogen cycling has been identified as an important factor in explaining these significant nitrate export differences (Williard et al. 1996). Many factors can cause differences in soil nitrogen cycling and subsequent nitrate exports, including differences in wet and dry atmospheric nitrogen deposition (Dise and Wright 1995, Aber et al. 2003), soil carbon:nitrogen ratios (Johnson and Lindberg 1992, Lovett et al. 2002), nitrogen saturation status (Peterjohn et al. 1996, Gilliam et al. 2001), forest species composition (Fenn et al. 1998, Lovett et al. 2002), stand age (Vitousek and Reiners 1975, Goodale and Aber 2001), and past land disturbance history including agriculture and severe fire (Raison 1979, Compton and Boone 2000, Goodale and Aber 2001).

Our research objective was to examine if past land use disturbance (severe fire), stand age (old-growth), and nitrogen saturation status influenced soil N mineralization and nitrification rates in 7 forested watersheds in northeastern West Virginia. We assessed each factor by comparing “treated” watersheds (burned, old-growth, and nitrogen saturated) to nearby “control” (logged, mature, and nitrogen limited) watersheds.

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Table 1.—Physical and vegetative characteristics of the seven study watersheds in West Virginia.

Watershed	Area (km ²)	Aspect	Geology ²	Dominant Overstory Vegetation
Fernow 4	0.39	NNW	CCP	<i>Quercus rubra</i> L., <i>Liriodendron tulipifera</i> L.
Fernow 10	0.15	N	CCP	<i>Q. rubra</i> L., <i>L. tulipifera</i> L., <i>Acer rubrum</i> L.
Shays Run	6.03	S	PVA	<i>Tsuga canadensis</i> L.
Otter Run	0.48	E	PVA	<i>Prunus serotina</i> L., <i>A. rubrum</i> L.
W. Three Spring	0.12	E	MCG	<i>A. saccharum</i> Marsh., <i>Tilia americana</i> L.
Karly Run ¹	0.28	SE	MCG	<i>A. saccharum</i> Marsh., <i>T. americana</i> L.
Freeland Run	1.28	NNW	MCG	<i>A. saccharum</i> Marsh., <i>Betula lenta</i> L.

¹A previously unnamed stream.

²CCP represents Catskill/Chemung/Pocono shale and sandstone; PVA represents Pottsville/Allegheny sandstone; MCG represents Mauch Chunk/Greenbrier shale/limestone (Reger 1923, Reger 1931).

Table 2.—Four-week mean (June through August 1998) net N mineralization and net nitrification rates in the upper 10-cm of mineral soil for 7 study watersheds in West Virginia.

Watershed	Watershed Type	Mean Stream Nitrate-N (mg L ⁻¹)	Net N Mineralization (mg kg ⁻¹ 4wk ⁻¹) ²	Net Nitrification (mg kg ⁻¹ 4wk ⁻¹)
Fernow 4	N-saturated	0.751	17.20 a	13.02 a
Fernow 10	N-limited	0.216	5.26 b	-0.17 b
Shays Run	Burned	0.043	4.00 a	5.77 a
Otter Run	Logged	0.111	11.76 a	8.00 a
W. Three Spring	Old-growth	0.753	14.05 a	11.11 a
Karly Run ¹	Mature	0.547	19.82 a	8.02 a
Freeland Run	Mature	0.500	14.62 a	8.53 a

¹A previously unnamed stream.

²Net N mineralization and net nitrification means with different letters are significantly different among basins at $\alpha=0.05$ using Tukey's HSD mean separation procedure.

There has been little research conducted on the long-term impacts of severe fire on forest soil nitrogen cycling (Gagnon 1965, Hornbeck and Lawrence 1996). Severe fires may potentially affect long-term forest nitrogen cycling by volatilizing significant amounts of soil nitrogen (Raison 1979). Thus, we hypothesized that a historically burned watershed would have significantly lower N mineralization and nitrification rates than a nearby historically logged watershed. Stand age can affect nitrogen leaching, because young aggrading forests normally have greater nutrient demands than mature or old-growth forests, and old growth forests have relatively low productivity resulting in greater soil N accumulation compared to C accumulation (Vitousek and Reiners 1975, Goodale and Aber 2001). We predicted that an old-growth (> 150 years old) watershed would have significantly greater N mineralization and nitrification rates than nearby mature (60 – 90 years old) watersheds. We further hypothesized that a nitrogen saturated watershed, characterized by increasing nitrogen export over an extended period of time, will have greater N mineralization and nitrification greater rates than a nitrogen limited watershed.

Study Areas

The study watersheds were located in Tucker and Randolph counties within the Monongahela National Forest and each was 100 percent forested with no major disturbances in the past 60 years. The watersheds are relatively small and contain first or second order streams (table 1). Predominant overstory species across the seven watersheds were sugar maple (*Acer saccharum* Marsh.) and northern red oak (*Quercus rubra* L.) (table 1).

The watersheds were classified according to their disturbance history (burned, logged, and old growth) and nitrogen saturation status (table 2). Burned watersheds experienced severe fires 60 to 90 years ago caused by the abundant slash left behind after the clearcutting of mid-Appalachian forests in the early 1900's. Current vegetation resulted from natural recovery. Historical fire records and maps from the Monongahela National Forest, WV were examined to locate these severely burned areas. The logged

watersheds were cut 60 to 90 years ago, and did not experience any severe fires, based on fire records. Stand age inventories for the Monongahela National Forest, WV were used to identify old-growth watersheds. Old-growth stands were defined as any stand not harvested within the past 150 years. The old-growth watershed was located adjacent to the Otter Creek Wilderness. The watersheds within each watershed type comparison (burned vs. logged, old-growth vs. mature, and nitrogen saturated vs. nitrogen limited) contained the same underlying bedrock geology (table 1). All of the study watersheds were located on previously unglaciated terrain.

Methods

Field Methods

Monthly net N mineralization and nitrification rates were measured in June, July, and August 1998 along three 16-m transects on each watershed using buried bags in 0-10 cm of mineral soil (Eno 1960). Protocols for the buried bag method were adopted from Hart et al. (1994). The three transects were located at the watershed mouth (near the stream sampling point) at a low elevation, near the middle of the watershed at a mid-slope elevation, and in the headwaters at a ridge top elevation. Nine sampling points were spaced at 2-m intervals along each 16-m sampling transect. The organic horizon was removed at each point and set aside. Then a 10-cm diameter by 10-cm deep mineral soil core was taken and gently extruded into a quart-sized zipper-lock plastic bag. Bags were closed, placed back in the resulting hole, and covered with the organic horizon material. After a four-week incubation period, the buried bags were collected and composited by threes to yield nine composite samples per watershed for post-incubation soil ammonium and nitrate measurement.

Another soil sampling transect was located adjacent to the buried bags to measure pre-incubation levels of soil ammonium and nitrate. At each transect, nine mineral soil cores were collected and composited in the field by threes and put in coolers on ice for transport to The Pennsylvania State University.

Monthly net N mineralization and net nitrification rates were computed according to the following equations:

$$\begin{aligned}\text{Net N mineralization} &= [(\text{NH}_4\text{-N} + \text{NO}_3\text{-N})_{t+1} - (\text{NH}_4\text{-N} + \text{NO}_3\text{-N})_t] \\ \text{Net nitrification} &= [(\text{NO}_3\text{-N})_{t+1} - (\text{NO}_3\text{-N})_t]\end{aligned}$$

where t represents the pre-incubation levels and $t+1$ represents the post-incubation levels. $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ were measured in mg per kg of soil. Net nitrification and N mineralization rates were expressed as mg of N per kg of dry soil per 4-week period ($\text{mg kg}^{-1} 4\text{wk}^{-1}$). Negative net mineralization values were possible and represented net immobilization of inorganic nitrogen by the soil microbes during the incubation period.

At each sampling transect, three additional soil cores were taken for gravimetric determination of soil moisture. One Taylor maximum-minimum thermometer was buried 5-cm in mineral soil to measure maximum and minimum soil temperatures during the incubation period. Soil temperature at the time of the site visits also was noted at the start and end of each incubation period.

In August 1998, sampling transects for standard soil chemistry were established adjacent to (within 3 meters) the 3 buried bag transects on each watershed. Nine sampling points were spaced at 2-m intervals along each 16-m sampling transect. The entire organic horizon and the top 10 cm of mineral soil were collected at each point. Organic and mineral soil samples were transported to The Pennsylvania State University, Agricultural Analytical Laboratory for standard soil tests [exchangeable Ca, Mg, K, and P; effective cation exchange capacity (CEC), acidity, and pH]. Subsamples of organic and mineral soil were analyzed for total C and N at the Northeastern Research Station's Timber and Watershed Laboratory in Parsons, WV.

Stream water grab samples were collected during baseflow from the 7 watersheds during August 1997 and March 1998. They were transported on ice to the Water Analysis Laboratory at The Pennsylvania State University for dissolved nitrate, ammonium, and organic N analysis.

Laboratory Measurements

Prior to analysis, organic and mineral soil samples were dried at 70 °C for up to 12 hours and then passed through a 2-mm sieve. Soil nitrate was extracted with an ammonium sulfate and boric acid solution and analyzed with an ion-specific electrode (Griffin 1995). Soil ammonium was extracted with a 0.5 N potassium chloride solution and determined with a Technicon Autoanalyzer (Keeny and Nelson 1982). Results were expressed as mg of NO₃-N and NH₄-N per kg of dry soil. Exchangeable Ca, Mg, and K and labile, mineralized P were determined with the Mehlich 3 method (Wolfe and Beegle 1995). Exchangeable acidity was extracted and measured by the SMP buffer method (Sims and Eckert 1995). Soil pH was measured in a soil water slurry (5 g : 5 ml) using an electronic pH meter (Sims and Eckert 1995). Effective CEC was calculated by summing the exchangeable Ca, Mg, K, and acidity (Ross 1995).

Dried sieved organic and mineral soil was prepared for C:N determination by placing approximately 6 mg of soil in a 5 mm x 9 mm tin capsule along with a small amount of vanadium pentoxide. Total C and N percentages in the soil were determined by combustion using an organic elemental analyzer (Carlo Erba NA 1500 CNS Analyzer, Valencia, CA) (Baccanti et al. 1993).

Moisture content of the soil for the net N mineralization and nitrification study was determined on a dry weight basis (Gardner 1965). Wet soil was weighed to the nearest 0.01 mg, dried in an oven at 105 °C for 24 hours, and re-weighed to determine the percent soil moisture.

Dissolved (<0.45 μm filtered) stream nitrate, ammonium, and organic N (difference between total dissolved and nitrate plus ammonium) were analyzed at the Water Analysis Laboratory, The Pennsylvania State University via cadmium reduction, automated phenate, and persulfate oxidation methods, respectively (American Public Health Association 1995).

Statistical Methods

Prior to performing analysis of variance procedures (ANOVA), data sets were analyzed to determine if error terms were normally distributed and homogeneous. Normal probability plots and Shapiro-Wilk test statistics (SAS Institute Inc. 1985) confirmed that all the data sets were distributed normally, and Bartlett's tests (SAS Institute Inc. 1985) showed error variances for the various ANOVA models to be relatively homogeneous; consequently, no data transformations were required.

An ANOVA model for the net N mineralization and nitrification data set was performed with watershed type, month, and elevation effects. Another ANOVA model was developed with soil chemistry data to determine if soil chemistry parameters were significantly different among watershed types. Tukey's HSD mean separation procedure was added to each of the ANOVA models (SAS Institute, Inc. 1985). Maximum R² improvement stepwise regression models were developed to predict net N mineralization and net nitrification rates from soil chemistry parameters, soil moisture, and soil temperature (SAS Institute, Inc. 1985).

Results and Discussion

N Saturation Effects

Comparisons of net N mineralization and net nitrification among different watershed types yielded only one statistically significant result (table 2). Fernow 4, which has been cited as the best example of a nitrogen saturated watershed in the northeastern United States (Peterjohn et al. 1996, Fenn et al. 1998), had significantly higher net mineralization and nitrification rates than Fernow 10. Christ et al. (2002) also found significantly greater soil net nitrification potentials on Fernow 4 than Fernow 10. The two watersheds are nearly adjacent, have similar management histories, and both serve as experimental controls in the Fernow Experimental Forest, WV. The observed difference in nitrate-N export in streamflow between Fernow 4 (5.12 kg ha⁻¹ yr⁻¹) and Fernow 10 (1.22 kg ha⁻¹ yr⁻¹) could be due to the significantly greater soil nitrate production rates (table 2). Mean soil chemistry differences may be causing these differences in nitrate production. Mean mineralized phosphorus concentrations were over five times greater on Fernow 4 compared to Fernow 10, indicating microbial populations on Fernow 4 are actively decomposing organic matter and rapidly cycling nutrients. Phosphorus has been cited as a limiting nutrient of N-fixing bacteria

(Vitousek and Howarth 1991). N-fixation on Fernow 10 may be limited by low phosphorus levels, resulting in smaller ammonium pools for nitrifying bacteria. Measured ammonium pools were significantly lower on Fernow 10 (13.62 mg kg⁻¹) compared to Fernow 4 (16.47 mg kg⁻¹), providing support to this hypothesis. The significantly lower calcium concentrations on Fernow 10 (0.344 meq 100g⁻¹) compared to Fernow 4 (1.267 meq 100g⁻¹) also could be limiting the activity of N-fixing bacteria and ultimately reducing soil nitrification rates (Alexander 1977).

Severe Fire Effects

Within the PVA geology category, Shays Run, a historically burned watershed, had lower net mineralization rates than logged watershed Otter Run (table 2). Even though the difference was not statistically significant, it was large enough to warrant discussion. Fire can cause significant N losses from forest ecosystems via volatilization of N in the litter layer and upper mineral horizons (Grier 1975, Raison 1979). Repeated small N losses due to low intensity fires or substantial one-time N losses due to an intense burn can deplete long-term soil N pools (Gagnon 1965, Carreira et al. 1994, Hornbeck and Lawrence 1996). Soil N pools on Shays Run apparently have been restored since the devastating fire on Canaan Mountain in the early 1920s, which completely consumed a thick organic horizon and consumed the top portion of the mineral soil (L. White, Monongahela National Forest - personal comm.); soil N pools (0.266%) on Shays Run were similar to Otter Run (0.301%). Given that all soil chemistry parameters were relatively similar for Shays Run and Otter Run, it is difficult to determine why the burned watershed had lower net mineralization rates. Perhaps the severe fire has had a long-term effect on the ammonifying bacteria populations.

Stand Age Effects

Net N mineralization and nitrification rates in West Three Spring, an old-growth watershed, were on the same order of magnitude as the other two logged watersheds with the same underlying geology (table 2). The old-growth watershed had the second-highest net nitrification rates among the seven watersheds sampled. This was a surprising result considering classical succession theory that predicts nitrification will be progressively inhibited through succession (Robertson and Vitousek 1981, Robertson 1982). However, recent research has shown that significant gross soil nitrification can occur in late successional forest ecosystems, and significant nitrate may become available if microbial immobilization is decoupled through a disturbance such as the buried bag technique (Stark and Hart 1999).

Predicting Net N Mineralization and Nitrification

Soil N, mineral P, moisture, and C:N ratios together explained 46 percent of the variation in net nitrification across all watersheds. Soil moisture helps determine nitrification rates by controlling rates of microbial activity (Stanford and Epstein 1974). Mineralized phosphorus levels are positively correlated with net nitrification rates, since they are both the result of an active microbial population. Competition for nitrogen between heterotrophic bacteria and nitrifying bacteria plays a major role in determining soil nitrate production, and the amount of labile C directly affects this competition (Riha et al. 1986). Heterotrophic bacteria are thought to be the most successful short-term competitors for N (Schimel and Firestone 1989), and C serves as the energy source for bacteria. Consequently, higher carbon levels result in greater N demand. Thus, soils with more available C have more heterotrophic demand for ammonium and less ammonium is available to be nitrified (Riha et al. 1986).

The prediction of net N mineralization was much weaker than net nitrification. Soil N and mineral P levels together explained only 19 percent of the variation in net N mineralization rates. C:N ratios were not significant in explaining net N mineralization rates.

Conclusions

The best example of a nitrogen saturated watershed in the northeastern United States, Fernow 4, exhibited significantly greater net N mineralization and nitrification rates than a nearby watershed with similar management history, Fernow 10. These results demonstrate that soil nitrogen dynamics are important in controlling nitrogen exports from forested watersheds. A historically burned watershed had lower net N mineralization rates than a nearby logged watershed that did not experience severe fire, indicating that past land disturbance can impact forest soil nitrogen cycling. An old-growth watershed and two mature (60-90

year old) watersheds had similar net N mineralization and nitrification rates, supporting the theory that soil nitrate production can be significant in late successional forested watersheds. Changes in factors that affect soil nitrogen cycling, such as soil C:N ratios, soil fertility, and soil moisture can significantly affect nitrate leaching from forested watersheds.

Acknowledgments

Funding for this project was provided by the USDA Forest Service, Northeastern Forest Experiment Station, Parsons, WV and the USDA Water Science Fellowship program. William Sharpe, Jon Chorover, Peter Deines, and two anonymous reviewers are thanked for their valuable comments on earlier versions of the manuscript.

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QUANTIFYING ABOVEGROUND CARBON STORAGE IN MANAGED FOREST ECOSYSTEMS IN OHIO

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ABSTRACT.—The amount of carbon sequestered was determined on managed even aged stands on sites in southeastern Ohio. Bottomland hardwood sites that consisted of sycamore (*Plantanus occidentalis*) and box elder (*Acer negundo*) were examined. The other forest types studied were monocultures of green ash (*Fraxinus pennsylvanica*), Austrian pine (*Pinus nigra*), and pitlolly pine (*Pinus rigida* × *taeda*), and five sites of white pine (*Pinus strobus*). These represented a chronosequence of ages for each forest type. There were nine .008-hectare plots located on each white pine site and three plots on each of the other sites. DBH and basal diameter were measured for all of the trees on each measurement plot. Trees were destructively sampled to determine the DBH, basal diameter, and dry weight of aboveground components (bole, crown stem, and branches and foliage) and samples were taken to determine the amount of carbon in tree components based on percent of weight. Regression analysis was used to determine the relationship between DBH or basal diameter and total carbon sequestered and subsequently applied to trees at the plot level to estimate total tree carbon per hectare. ANOVA was used to test for differences in carbon content between components and species.

Carbon is emitted into the atmosphere from both natural and anthropogenic sources. It is the emissions from anthropogenic sources that contribute to the accelerated greenhouse effect popularly called “global warming”. Forestlands are capable of mitigating this effect by absorbing and storing carbon. Through forest management practices greenhouse gas emissions can be offset. Ohio forest products and energy industries (and others) are interested in the potential use of their lands for carbon sequestration. The term carbon sequestration implies the absorption and storage of carbon from the atmosphere into a long-lived terrestrial pool a medium such as a forest ecosystem.

Forests store two-thirds of terrestrial carbon in soil and biomass by absorbing carbon dioxide from the atmosphere and converting it into organic material, an equivalent of nearly 1 trillion tons (Brown et al. 1993). Of all the plant kingdom, forests provide the most long-lived storage sink in the carbon cycle tying carbon up in wood and soil accumulation for years before returning it to the atmosphere by respiration, decomposition, erosion, or burning. Native forests cover some 3,400 million hectares worldwide, with an additional 1,700 million hectares of other wooded lands. Forest plantations comprise of roughly 100 million hectares (Austin et al. 1998). Historically, forests have been a net source of atmospheric CO₂, as 80 percent of the world’s original forest cover has been lost (Austin et al. 1998). Moreover, under current worldwide practices the world could lose 650 million hectares over the next 60 years, releasing up to 77 billion tons of carbon emissions in the process (Trexler and Haugen 1995). Because of these trends, afforestation, reforestation, and sustainable forest management practices become critical in creating and maintaining worldwide carbon stores. At the present time, six hectares are deforested globally for every hectare planted (Totten 1999).

Plantation forestry and sustainable management practices of natural forests will need to be a focus to offset these losses. Increased timber growth and the resulting carbon sequestration in forests can be accomplished through improved forest management practices. This includes regeneration, which involves the replanting of under-stocked forest stands (e.g., those damaged by disease, insects, or fire) and the harvest and regeneration of mature stands. Stocking control is another approach to increase timber growth by thinning stands where competition is retarding timber growth. Increasing the

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Table 1.—Descriptions of selected sites for forestry component of this study.

Landowner	Classification	Forest Type	Planted	Age (as of 2002)
MeadWestvaco	Abandoned agric. land	White pine	1995	7
			1993	9
			1991	11
			1988	14
			1985	17
AEP ¹	Marginal agric. land	Bottomland	1997	5
			1993	9
		hardwoods	2000	2
			1998	4
			1995	8
	Reclaimed grassland	Mixed	2000	2
			1997	5
		hardwoods	1994	8
			Austrian Pine	2001
		1997	5	
		1993	9	

¹American Electric Power Company.

production and harvesting of wood, or biofuels, for use as a fossil fuel substitute will also augment sequestration as will afforestation or the planting of trees on previously unforested land.

Accurate data for sites over time for various species do not exist or are limited for Ohio. Sites in this study were selected that represent a chronosequence of monotypic and mixed species forests that were planted for different management objectives. Trees were destructively sampled on each site and understory was sampled to determine the total aboveground carbon sequestered. These data and the subsequent analysis were used to model the sequestration potential over time. The projections will be substantiated by future measurements from these sites. The results may be used to provide a basis by which industries can measure and manage for carbon credits.

The objective of this study was to determine the amount of carbon sequestered in selected managed forests in Ohio by determining the rate of sequestration by measuring a chronosequence of forests, validating these rates by installing permanent plots to be monitored over time, and developing species specific relationship between biomass amount and carbon.

Biometric Model

Many of the experiments in this area in the past have used an allometric model (Kimble et al. 2003, Canary et al. 2000, Patenaude et al. 2003). This type of model estimates the biomass of the whole tree based on the measurement of part of the tree. The formulae that are used to make the biomass estimates are taken from other sources on other sites. The disadvantage of this model is that the results are based on data from other sources that may or not characterize the site under examination.

This project uses a biometric model. The biometric model uses data from the sites being examined to develop formulae that characterize the species on the site. The advantage of using the data from the sites under study to make estimates is that the allometric formulae designed and used in this study are definitely characteristic of the trees on the plot. Conspicuously missing from the literature are models that create their own allometric equations from biometric data for all parts of the study.

Site Selection and Description

A total of 16 study sites owned by MeadWestvaco, Corp. and American Electric Power (AEP) were selected that represent five different forest conditions in a chronosequence in southeastern Ohio. These sites were visited and selected with assistance from foresters from both of these corporate organizations. Descriptions of the selected sites for this study are listed in Table 1.

The sites selected had similar forest and soil characteristics, such as drainage, slope, etc. within the five different forest conditions selected. A chronosequence within each forest type was selected to provide a rate of carbon accumulation over time so that cumulative curve can be modeled. These sites will be monitored (measured) periodically to refine the model carbon accumulation curve for better future predictions.

Sites were selected that represent planted forest types at different stages in their development as well as historical use. The species represented are white pine (*Pinus strobus*), Austrian pine (*Pinus nigra*), green ash (*Fraxinus pennsylvanica*), pitch pine (*Pinus rigida* × *taeda*), and bottomland hardwoods (*Plantanus occidentalis* and *Acer negundo*). There were five *Pinus strobus* sites represented, two bottomland hardwood sites and three each of the remaining species. The *Pinus strobus* and *Pinus rigida* × *taeda* sites were located on abandoned agricultural land. The *Fraxinus pennsylvanica* and *Pinus nigra* sites were located on reclaimed surface-mined land, and the bottomland sites were marginal agricultural land.

Permanent square .08-hectare plots were placed on each site with at least a one-chain buffer separating plots. Within each plot a circular .008-hectare measurement plot was placed. Each tree was permanently labeled with an aluminum tag. An aluminum nail was placed at 1.4 meters above ground level on trees large enough to hold a nail to facilitate future measurement of dbh. Trees greater than or equal to 1.3 centimeters DBH within the measurement plot were measured for dbh and basal diameter. Trees less than 1.3 cm dbh were measured for height and basal diameter.

Laboratory Methods

Destructive Tree Measurements

Trees were cut at the ground line, and separated into components of bole, branches, and foliage. Large trees were cut into manageable pieces and weighed in the field to the nearest 0.05 kg. Small trees were brought back to the lab and weighed to the nearest 0.01 g.

Trees destructively sampled were taken in equal numbers from each site and in sizes to represent the range of sizes of trees in the study. The number of samples from each site was greater for the young trees because of the difficulty in destructively sampling very large trees. The samples were taken from near the measurement plots but not on the plots so as to not disturb possible future study.

Samples were taken from all trees that represented the different tree components and weighed. In order to ensure that the samples represented the whole component of each tree, samples were taken from different parts of each. In stems, the sample consisted of pieces from the base, tip, middle, $\frac{1}{4}$ and $\frac{3}{4}$. Branches were sampled that represented a range of sizes and leaves were taken from different parts of the plant. Samples were dried in ovens for three days at a constant temperature of 60°C. Samples were then reweighed after drying to determine moisture content. The subsequent moisture content value was used to determine the total dry weight of the tree and its components.

Surface samples, which includes all surface litter above the mineral soil, herbaceous and woody plants, were collected on each measurement plot from a 1.0 m² plot centered on the measurement plot. One sample was taken from each plot for a total of 9 per site in the white pines and 3 per site in the other sites. The total surface sample was weighed to the nearest 0.05 kg, and a grab sample was taken. This grab sample was weighed and dried for three days at 60°C to determine moisture content. The percent moisture content was applied to the total surface sample to estimate the total dry weight.

Carbon Determination

At least 0.04g of sample and usually more was collected from all tree components and surface samples for the purpose of determining carbon content. This small carbon sample was randomly collected from each tree component and ground in a Wiley mill to pass a 2mm sieve. These samples were sent to the STAR laboratory at the OARDC in Wooster, Ohio.

Table 2.—Equations for predicting total aboveground tree carbon (t/ha) by age (yrs) for the five forest types examined in this study in southeastern Ohio.

Forest type	Equation ^a	R ² x 100
Austrian pine	$C = 0.0040e^{0.9471(A)}$	99.94
Pitlolly pine	$C = 0.0582(A)^{3.2503}$	99.98
Green ash	$C = 0.0084(A)^{1.7301}$	89.47
Bottomland hardwoods	$C = 0.0024(A)^{3.9891}$	100.0
White pine	$C = 0.4340(A)^{1.8618}$	78.02

^aC = the total carbon in metric tons/ha for only the aboveground portion of all trees, and does not include forest litter above the mineral soil, and any herbaceous and woody understory; A = the stand age in years.

Carbon analysis was run on an Elementar Americas, Inc., Vario Max Carbon Nitrogen Combustion Analyzer, with a procedure described in the international standard publication ISO10694:1995(E). The amount of carbon measured in the samples by this method was expressed in units of percent carbon by dry weight.

Analytical Methods

Analysis of variance (ANOVA) was performed on the carbon content values by tree component, using species as the source of the variation, to determine if difference existed in carbon content among species.

Tree carbon equations were derived using regression analysis relating carbon weight with BD or dbh (depending on which best fit the data). The total weight of carbon was determined by multiplying the dry weight by the percent carbon.

These equations were applied to the data from the measurement plots to estimate the total carbon on each plot. The independent variable (either basal diameter or dbh) was used to determine the amount of carbon per tree on each plot. The estimated carbon of each tree were summed for each plot, and divided by plot size to give the carbon stored in grams per hectare, and converted to metric tons.

Carbon content for the surface samples was determined using the same method as described for the destructive samples. The values for the plots on each site were averaged to determine the weight of carbon for the surface.

The total tree carbon per hectare was averaged for each site, and was added to the carbon content of surface samples to give a total value of carbon per hectare on the site. Regression analyses were performed on the data to show the relationship between total carbon per hectare and age.

All statistical analysis was performed with Minitab (Meyer and Krueger 2001) or SAS (SAS Institute 1990).

Results

Total Carbon (plot)

The plot data were first considered without the surface samples examining only the total aboveground tree carbon. The equations with R² and standard error data are located in Table 2. The model that was used was chosen since it showed the best fit to the data by R² and by the fit of the predicted values against the actual values. This model was chosen over cubic, quadratic, and linear models as well as some other non-linear models. Figure 1 shows the graph of the model for Austrian pine.

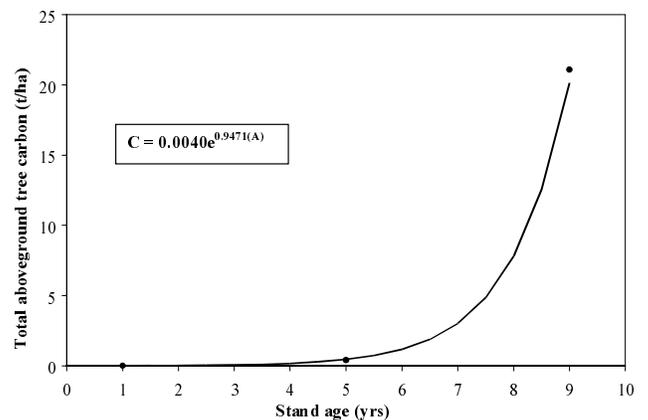


Figure 1.—Graph of total aboveground tree carbon (t/ha) versus stand age (y) for *Pinus nigra*.

Table 3.—Formulae of carbon stored by age per hectare with R² values with surface samples included.

Species	N	Formula	R ² x 100
<i>Pinus nigra</i>	3	C = 2.1975 A	72.87
<i>Pinus strobus</i>	5	C = 0.6990 A ^{1.7286}	80.77
<i>Fraxinus pennsylvanica</i>	3	C = 0.6018 A	24.22
<i>Pinus taeda x rigida</i>	3	C = 0.7908 A ^{1.9394}	99.71
Bottomland hardwoods	2	C = 0.0609 A ^{2.5497}	100.00

C = carbon (t); A = age (yrs)

Austrian pine, bottomland hardwood, and pitch pine sites showed a very strong reliability with R² values around 100%. The green ash sites showed a somewhat less but still fairly strong correlation (89.47%). The R² of 78.02% for white pine suggests a weaker reliability but still high enough to suggest a good fit.

Analyses were attempted using individual plot data as opposed to average plot data to determine if this would produce a better fit. The data were much less reliable for every species and is not reported here.

Table 3 shows the formulae for the models of carbon sequestered over time per hectare by site composition with surface samples considered. Bottomland hardwoods and *Pinus taeda x rigida* show a very strong reliability (R² around 100%). *Pinus strobus* has a reasonable level of reliability (R² = 72.87). *Pinus nigra* and *Fraxinus pennsylvanica* were better characterized by a linear regression. *Fraxinus* had a very unreliable fit (R² = 24.22). The *Pinus nigra* and more especially the *Fraxinus pennsylvanica* were largely influenced by surface samples in their total aboveground carbon. This is why the models without surface samples are more reliable for these forest types.

Nonlinear formulae of this form:

$$C = b_1 * \exp(-b_2 * \exp(b_3 * A))$$

were attempted to find if these would yield better results than those used in the study. The formula for *Pinus strobus* without surface samples was:

$$C = 84.2505 * \exp(-0.000016 * \exp(4.4878 * A))$$

and with surface samples:

$$C = 93.9602 * \exp(-0.000043 * \exp(4.0898 * A))$$

and for *Fraxinus pennsylvanica* without surface samples:

$$C = 1.7801 * \exp(-0.0173 * \exp(1.0209 * A))$$

with C representing carbon and A representing age. The resulting R² values were 84.08%, 84.20%, and 50.61%. These data also did not show as good a fit of the predicted values to the residuals. This is not an improvement on the models used in this section.

The graphs in this paper that were not linear showed a convex form. This is similar to the findings of Law et al. (2003), Pussinen et al. (2002), and Dieter and Elsasser (2002). The trees show a convex growth at the beginning of the curve followed by a concave shape toward the end. All of the ages in this study are in the range of the early part of these curves.

The total carbon per plot by species and land classification is shown in table 4. There are striking differences in C sequestered by species and land type. Austrian pine at age 9 stored much more C than green ash on the same type of sites at age 8 (30 and 4 t/ha respectively). Bottomland hardwoods on bottomland sites stored much more carbon at age 4 than pitch pine at age 5 (11 and 3.1 g/ha). White pine on abandoned agriculture sites changed dramatically in carbon storage between age 11 and 14 (24 and 89 t/ha).

Table 4.—Total carbon per plot (t/ha) by species and land classification.

Classification	Forest Type	Age	Total C plot (t/ha)	C w/o litter (t/ha)
Abandoned agric. land	White pine	7	26	22
		9	20	15
		11	24	19
		14	89	79
		17	89	81
Marginal agric. land	Bottomland	5	3.1	0.90
		9	19	18
	Pitlolly pine	2	3.3	0.75
		4	11	5.6
		8	28	24
Reclaimed grassland	Green Ash	2	3.9	0.011
		5	3.2	0.20
		8	4.0	0.22
	Austrian Pine	1	3.2	0.014
		5	3.3	0.39
		9	30	27

¹American Electric Power Company.

Table 5.—Analysis of variance statistics on the percent carbon based on dry weight by tree component. Species was the source of variation used in the analysis.

Tree component	N	Mean carbon (% of dry weight)	F-statistic	Prob>F
Bole	73	0.4950	99.65	0.0001
Branches	67	0.4994	88.32	0.0010
Foliage	73	0.5024	112.17	0.0010

Table 6.—Carbon content (percent of dry weight) by species for tree components.

Species	N	Carbon Content ¹ (fraction of dry weight)							
		Bole ²		Branch		Foliage		Branches and Foliage	
		N	Mean	N	Mean	N	Mean	N	Mean
<i>Pinus taeda x rigida</i>	14	14	0.5110 A	14	0.5121 A	14	0.5204 A	14	0.5158 A
<i>Pinus nigra</i>	20	20	0.5087 A	20	0.5161 A	20	0.5131 AB	20	0.5146 A
<i>Pinus strobus</i>	12	6	0.5072 A	6	0.5143 A	12	0.5192 B	12	0.5151 A
<i>Plantanus occidentalis</i>	8	8	0.4741 B	8	0.4797 B	8	0.4969 C	8	0.4883 B
<i>Acer negundo</i>	7	7	0.4736 B	7	0.4797 B	7	0.4613 D	7	0.4705 C
<i>Fraxinus pennsylvanica</i>	12	12	0.4679 B	12	0.4739 B	12	0.4746 E	12	0.4742 D

¹Values followed by the same letters down columns are not significantly different at the p = 0.05 level, Duncan's multiple range test.

²The bole is defined as the woody component from the ground to the tip, minus branches.

Carbon Content by Species

Table 5 displays the ANOVA statistics performed on the carbon percent by tree component. The model statistics indicated there were significant differences in carbon content among species. Duncan's new multiple range test was performed to determine differences in carbon content between species by tree component (Table 6).

As is shown in the Table 6, the carbon content for the bole is statistically similar among hardwoods and among pines but these values are statistically different between pine and hardwood species groups. The carbon percentages for hardwoods average 47% while the pines average 51%.

Greater variability was present in the foliage (Table 6). *Pinus taeda x rigida* is similar to *Pinus rigida* but different from *Pinus strobus*. Even though Duncan's test shows this difference, it may be an artifact of

Table 7.—Equations for predicting total aboveground tree carbon (C, g) for seedlings and saplings in southeastern Ohio using basal diameter (BD, cm) or diameter at breast height (DBH, cm).

Species	N	Prediction equation	R ² x 100
<i>Acer negundo</i>	7	C = 8.1484(BD ^{2.7219})	99.48
<i>Fraxinus pennsylvanica</i>	12	C = 10.9910(BD ^{2.6685})	96.62
<i>Platanus occidentalis</i>	8	C = 10.1620(BD ^{2.5998})	99.48
<i>Pinus nigra</i>	20	C = 9.5266(BD ^{2.8408})	99.02
<i>Pinus taeda x rigida</i>	14	C = 10.5800(BD ^{2.8604})	98.26
<i>Pinus strobus</i>	12	C = 167.4000(DBH ^{1.9167})	93.74

the data due to small sample sizes and greater variability measured in foliage. The hardwoods are significantly different from the pines and from each other. *Platanus occidentalis*, *Fraxinus pennsylvanica*, and *Acer negundo* have percent carbon values based on dry weight of 49.7%, 47.5%, and 46.1% respectively. These values for hardwoods are lower than those observed for pines.

When considering the branches and foliage together (Table 6), the pine group together with similar values (51.6%, 51.5%, and 51.5%), *Platanus occidentalis* stands alone (48.8%), and the other hardwoods group together (47.4% and 47.1%).

Many of the studies in the literature use one standard percentage of carbon for all trees. For many the value is 50% (Nowak and Crane 2002, Brack 2002, Kimble et al 2003, Makundi et al 1995, Dieter and Elsasser 2002), with one at 45% (Nowak 1993) and one at 46% (Patenaude et al 2003). Canary et al. (2000) calculated the values for fir and found a range of 48.1 % to 54.6% with an average of 51.2%. All of these values seem to be consistent with the values determined by this project. The values calculated for hardwoods around 47% and pines around 52% are very close to the values of Canary et al. (2000) and not far from standardized values of 45% and 50% for all species. The finding of an average carbon percent by Canary et al. (2000) of 51.2% for fir goes to further the idea that conifers have a similar carbon content as one another as that values is likely not significantly different than the values found in this study for conifers.

Carbon Equations (individual tree)

The data for the individual tree carbon models had very good R² values ranging from with 93.74% to 99.48% (Table 7). A comparison of the R² values and predicted versus residual plots, indicate individual tree carbon models shows a good fit of data per species. All of the models show a high reliability (R² > 93%).

Figure 2 shows the graph for the model of the carbon data for *Acer negundo*. Carbon storage begins to increase dramatically after seedlings become established and more rapid growth occurs. For example, the ratio of carbon (g) to tree size (BD) increases from about 30:1 to 180:1 for *Fraxinus pennsylvanica*.

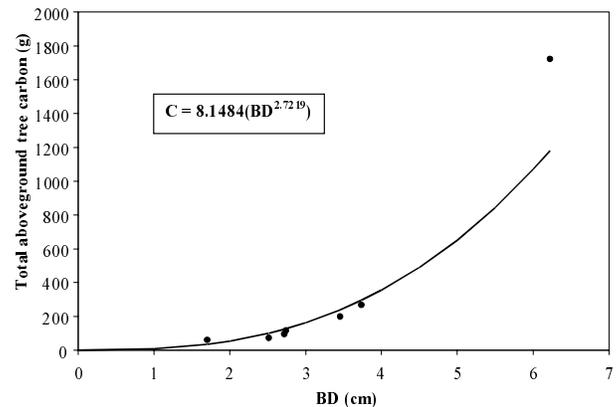


Figure 2.—Graph of total aboveground tree carbon (g) versus basal diameter (g) for *Acer negundo*.

Discussion

The greatest contribution of this paper is the use of equations of carbon storage by age. This model is only valid for plantations since it requires an even age. The high R^2 (78-100) of these formulae suggests that this is a valid option for plantations. This is significant since equations of stand C by age could present a potential shortcut in estimating plantation C storage. This would replace having to measure individual trees or compare inventories to carbon equations for individual trees.

The results from the comparison of land use and species to carbon storage are important (Table 4). The results suggest that species types should be selected that have the best success on the particular site that is being used. Management options should consider this when managing for optimal carbon sequestration. For example, Austrian pine had much better success than green ash on reclaimed mine sites.

Empirical models developed are useful for predicting aboveground biomass in the carbon pool. The equations predicting carbon per tree from basal diameter are all valid for southern Ohio for several species ($R^2 \times 100 > 88$). This high degree of reliability shows that these models can be used in place of destructive sampling in future studies. The equations developed in this study to relate C to basal diameter or DBH are preferable on these sites to equations drawn from the literature since variation may exist between the stands from which the literature values are drawn and the stands under study in this paper.

The differences in carbon content for different species pose questions as to what in the make-up of the trees causes them to have these differences. Pines have a carbon percentage around 52%, while hardwoods have a carbon percentage around 48%. A study in the biochemistry of these species may produce interesting findings in how different species function.

Site conditions and spacing are factors that may be worth considering in future studies. These may have an effect on the carbon sequestered by particular sites. More replications and a broader range of ages would help support the data.

A useful part of this study is the allometric equations derived from the destructive samples. Assuming these values to be statistically significant, which they seem to be, at least for the range of values tested, in most cases, these would make destructive sampling of these species not necessary in future studies of these species unless as a check.

On the whole, this study proved very effective. The results were in general very reliable. The objective of determining the carbon sequestered by species was met with a high degree of reliability. The models should prove useful in future studies.

Acknowledgments

The authors of this paper would like to thank AEP and MeadWestvaco for the use of their lands for this study. The authors also acknowledge The Ohio State University for use of their facilities and equipment in this study. The authors especially would like to thank Drs. Rattan Lal and Randall Heiligmann for their reviews and assistance in making this project better.

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SPATIAL, TEMPORAL, AND RESTORATION TREATMENT EFFECTS ON SOIL RESOURCES IN MIXED-OAK FORESTS OF SOUTHEASTERN OHIO

R.E.J. Boerner and Jennifer A. Brinkman[†]

ABSTRACT. —As part of a larger study of the use of fire and thinning to restore ecosystem function in eastern forests, we quantified spatial, temporal, and treatment-related variations in soil pH, available P, and N mineralization over two years in two southern Ohio mixed-oak forests (Zaleski State Forest and Raccoon Ecological Management Area/REMA). In each site, two watershed-scale treatment units of ~25 ha were sampled on a 50m grid for analysis of spatial autocorrelation and for assessment of temporal variability and treatment effects. Sampling occurred in summer 2000 (prior to treatment) and summer 2001 (after one unit in each site had been thinned+burned). Nutrient status differed more between sites than between treatment units within sites. Semivariance analysis of pretreatment samples demonstrated that pH, available P, and N mineralization were strongly structured spatially in all four treatment units (spatial structure $\geq 67\%$). There were no significant temporal differences in pH or N mineralization (2000 vs 2001) in control areas of the two sites; however, available P did decrease between 2000 and 2001. Neither patch size nor spatial structure changed significantly over this time. Analysis of covariance of post-treatment soil status indicated that the thinning+burning treatment resulted in significant increases in available P and soil pH at REMA but not Zaleski. Semivariance analysis indicated that the thinning+burning treatment increased patchiness (decreased patch size) in pH, but decreased patchiness in available P. Ecosystem restoration treatments affected both overall nutrient status and spatial structure in ways that should influence how plants and communities respond to such treatments.

Introduction

Forests dominated by oak (*Quercus* spp.) and hickory (*Carya* spp.) once covered much of the eastern United States. Prior to extensive alterations of this landscape by Euro-americans in the 19th century, these forests were subject to frequent, low intensity fires (Guyette and Cutter 1991, Sutherland 1997). These fires generally occurred during the dormant season and most were caused by human activities. Analysis of pollen and charcoal profiles suggests that fire return intervals were relatively constant over the last three millenia despite a shift from fewer large, regional fires to a larger number of smaller fires as Native American populations changed (Delcourt and Delcourt 1997). During the 1920's and 1930's fire suppression became both widespread and effective in this region, thus altering the fire regime under which these forests had developed. In addition, these ecosystems have been subjected to a variety of stresses and disturbances which operate on longer time scales, such as removal of a large proportion of standing biomass (e.g. periodic cutting, clearing for agriculture), weather-induced mortality (ice storms, drought), and chronic atmospheric deposition, especially of N. Thus, the mixed-oak and oak-hickory forest ecosystems that exist in this region today have been shaped by a combination of human and natural processes quite distinct from those that these ecosystems experienced for millenia prior to Euro-american settlement.

Restoration of oak forest ecosystems to conditions more similar to those that existed prior to widespread landscape alteration has begun over the last decade in a number of states. Approaches to restoration have utilized passive restoration (restricting public access), functional restoration (reintroduction of dormant season fire), structural restoration (modification of size/age-frequency distributions and species abundances by mechanical means), and, more recently, by a combination of structural and functional restoration.

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When prescribed burning is used for fuel and understory control in commercial conifer plantations, the plots to be burned are often modest in size and relatively homogeneous. Results obtained from burning one part of a larger plantation can then be extrapolated to estimate effects on larger areas by simple, linear scaling. In contrast, the forested landscapes of the Appalachian Mountains and surrounding plateaus which are the focus of current mixed-oak forest restoration efforts are often highly dissected, rugged, and heterogeneous in geomorphology, soils, and microclimate (e.g. Wolfe et al. 1949). In such complex landscapes, a specific understanding of the scale-dependency of various ecological processes and properties is necessary before results from small study plots can be scaled-up to the landscape level.

In earlier studies of the efficacy of single and multiple prescribed burns for restoration of mixed-oak forests in four sites in southern Ohio we demonstrated that fires effects on forest floor and soil C and nutrient relations exhibited strong scale dependencies ranging 10's of km (intersite differences) to 100's of m (aspect and elevation-related differences) (Boerner et al. 2000, Boerner et al. 2004). We have also demonstrated significant spatial heterogeneity and patterning at scales of 0.02-0.2 m in microbial biomass, soil organic matter and soil chemistry in unburned watersheds of this region (Morris and Boerner 1999, Decker et al. 1999); however, to date no data exist with which to determine if the effects of fire or other restoration treatments have significant spatial dependency at ranges between 2 m and 100 m, even though it is within this scale range that most routine sampling takes place. This led us to conduct a set of experiments to determine the degree to which spatial structure at ranges from a few m to a few hundred m was present in these study sites, how that spatial structure might be affected by a combined functional+structural restoration treatment, and how an understanding of spatial structure might affect the determination of the effect/efficacy of such treatments and the design of future restoration efforts.

The specific questions we sought to answer were:

- Do two study sites ~8 km apart vary more than contiguous watershed-scale treatment units within those sites?
- Are contiguous treatment units valid pseudoreplicates for large scale experiments?
- Do soil nutrient properties vary significantly between years in the absence of treatment or disturbance?
- How much do spatially-explicit factors contribute to between-unit and between-year variance?
- To what degree do ecosystem restoration treatments (i.e. reintroduction of low intensity fire and thinning to presettlement tree density) affect soil nutrient properties?
- What proportion of the treatment effect is the result of spatially-explicit factors?

Methods

Study Sites

The two study sites were located in Vinton County on the unglaciated Allegheny Plateau of southern Ohio. Each site was a block of 100-150 ha occupied by mixed-oak forests that developed following cutting for charcoal production 100-150 yr ago. One study site was located in the Raccoon Ecological Management Area (hereafter REMA) (39°11'N, 82°22'W), a research area managed cooperatively by MeadWestvaco Corporation and the U.S.D.A. Forest Service Northeastern Research Station. The second was located in Zaleski State Forest (39°21'N, 82°22'W), a site managed by the Ohio Department of Natural Resources, Division of Forestry. The two study sites were approximately 8 km apart.

The parent materials underlying the study sites were sandstones and shales of Pennsylvanian age. The soils were silt loams formed from colluvium and residuum, and were predominantly Alfisols (Boerner and Sutherland 2003). The climate of the region is cool, temperate and continental with mean annual temperature and precipitation of 11.3 C and 1024 mm (Sutherland et al. 2003). Microclimatic gradients generated by the steep, dissected topography of the region cause S, SW and W facing slopes to be drier and warmer than NW, N and E facing slopes (Wolfe et al. 1949).

A 50 m grid was established within each treatment unit using a random starting point. All grid points were GPS-located and permanently marked. For this study only two of the treatment units within each study area were used. One of the two units within each study area was randomly assigned to be a control and the other to be given restoration treatment consisting of thinning from below to presettlement tree basal area (~16 m²/ha from typical current basal area of 25-30 m²/ha) in November-December 2000 and reintroduction of low intensity, dormant season fire (burned in April 2001). Details of the thinning are given by Yaussy (2001) and fire behavior is described by Iverson and Hutchinson (2002).

Field Methods

Soil samples of approximately 400 g fresh mass were taken of the top 15 cm (O_a+A horizon) at a random point within 1 m of each of the grid points in July of 2000 and 2001. This sampling intensity yielded N=63 and 69 per year for the two REMA treatment units and N=43 and 52 per year for the two Zaleski treatment units. 2001 samples were taken within 50 cm of the 2000 samples, and all samples were returned to the laboratory under refrigeration.

Laboratory Methods

Each soil sample was air dried and sieved to remove roots and particulate material >2mm. A subsample of approximately 15 g of soil was extracted with 0.5M K₂SO₄, and analyzed for NH₄⁺ and NO₃⁻ using microtiter colorimetry (Hamilton and Sims 1995). Soil pH was determined in a 1:5 soil slurry of 0.01M CaCl₂ (Hendershot et al. 1993), and available P by the ascorbic acid method (Watanabe and Olsen 1965).

A second subsample of approximately 50 g was placed in an incubation chamber and artificial rainwater added to bring the soil up to 70% of field capacity. The soil samples were incubated for 27-29 days at 22-28 C. Every third day each soil sample was weighed and sufficient artificial rainwater (Lee and Weber 1979) added to bring the moisture content back to randomly chosen level within the range of 50-70% of field capacity (Morris and Boerner 1998). Laboratory incubations were chosen for use in this study because the manner in which the moisture regime of the incubating samples was maintained recreated the frequent fluctuations in soil moisture characteristic of the growing season in our ecosystems reasonably well (Morris and Boerner, 1998). At the end of the incubation period, a subsample of 15 g of the incubated soil was extracted and analyzed for NH₄⁺ and NO₃⁻ as above. Net N mineralization was determined by subtracting the NH₄⁺ and NO₃⁻ content of the initial samples from that of the incubated samples.

Data Analysis

All response variables could be transformed to normality using a log transformation (PROC UNIVARIATE; SAS 1995). Analyses of variance and covariance (using pretreatment conditions as the covariate) were designed for each of the specific experimental questions listed above, and were accomplished using the GLM procedure of SAS (SAS 1995). Semivariance analysis was accomplished using GS* (Gamma Design Software, Plainwell, MI 49080) using untransformed data, lag distances 0.8 of maximum, and both isotropic and anisotropic models. Spatial structure is reported as the proportion of nugget variance (C_o) + structural variance (C) accounted for by structural variance. The range estimate reported is the direct range estimate (A_o) for linear/sill and spherical models and 1/3 of the estimated range (3A_o) for exponential models. Only best fit models are reported here, and point kriging based on the best fit model was used to interpolate between grid points and visualize spatial pattern.

Results

Partitioning of variance within and between sites indicated that potentially limiting soil properties varied significantly between sites (i.e. REMA vs Zaleski) but not between treatment units within each of the sites (table 1). During the pretreatment year (2000) soil properties varied little between the two treatment units within each study site (table 1).

Table 1. —Analysis of variance comparing variations between study sites ~8 km apart to those between treatment units within a site. Overall N=227, N=132 for REMA, and N=95 for Zaleski.

	Between sites	Between units within sites
Soil pH	F=16.11, p<0.001	F=0.88, p<0.417
Available P	F=7.85, p<0.006	F=1.65, p<0.195
N mineralization	F=7.38, p<0.008	F=0.35, p<0.704
	Between units at REMA	Between units at Zaleski
Soil pH	F=1.23, p<0.270	F=0.37, p<0.544
Available P	F=1.14, p<0.289	F=3.80, p<0.066
N mineralization	F=0.61, p<0.437	F=0.01, p<0.907

Semivariance analysis indicated that 73-80% of the variation in soil pH, available P, and N mineralization rate in REMA soils in 2000 was attributable to spatial structure, and this differed little between units within the REMA study site (table 2). At REMA, pH, available P, and N mineralization reached their maximum in the northern portion of the treatment unit (fig. 1); however the actual positions of maxima in space differed among soil parameters within a unit. This was a common result across all treatment units and years we sampled. Patch size (defined by the maximum range of significant spatial autocorrelation) was greater in the REMA control unit (120->369m) than in the REMA thin+burn unit (54-98m) (table 2) and these differences in patch size were clearly apparent in the kriged maps (fig. 1).

At Zaleski, >67% of the variance in soil pH, available P, and N mineralization rate were attributable to spatial structure (table 2). The pattern in space and maximum patch size of soil pH and available P in pretreatment soils differed little between the control and the unit to be thinned and burned (fig. 2). Small, distinct patches of low and high N mineralization rate soils were present in the kriged maps of the Zaleski control but not the Zaleski thin+burn, and as a result patch size was smaller in the former than the latter (fig. 2, table 2).

In the absence of treatment and/or disturbance, soil pH and N mineralization rate did not differ significantly between 2000 and 2001 samplings in control units in the two sites (REMA pH/2000: 3.89 ± 0.08 [std. error] vs pH/2001: 3.77 ± 0.06 , $p<0.161$; N mineralization/2000: 12.51 ± 0.91 mgN/kg soil/dy vs N mineralization/2001: 11.33 ± 0.86 , $p<0.346$, and Zaleski pH/2000: 3.63 ± 0.06 vs pH/2001 3.73 ± 0.05 , $p<0.220$; N mineralization/2000: 10.46 ± 1.07 mgN/kg soil/dy vs 10.65 ± 1.03 , $p<0.899$). In contrast, there was significantly less available P in 2001 than in 2000 in the control units in both sites. Available P decreased from 2000 to 2001 by 33% at Zaleski (2000: 231.4 ± 21.3 μ gP/kg soil vs 2001: 154.3 ± 19.3 , $p<0.009$) and 69% at REMA (2000: 348.1 ± 23.0 μ gP/kg soil vs 2001: 109.0 ± 11.3 , $p<0.001$).

The proportion of total variance attributable to spatial structure and the maximum patch size of soil pH and N mineralization differed little from 2000 to 2001 in the control units at the two sites, with the sole exception of pH patch size at REMA (figs.1 and 3, table 2). The latter difference in patch size was the result of the few, scattered patches of relatively high or low pH scattered within the larger matrix of intermediate pH in that unit. The decrease in available P from 2000 to 2001 noted earlier was apparent in the kriged maps, but neither the degree of spatial structure nor the patch size changed over that time in either site (table 2).

Analysis of covariance of the effect of the thinning+burning treatments at the two sites (using pretreatment conditions as covariates) indicated that only soil pH was affected by treatment alone (table 3). There were significant interactive effects of treatment and study site on both soil pH and available P, whereas N mineralization rate was not affected significantly by the thin+burn treatment

Table 2.—Semivariance analysis of soil parameters in samples taken in 2000 (pre-treatment) and 2001 (posttreatment) in two treatment units within each of two forested sites in Ohio. N=63,69 for REMA and N=43,52 for Zaleski. Spatial structure is the percent of nugget semivariance + structural semivariance represented by structural semivariance only. Range is an estimate of patch size as defined by the maximum distance at which samples are spatially autocorrelated.

Site/Unit	Year	Model fit (r ²)	Spatial structure	Range (m)	Model form
<u>Soil pH</u>					
REMA/Control	2000	0.783	81.4%	>369	exponential
REMA/Thin+Burn	2000	0.462	85.7%	98	spherical
Zaleski/Control	2000	0.424	67.4%	183	exponential
Zaleski/Thin+Burn	2000	0.872	68.0%	181	spherical
REMA/Control	2001	0.070	79.5%	>369	linear
Zaleski/Control	2001	0.773	99.9%	195	spherical
REMA/Thin+Burn	2001	0.472	93.5%	131	exponential
Zaleski/Thin+Burn	2001	0.279	78.5%	41	exponential
<u>Available P</u>					
REMA/Control	2000	0.254	76.6%	120	linear/sill
REMA/Thin+Burn	2000	0.086	80.0%	54	exponential
Zaleski/Control	2000	0.820	99.9%	68	spherical
Zaleski/Thin+Burn	2000	0.239	99.9%	54	linear/sill
REMA/Control	2001	0.503	83.1%	116	exponential
Zaleski/Control	2001	0.313	92.5%	61	spherical
REMA/Thin+Burn	2001	0.967	65.6%	261	linear
Zaleski/Thin+Burn	2001	0.022	84.0%	50	spherical
<u>N mineralization</u>					
REMA/Control	2000	0.884	73.8%	>369	exponential
REMA/Thin+Burn	2000	0.621	73.3%	62	exponential
Zaleski/Control	2000	0.805	99.8%	147	spherical
Zaleski/Thin+Burn	2000	0.995	66.5%	357	spherical
REMA/Control	2001	0.920	79.9%	294	spherical
Zaleski/Control	2001	0.701	97.8%	138	spherical
REMA/Thin+Burn	2001	0.781	53.0%	374	spherical
Zaleski/Thin+Burn	2001	0.360	79.4%	69	exponential

Table 3.—Analysis of covariance of soil parameters in relation to study site, restoration treatment, and the interaction between the two, using pretreatment conditions as the covariates. N=227.

	Treatment	Site-by-treatment interaction
Soil pH	F=4.56, p<0.034	F=5.61, p<0.019
Available P	F=1.87, p<0.174	F=12.12, p<0.001
N mineralization	F=1.99, p<0.166	F=0.66, p<0.419

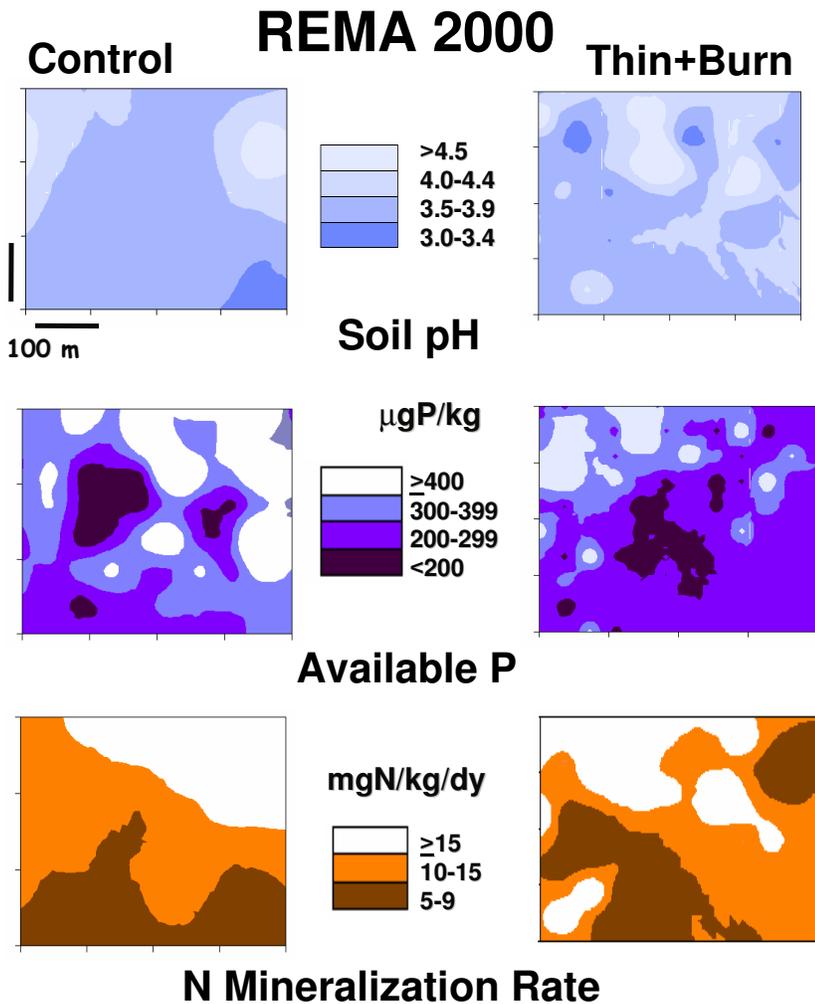


Figure 1.—Interpolated maps of soil parameters in the REMA Control and Thin+Burn units during the pretreatment year (2000). See Table 2 for spatial statistics associated with these kriged maps.

either overall or in relation to site (table 3). Overall, soil pH in the thin+burn units (3.85 ± 0.03) exceeded that of the control units (3.75 ± 0.01) significantly, and at REMA the difference of 0.24 pH units between the control (3.77 ± 0.06) and thin+burn (4.01 ± 0.07) units was significant at $p < 0.001$. At REMA, available P in the treated unit ($179.9 \pm 20.3 \mu\text{gP/kg}$ soil) was 65% greater than that of the control unit (109.0 ± 11.3), whereas at Zaleski available P in the treated unit (110.0 ± 11.1) was 29% lower than that of the control (154.3 ± 19.3); the difference at REMA was significant ($p < 0.006$) whereas the one at Zaleski was not ($p < 0.195$).

Maximum pH and available P patch sizes increased from 2000 to 2001 by 25% and 475%, and the proportion of variance in available P attributable to spatial structure decreased by approximately 1/3 over that time (table 2). The increase in available P from 2000 to 2001 in the REMA treatment unit was apparent in the kriged maps as patches of moderate and high P appeared after treatment in what was a matrix of low soil P prior to treatment (figs. 1 and 3). Although the structural variance in soil pH and available P changed little from 2000 to 2001 in the treated unit at Zaleski, maximum patch size for pH appeared to decrease by approximately 77% (table 2). At Zaleski, smaller patches of relatively high soil P that were present prior to treatment were absent after treatment (figs. 2 and 4). The spatial structure of N mineralization changed little from 2000 to 2001 in either study site (table 2, figs. 2 and 4).

Discussion

The assessment of the success of ecosystem restoration prescriptions often focuses on determining whether predetermined, average conditions have been achieved. It is our premise that ecosystem

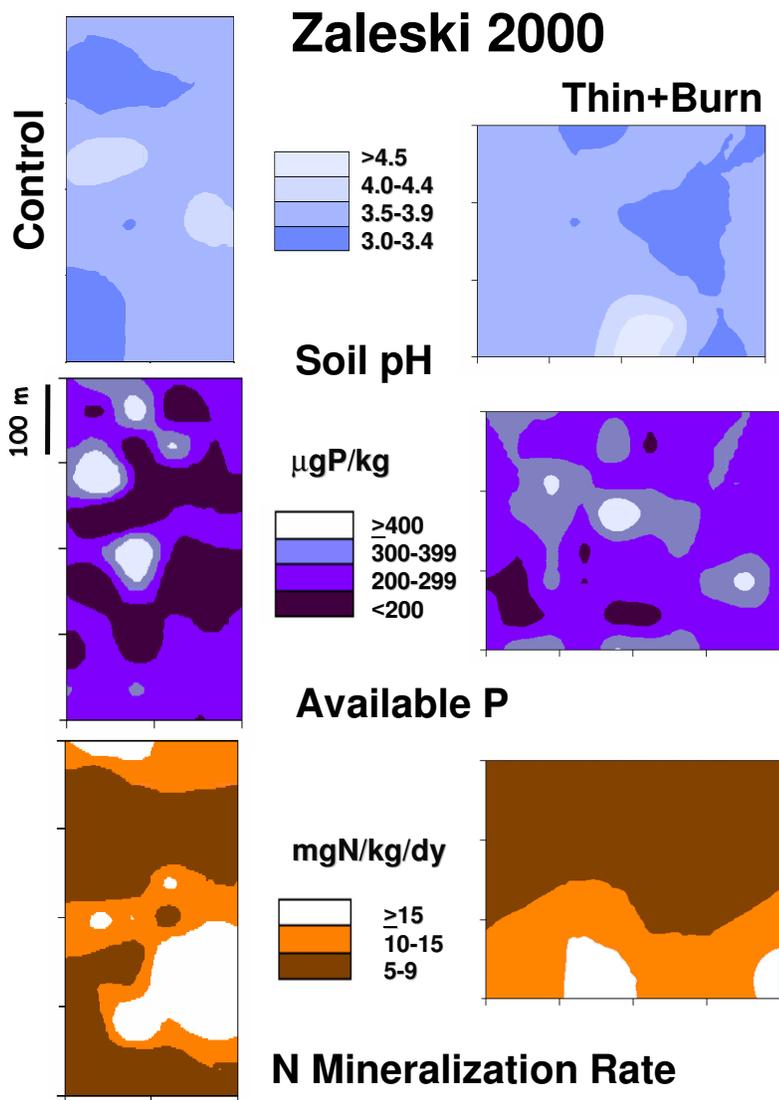


Figure 2.—Interpolated maps of soil parameters in the Zaleski Control and Thin+Burn units during the pretreatment year (2000). See Table 2 for spatial statistics associated with these kriged maps.

restoration at the landscape scale requires and additional understanding of, and an ability to measure, spatial heterogeneity at multiple scales. Even in regions where tree diversity and topographic heterogeneity are less than in our Ohio landscape, distinct spatial structure in tree stem and age distributions were common prior to Euro-american intervention (Selmants et al. 2003), and based on the well-documented effects of individual trees on soil properties near their bases (Boerner and Koslowsky 1989; Decker et al. 1999), it is reasonable to postulate that such spatial structuring existed in the forest floor and soil as well.

The first part of this study was designed to determine what the spatial pattern of potentially-limiting soil resources was in this heterogeneous landscape prior to the onset of restoration treatments, and to determine how stable this spatial pattern was in time and space. As has been the case in our prior studies in this region (e.g. Boerner et al. 2003, 2004, Boerner and Brinkman 2003), we found that mean soil properties varied much more between study sites separated by km than between neighboring watersheds. Similarities between/among neighboring watershed-scale treatment units in this region (this study, Boerner et al. 2003, Boerner and Sutherland 2003) lend strong validity to the use of such large land units as replicates (or pseudoreplicates) for experimentation, while the significant differences between sites just a few km apart (Boerner et al. 2003) argue strongly for approaching such experiments using complete block or Latin Square designs.

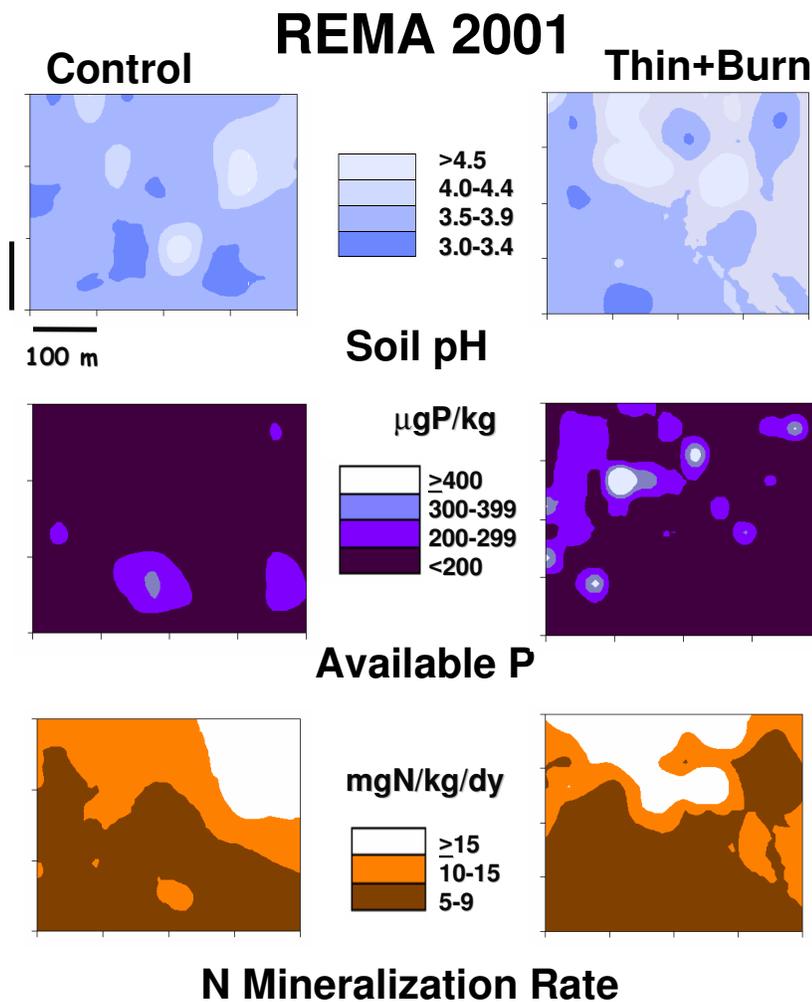


Figure 3.—Interpolated maps of soil parameters in REMA Control and Thin+Burn units during 2001. See Table 2 for spatial statistics associated with these kriged maps.

The soil properties we chose for measurement all exhibited strong spatial structure at ranges of 15-360 m across the landscape, with at least two-thirds of the total variance among samples taken in each treatment unit of ~25 ha being attributable to spatial structure. The patch size at which this spatial structure occurred (defined as the maximum range of significant spatial autocorrelation among samples) varied by a factor of 2-6 among soil parameters between the two treatment units at REMA, and by a factor of approximately 2.4 for N mineralization at Zaleski. At REMA, apparent patch size was always greater in the control unit than in the unit to be thinned and burned, and this difference correlates well with the geomorphology of the two units. Approximately half the control unit is occupied by a relatively flat area classified as xeric in the Integrated Moisture Index (IMI) system of Iverson et al. (1997), whereas the unit to be thinned and burned exhibits a more equitable distribution of relatively xeric, intermediate, and relatively mesic IMI areas. Thus the geomorphology of the control is characterized by fewer and larger patches than is the treatment unit, and this carried over into the soil properties we quantified. In contrast, the difference in patch size in N mineralization at Zaleski was due to the presence of a few very small but distinct patches of relatively higher or low activity nested within larger areas of intermediate activity being present in the control unit but not the treatment unit. These small, unique patches do not correlate well with geomorphological features and are more likely the result of finer-scale variations in organic matter deposition or content, as has also been documented in other forest types (e.g. Boerner and Koslowsky 1989, Bruckner et al. 1999).

The relatively low range in soil pH (3.6-4.4) we observed was notable, as alfisols and inceptisols formed on these parent materials are often typified as having pH of 4.0-5.0. To some degree, the

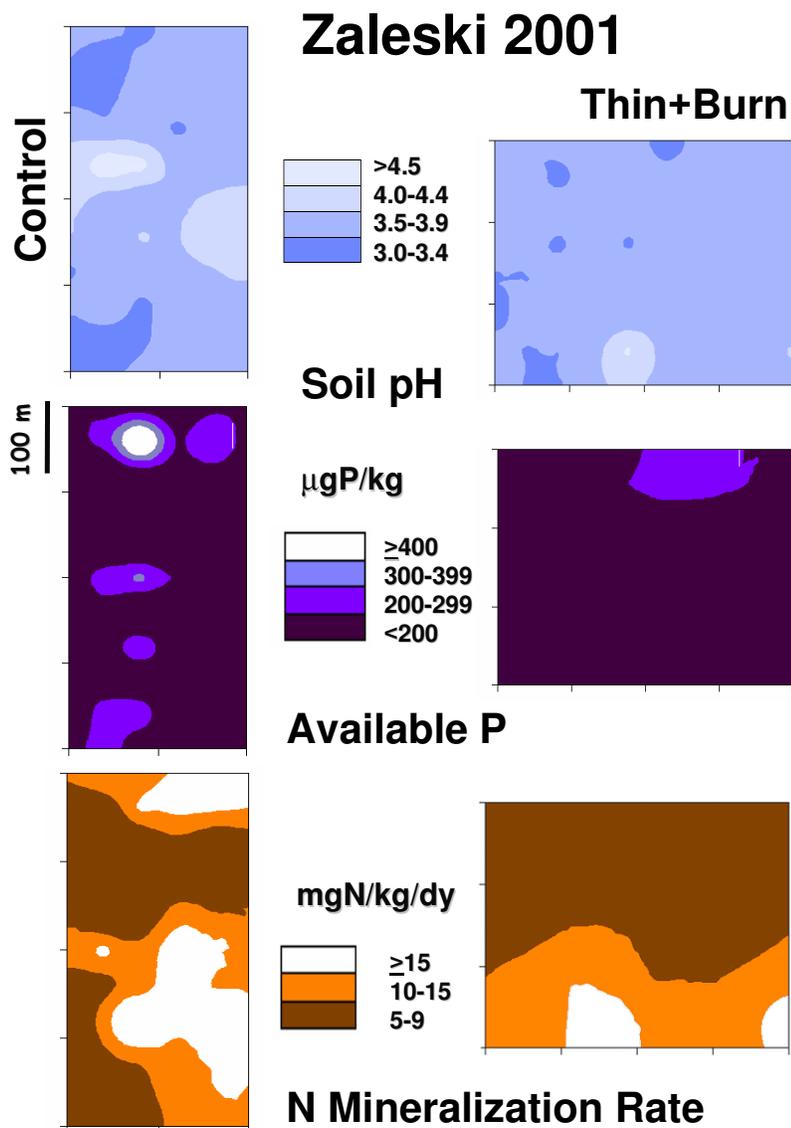


Figure 4.—Interpolated maps of soil parameters in Zaleski Control and Thin+Burn units during 2001. See Table 2 for spatial statistics associated with these kriged maps.

relatively low pH we observed may have been a byproduct of differences in methodology, as pH values in CaCl_2 (as we used) can be 0.5-0.7 pH unit lower than those recorded in water. However, enhanced acidification induced by chronic deposition of N in precipitation and resultant enhanced rates of nitrification may also be a contributing factor. For example, Boerner and Brinkman (2004) report similar soil pH, soluble Al concentrations of 100-400 mg/kg soil, and molar Ca:Al ratios of <1 in soils of sites located contiguous to REMA in which nitrification rates have increased 4-10 fold over the last decade.

We observed little interannual variation in soil pH or N mineralization in the control treatment units, though available P avail did decrease in both control units from 2000 to 2001. Although the magnitude of the year-to-year variation in available P was statistically significant and appeared relatively large on a proportional basis, it represented an average change of less than 0.10-0.25 mgP/kg soil, a degree of temporal variability that was small compared to spatial variability within a treatment unit. In addition, spatial structure and patch size of soil pH, available P, and N mineralization differed little between years. If anything, we were surprised by how little interannual variation in soil nutrient status we observed given how variable our summer weather often is from year to year.

Studies of spatial dependency in soil biological and chemical properties have reported a broad range of patch sizes/spatial autocorrelation ranges. Among those soil properties reported to exhibit spatial dependency at scales <1 m are soil pH in a Norway Spruce (*Picea abies*) forest (Bruckner et al. 1999), soil pH in Norway spruce, sugar maple (*Acer saccharum*), and red pine (*Pinus resinosa*) forests (Riha et al. 1986), soil pH, N, and K in a sugar maple forest (Lechowicz and Bell), soil organic C in Ohio beech-maple (*Fagus grandifolia*-*Acer saccharum*) forests and successional old fields (Boerner et al. 1998), and both nitrification and denitrification in a white fir (*Abies alba*) forest (Lensi et al. 1991). Spatial patterning at scales of 1-10 m are reported for inorganic N and available P in Ohio beech-maple forests and successional old fields (Boerner et al. 1998), inorganic N and organic C in a Douglas-fir (*Pseudotsuga menziesii*) forest (Antos et al. 2003), and C mineralization and soil moisture a red maple (*Acer rubrum*) forest and successional old field in Rhode Island (Görres et al. 1998). Finally, coherent spatial structure at coarser spatial scales have also been reported, including ranges of 10-20 m for P, Ca, Mg, and CEC in North Carolina forests (Palmer 1990), 10-20+ m for soil pH, P, and organic matter in an old field in Italy (Castrignano et al. 2000), >20 m for N mineralization in a Norway Spruce forest (Bruckner et al. 1999), 8-62 m for soil moisture in a slash pine (*Pinus elliotti*) plantation in South Carolina (Guo et al. 2002), up to 20 m for inorganic N in a Michigan successional field (Robertson 1987), and 13-42 m for soil pH N mineralization, and available P in a tropical dry forest in the West Indies (Gonzalez and Zak 1994). One must be cautious in attempting to synthesize these studies into a general model for hierarchic spatial dependency in soil resources as the diversity of study sites and, perhaps most importantly, differences in the sampling schemes and scales of possible spatial resolution differ so much among studies.

Although nested or hierarchic spatial scales are commonly postulated as being common in the field (e.g. Palmer 1990, Ettema and Wardle 2002, Franklin and Mills 2003), the detection of multiple scales of variation requires complex, hierarchic sampling designs that are uncommon in the literature. In most cases, the range at which spatial dependency can be and is reported is severely constrained by the sampling design to a single, relatively narrow range. In this study, we report observations of spatial dependency at ranges of 40-200 m for soil pH, 50-260 m for available P, and 14-375 m for N mineralization. The limitations of our sampling design limited us to detection of ranges from 10-400 m, and would thus miss entirely fine-scale spatial dependency based on single tree influences (e.g. Boerner and Koslowsky 1989, Bruckner et al. 1999) or patterns of coarse woody debris on the forest floor (e.g. Morris 1999) and coarse-scale variations based on differences in parent material among sites separated by km (Boerner et al. 2003, 2004). However, combining this study with those done earlier in our study sites at coarser (Decker et al. 1999, Boerner et al. 2000, Boerner et al. 2003, 2004, Boerner and Brinkman 2003) and finer scales (Decker et al. 1999, Morris and Boerner 1999, Morris 1999) allow us to establish the hierarchy of spatial scaling of soil properties we feel is necessary to fully evaluate the efficacy of broad-scale ecosystem restoration and management efforts.

The second portion of the study was designed to determine the initial effects of a combined structural and functional restoration treatment on soil resources, both overall and in relation to spatial structure. Overall, we found that the combined restoration treatment resulted in increased soil pH and available P at REMA, but not at Zaleski. Fire-induced increases in soil pH, available P, and base saturation have been reported commonly in eastern forests (review by Boerner 2000), and a similar increase in pH was observed following fires in studies in nearby watersheds (Boerner et al. 2004). Variations in this response among studies in the literature are the result of differences in initial base saturation, fire intensity, and the length of time since fire (Boerner 2000). Our intersite differences seem to be predominantly the result of differences in fire behavior, with the fire at the REMA site having been somewhat more intense than that at the Zaleski site (Iverson, unpublished data).

We observed no change in N mineralization rates after thinning+fire in these two sites. These results were consistent with those following fire only in neighboring study sites (Boerner et al. 2004), but clearly not with studies in a variety of other ecosystems which show increase in N availability after fire or fire+cutting. For example, fire-induced increases in TIN and/or N mineralization have been reported in *Pinus ponderosa* forests in western North America (Wagle and Kitchen 1972), mixed pine (*P. echinata*

and *P. taeda*) forests in east Texas (Webb et al. 1991) and California chaparral (Debano et al. 1979). Knoepp and Swank (1993) reported increases in N mineralization that persisted at least two yr after cutting and burning, though the fires employed in that study were considerably more intense than those in our study sites. In a Douglas-fir forest in the Pacific Northwest, cutting and burning resulted in an initial increase in N availability, but it lasted less than one yr (Antos et al. 2003), and Phillips and Goh (1985) reported that the increases in NH₄ production were greater in a southern beech (*Nothofagus* spp.) site in New Zealand that was cut and burned than in a site that was only cut. One must use caution in extrapolating the effects of fire in one ecosystem to those in far different ones. For example, Vance and Henderson (1984) found that annual burning over 30 yrs in a Missouri oak forest reduced N mineralization and TIN to a greater extent than did periodic burns over the same period, and the sites studied by Vance and Henderson (1984) are more similar to our study sites than are the others cited as experiencing an increase in N mineralization after fire.

Our thinning and burning treatment tended to homogenize the soil nutrient distribution. We noted decreases in spatial structure, increases in patch size, and the loss of very small, distinct patches of differing nutrient availability. This result is consistent with those of Guo et al. (2002) who observed a loss of spatial structure in soil characteristics following cutting in a South Carolina slash pine stand. If the short term results we present here persist over a longer period, there may be consequences for community structure in these forests. Studies in southern forests have demonstrated that human-induced change in spatial structure of forests can affect ecological processes such as the spread of fungal diseases and insect pests (Perkins and Matlack 2002). We will continue to evaluate the effects of these treatments on spatial structure of soil resources through a second fire cycle in expectations of determining whether the changes we documented in this study are persistent or ephemeral.

Acknowledgments

We thank Bob Ford, Elizabeth Swinehart, and Ryan Strachan for field assistance, and Marci Knox and Adrea Lovejoy for lab assistance. This is contribution number 41 of the National Fire and Fire Surrogate Research Project (FFS). This research was funded through the National Fire Plan of the U.S.D.A. Forest Service. Although the authors received no direct funding from the U.S. Joint Fire Science Program (JFSP), this research could not have been accomplished without JFSP support of the existing FFS research project.

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SOIL COMPACTION AFFECTS GROWTH OF YOUNG SHORTLEAF PINE FOLLOWING LITTER REMOVAL AND WEED CONTROL IN THE MISSOURI OZARKS

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ABSTRACT.—More frequent use of heavy equipment in intensive forest practices can lead to soil compaction and reduced productivity. The growth of 8-year-old planted shortleaf pine (*Pinus echinata* Mill.) and seasonal soil moisture stress and soil temperature were measured on cherty silt loam soil from which surface organic materials (whole tree plus leaf litter) had been removed before applying compaction treatments. Three levels of compaction and two levels of understory control were compared. Trees were significantly taller and had more height and diameter at breast height (d.b.h.) growth in compacted treatments than in the no compacted treatment and when the understory was absent than when present. Mean seasonal soil moisture stress was much higher for compacted treatments than for the no compaction treatment during the measurement period, except for September. But differences were significant only for May (severe compaction>no compaction) and September (severe compaction<no compaction). Soil temperature, although higher for compacted treatments than for the no compaction treatment, was not significantly different between treatments. Both soil moisture stress and soil temperature were higher when the understory was present than when absent.

Compaction affects nearly all properties and functions of the soil, physical as well as chemical and biological. Compaction causes a rearrangement of the soil particles resulting in the alteration of pore size distribution, decreased porosity, and changes in the movement and content of heat, air, water, and nutrients in the soil (Grable and Siemer 1968). These changes in bulk density affect infiltration, drainage, water availability, aeration, root exploration, and nutrient uptake, all of which can directly influence soil productivity.

The compaction of soils during forestry operations reduces the rate of establishment of natural regeneration, has been shown to reduce tree growth for periods spanning at least a decade, and can have deleterious effects on tree form (Wronski and Murphy 1994). Forest harvesting, especially whole tree harvesting and litter raking, can remove considerable biomass, leaving the residual forest floor significantly disturbed. Severe compaction and organic matter removal can affect both the quality and productivity of the soil. Studies conducted in North America have shown that tree growth and forest productivity decrease due to compaction. For example, the reduction in ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) height and volume growth attributed to soil compaction in southwest Oregon was still evident 17 years later (Froehlich 1979). On the Coastal Plains of South Carolina, one-year-old loblolly pine height growth on compacted skid trails was 40 to 50 percent less than height growth on non-compacted soil (Hatchell and others 1970).

While there is ample evidence supporting the reduction in growth caused by soil compaction, it has not always produced negative growth effects (Miller and others 1996, Cochran and Brock 1985). For example, on several California sites, the effect of compaction on 4-year-old ponderosa pine varied with soil texture and soil water (Gomez and others 2002a). These authors reported that compaction reduced the growth young ponderosa pine in California on fine textured soils due to increased soil strength, but soil compaction increased growth on a sandy textured soil due to increased water holding capacity.

Coile (1948) reported that the growth of shortleaf pine increased as soil available water increased in the subsoil in the North Carolina Piedmont. Mohr (1896) concluded that sands were a poor growth medium for shortleaf and on the better sites subsoil textures were heavier than topsoil textures. Each soil has unique properties that interact to respond to compaction in different ways to affect tree growth.

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Understanding the effects of compaction on site productivity is among many factors affecting the development of young trees.

The specific objectives of this work were to: (i) assess the height and diameter response of shortleaf pine to soil compaction after 8 years and (ii) assess the effect of understory treatments with and without compaction on height and diameter of shortleaf pine.

Materials and Methods

Site Description

The study is in one of the USDA Forest Service's Long-Term Soil Productivity (LTSP) sites and located in the Carr Creek State Forest in Shannon County, Missouri. The LTSP study, composed of large-scale field experiments located at sites across the United States, was developed to assess the effects of soil compaction and surface organic matter removal on site productivity across a range of forest sites (Powers and others 1990). The silt loam soils on the Missouri site are primarily of the Clarksville series (Loamy-skeletal, mixed mesic Typic Paleudults). Soils are derived from Ordovician and Cambrian dolomite with some areas of Precambrian igneous rock (Missouri Geological Survey 1979). Weathered material has formed a deep mantle of cherty residuum (Gott 1975). Initial soil chemical properties of the 0-20 cm depth were: pH (1:1 water) 5.7; total C, 3.3 %; total N, 0.11%; P, 16.9 mg/kg; Ca, 789 mg/kg; and Mg, 61 mg/kg (Ponder and others 2000). Prior to harvest, the site had a well-stocked, mature, second-growth oak (*Quercus* spp.)-hickory (*Caraya* spp.) forest. The site index indicated that the height of 50-year-old black oak (*Q. velutina* Lam.) should range from 74 to 80 feet on this site (Hahn 1991). Mean annual precipitation and temperature is 112 cm and 13.3° C, respectively (Barnton 1993). Moisture drains easily through the soil to subsurface channels.

Experimental Design

The LTSP study includes nine treatments derived from combinations of three levels each of organic matter removal and soil compaction. The three levels of organic matter removal included: (1) merchantable boles removed (boles only), (2) all living vegetation removed (whole tree), and (3) all living vegetation plus forest floor removed, exposing mineral soil (whole tree + forest floor). Merchantable boles included trees with diameters at breast height (d.b.h.) of 25 cm or larger. The three levels of compaction included: (1) no compaction (C_0), (2) moderate compaction (C_1), and (3) severe compaction (C_2). The targeted bulk density of severe compaction treatment was an increase of 30% more than the bulk density of the no compaction treatment. The moderate soil compaction treatment was intermediate between the severe compaction and no compaction treatments. The latter was accomplished by using heavy road construction equipment. Mean bulk density increased to up to 1.8 g cm³ compared to 1.3 g cm³ for the no compacted treatment. A complete description of the site and the LTSP installation are provided elsewhere (Ponder and Mikkelsen 1995).

For this report, three levels of compaction and two levels of vegetation management were used. For the first 2 years after planting, a 3-foot radius area around each shortleaf pine seedling was sprayed annually in the spring with a mixture of glyphosate and simazine mixture to control weeds. Beginning in the 3rd growing season, half of each plot was kept weed-free (understory absent) to permit planted trees to grow freely without weeds. Weeds were not controlled (understory present) in the other half of the plot.

Seedling heights were measured after planting and annually thereafter. Diameter at breast height was measured when trees reached 1.4 m tall or taller. Soil moisture stress (soil moisture resistance) and temperature to a depth of 30 cm were measured monthly during the growing season using Soiltestä moisture cells in the no compaction and severe compaction treatments at 10 cm increments from 0 to 30 cm deep for some years. Current-year shortleaf pine leaves were collected in August of years 4 and 8 for macronutrient analyses. For this report, 5- and 6-year data for soil moisture stress and temperature and 4- and 8-year data for leaf nutrient concentrations were used. Data were not available for all years for leaf nutrients nor soil moisture stress and temperature. All measurements and analyses were done according to standard procedures. Rainfall measurements were also recorded at the weather station on the study site.

Table 1.—The effect of soil compaction and understory on mean annual survival of 8-year-old shortleaf pine in Missouri.

Treatment	Year							
	1	2	3	4	5	6	7	8
Soil compaction	Survival (Percent)							
No	81	81	78	75	74	72	72	72
Moderate	67	66	66	66	66	66	66	65
Severe	75	74	74	74	72	72	70	70
Understory								
Absent	73	73	72	72	70	69	69	69
Present	74	74	73	72	72	72	72	70

Table 2.—The effect of soil compaction and understory control on 8-year height and d.b.h. (diameter 4.5 ft above the ground) growth of shortleaf pine planted after forest floor removal.¹

Treatment	Total height growth	Total d.b.h. growth
Soil compaction	cm	cm
None	273±171a ²	32±13a
Moderate	304±136a	34±12a
Severe	373±132b	39±13b
Understory		
Absent	310±151a	40±13a
Present	324±142a	31±12b

¹Values after the ± symbol are mean standard deviations.

²For soil compaction or understory treatment, columnar means followed by the same letter are not significantly different at $\alpha = 0.05$ Tukey's Studentized Range test.

Statistical Analyses

The experiment was analyzed as a split-plot design with three levels of soil compaction two levels of understory as the subplots. Survival was analyzed using the PROC LIFETEST procedure described in "Survival Analysis Using the SAS System" (Allison 1995). Annual tree survival data were coded according to tree status, 1 for live trees and 0 for dead trees and subjected to an ANOVA. Survival differences among treatment variables were tested using the Tukey's Studentized Range test at the $\alpha = 0.05$ level of significance. Total height and d.b.h. growth through year 8, leaf nutrients for years 4 and 8, and soil moisture stress and soil temperature were analyzed using analysis of variance with the PROC GLM procedures in SAS Version 8.2 (SAS Institute, Cary, NC). Prior to analysis, soil moisture stress and soil temperature data were averaged for the fifth and sixth growing seasons. Data for all depths were combined. All statistical tests were performed at the $\alpha = 0.05$ level of significance.

Results

Survival and Growth

Eight years after planting, survival was not significantly affected by soil compaction or understory treatments (Table 1). Most of the mortality occurred during the first year. This mortality may have been due to the less than ideal soil moisture during the planting period (May through June). Survival after year 1 through year 8 declined only by 10, 2 and 5 percent for no, moderate, and severe compaction treatments, respectively. For understory treatments, survival at the end of the second year was 73 and 74 percent for understory absent and understory present, respectively and declined by 4 percent between year 1 and year 8 for both understory treatments. The interaction between compaction and understory was not significant.

Total height and diameter growth were significantly better for compaction treatments than for the no compaction treatment (Table 2). Differences between moderate and no compaction treatments were not significant for either measurement at the end of 8 years. Differences in tree height between

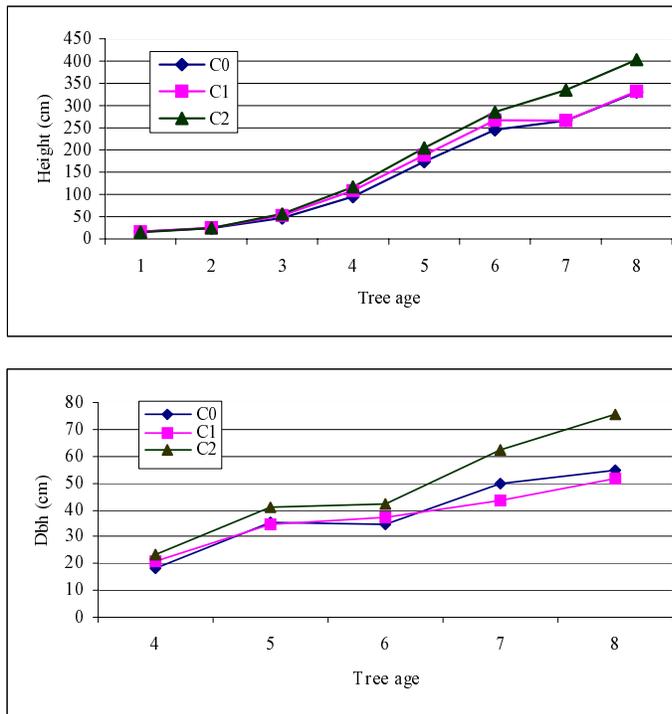


Figure 1.—Annual height and diameter at breast height (d.b.h.) for shortleaf pine growing in compacted soil (no compaction=C0, moderate compaction=C1, severe compaction=C2) in the Missouri Long-Term Soil Productivity (LTSP) study.

treatments began to appear between years 4 and 5, when most trees were tall enough to measure d.b.h. (Fig. 1). The presence or absence of an understory was significant only for d.b.h. growth. Shortleaf pine with the understory absent had more d.b.h. growth than when the understory was present (Table 2). Interaction between soil compaction and understory treatments was not significant.

Leaf Nutrient Concentration

Leaf macronutrient content is presented for the fourth and eighth growing seasons (Table 3). Soil compaction had no significant affect on leaf nutrient concentrations in the fourth or sixth growing season, but understory did (Table 3). Leaf N was higher when the understory was absent than when the understory was for both growing seasons. Calcium and Mg were significantly higher for shortleaf pine leaves in the understory present treatment than in the understory absent treatment for fourth growing season but not after six growing seasons. Both P and K were higher for leaves from trees in the understory absent treatment than in the understory present treatment for year 8, but not for year 4. There was a significant interaction for Ca in year 6 (Fig. 2A). Leaf Ca was lower when understory was absent compared to when present for severe compaction. The reverse was true for the no compaction treatment.

Soil Moisture Stress

Soil moisture stress and soil temperature data are presented only for two treatments; no compaction and severe compaction (Table 4). Soil moisture stress readings were higher for severe compaction than for no compaction, except for September. However, readings differed significantly between treatments only for the months of May and September. Differences between understory treatments were significant for all months. But, there were significant interactions for both May and September between understory and soil moisture stress (Figs. 2B and 2C). The interactions showed that regardless of compaction treatment, moisture stress was highest when understory was present. For May, the mean soil moisture stress measurement for severe compaction with understory present was $84.7 \text{ ohms } 10^{-1}$ and $29.5 \text{ ohms } 10^{-1}$ when understory was absent compared to an average of $17.0 \text{ ohms } 10^{-1}$ for these understory treatments and no compaction. For September, mean soil moisture stress was considerably

Table 3.—The effect of soil compaction and understory on leaf macro nutrient concentration of shortleaf pine during the fourth and eighth growing season.

Treatment	Nutrient				
	Nitrogen	Phosphorus	Potassium	Calcium	Magnesium
Fourth growing season					
Soil compaction					
No	14.45a ¹	0.88a	6.39a	2.60a	0.71a
Moderate	14.38a	0.96a	6.12a	2.06a	0.67a
Severe	14.91a	1.02a	6.52a	2.02a	0.66a
Understory					
Absent	15.34a	0.96a	6.53a	1.90a	0.63a
Present	13.91b	0.95a	6.18a	2.50b	0.72b
Interaction	0.3703 ²	0.343	0.862	0.568	0.607
Eighth growing season					
Soil compaction					
No	14.97a	1.07a	5.60a	1.76a	0.84a
Severe	14.90a	1.08a	5.59a	2.20a	0.93a
Understory					
Absent	16.57a	1.12a	5.99a	1.90a	0.86a
Present	13.29b	1.03b	5.20b	2.05a	0.91a
Interaction	0.2186	0.634	0.194	0.007	0.067

¹For soil compaction or understory treatment, columnar means followed by the same letter are not significantly different at $\alpha = 0.05$ Tukey's Studentized Range test.

²*p-value.*

higher for no compaction with understory present (158.2 ohms 10^{-1}) than for severe compaction with understory present (80.7 ohms 10^{-1}).

Soil temperature was generally a little higher in the severe compaction treatment than in the no compaction treatment but differences were not significant except for August (Table 4). Differences in soil temperature between understory treatments were significant for all months; soil temperature was higher when the understory was present than when absent. Interactions were not significant.

Discussion

Bulk density data were not presented here but were presented earlier (Ponder and others 1999). The data showed that severe soil compaction effectively increased soil bulk density over no soil compaction. The percent change in bulk density between the no compaction treatment and the severe compaction treatment was 22, 29, and 26 percent for the 10-, 20-, and 30-cm depth increments, respectively. While the bulk density of the moderate compaction treatment was increased above that for the no compaction level, it was less than that for severe compaction. At a given bulk density, soil water content determines root growth potential by influencing soil strength, aeration, and plant available water. Neither soil strength nor plant available water content was measured for this report. However, in a laboratory study, Siegel-Issem (2002) found that compaction moderately increased available water and decreased aeration porosity in the Clarksville soil from this site.

Soil compaction was beneficial to the growth of shortleaf pine growing in the Clarksville silt loam on this Ozark site. However, neither soil moisture stress nor soil temperature data during the months of May thru September explain why. These data showed compacted plots usually had both higher soil

moisture stress and temperatures most of the measurement period (Table 4), indicating that compacted plots were drier and hotter than plots with no soil compaction. As soils became drier, soil temperatures increased, heating the air above the soil surface and increasing evaporation. Sauer and others (2002), in a water balance examination, reported that evaporation accounted for 91 percent of the rainfall that fell on a Clarksville soil in Arkansas dominated by tall fescue (*Festuca arundinacea* Schreb.) cover. Excavations at their site showed that the greatest plant root density was near the surface and that the soil layers above 0.5 m appear to be primarily responsible for supplying water for evaporation.

Soil moisture stress variation from year to year depends a lot on rainfall during the measurement period. For example, during this study period, plots received approximately 65.3 cm of rain during May through August and a total of 137.1 cm during one year compared to 20.3 cm during May through August and a total of 79 cm for another year.

Several authors have reported that the performance of young conifers was either better or not significantly different between compacted and not compacted soil treatments (Conlin and van den Driessche 1996, Page-Dumroese and others 1998, Heninger and others 2002, Gomez and others 2002a). For example, soil compaction had little effect on height growth of 13-week-old laboratory-grown lodgepole pine (*P. contorta* Dougl. ex Loud.), with the tallest seedlings occurring at the greatest compaction rate (Conlin and van den Driessche 1996). In another study, Miller and others (1996) reported that average height for 8-year-old Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*), Sitka spruce (*Picea sitchensis* (Bong.) Carr.), and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), did not differ among soil compaction treatments. Western white pine (*P. monticola* Dougl. ex D. Don) seedlings were tallest after soil compaction (Page-Dumroese and others 1998).

Foil and Ralston (1967) reported that loosening the soil from its normal structure reduced the height of loblolly pine (*P. taeda* L.) on light textured soils, but stimulated it on clay soils. They concluded that loosening the light textured soil affected its aeration porosity reducing the amount of available water and nutrients to suboptimal levels. Archer and Smith (1972), Stone and Elioff (1998), and more recently Gomez and others (2002a) demonstrated this concept. The implication is that some level of compaction of sandy loam (medium textured) soils can increase soil moisture availability while maintaining adequate

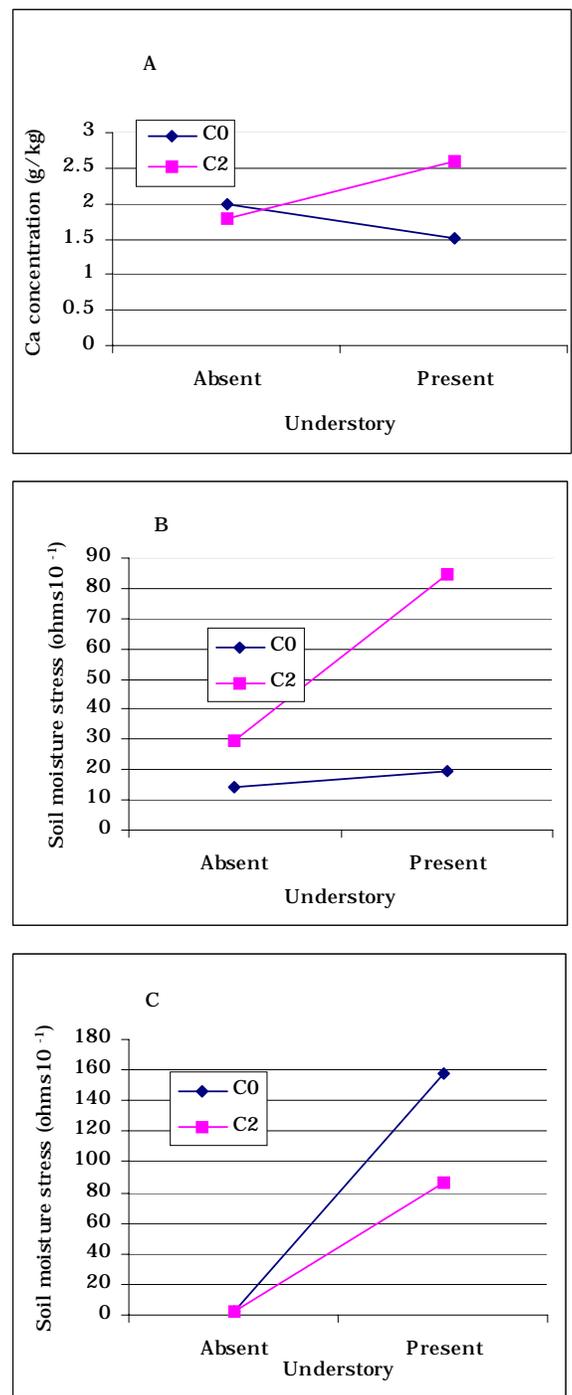


Figure 2.—Interactions between soil compaction (no compaction=C0 and severe compaction=C2) and understory treatments for (A) leaf Ca, (B) soil moisture stress in May, and (C) soil moisture stress in September.

Table 4.—The effect of soil compaction and understory control treatments effect on monthly soil moisture stress and temperature.

Treatment	Level	Month				
		May	June	July	August	September
Soil moisture stress (ohms 10 ⁻¹)						
Soil compaction	No	16.9	26.2	24.2	3.6	54.0
	Severe	52.6	30.4	25.1	6.1	37.7
	<i>p-value</i>	0.001	0.595	0.933	0.126	0.007
Understory	Absent	22.0	1.8	2.8	0.4	2.1
	Present	47.7	56.4	46.2	9.4	109.4
	<i>p-value</i>	0.001	0.001	0.001	0.001	0.001
Interaction	<i>p-value</i>	0.006	0.538	0.894	0.106	0.006
Soil temperature (°C)						
Soil compaction	No	15.1	12.1	14.1	14.3	3.5
	Severe	17.0	14.3	13.4	15.5	6.9
	<i>p-value</i>	0.105	0.191	0.781	0.047	0.344
Understory	Absent	13.1	9.8	10.8	11.6	3.3
	Present	18.5	16.3	16.1	17.8	8.5
	<i>p-value</i>	0.001	0.001	0.002	0.001	0.042
Interaction	<i>p-value</i>	0.873	0.287	0.187	0.350	0.262

aeration, and does not constitute a physiological problem to trees (Gomez and others 2002b). However, in the present study, soils are cherty silt loams and not sandy loams. But it seems probable that the cherty silt loam is behaving similar to compacted sandy loam. Voids between the fine-earth fraction (<0.002-m diam.) and abundant rock fragment surfaces are the dominant macropores in the chert silt loam soils (Sauer and Logsdon 2002). Upon compaction, soil micropores increase and soil macropores decrease, creating soil moisture characteristic of a finer textured soil. Nash (1963) maintained that soil moisture is the principal factor limiting growth of shortleaf pine, especially in Missouri. It grows best on soils with silt loam and fine sandy loam textures (Lawson 1992).

Additionally, shortleaf pine is apparently more adaptable to the physical changes in the soil environment on this site than the hardwoods that are also being studied (Ponder 2003). The hardwoods grew less in the compacted treatments (data not shown). It has been shown that top growth is less sensitive than root growth to soil disturbance in the early years (Singer 1981, Heilman 1981). Observations of root systems developed in heavy compacted soils show that lateral roots comprised a larger percentage of total root weight than was generally the case in less compact soils (Rodney and Ralston 1967). Work is being initiated in the Missouri LTSP study to compare tree root development between compaction treatments.

While soil moisture stress is an indicator of the soil moisture level, it may not be sensitive enough to effectively relate directly to plant physiological conditions. Soil moisture stress levels created by the two understory conditions were much farther apart but growth differences were significant only for d.b.h. growth (Table 1). This suggest that some of the growth advantage of shortleaf pine in compacted soil may be at a period of the growing season when soil moisture stress is more favorable in compacted soil and not limited to the time of the growing season when there are large differences in soil moisture stress attributed to understory treatments or compaction. Also, the more rapid growth on the compacted plots may have created the greater moisture stress found in May. Moisture stress was less on the compacted plots than on non-compacted plots in September after tree growth had nearly ceased because the compacted plots inherently held more moisture.

Leaf macronutrients were not significantly affected by soil compaction. Earlier, Gomez and others (2002b), using N¹⁵ and ponderosa pine, reported that they did not observe any significant effects of compaction on leaf N with total removal of the forest floor at two LTSP installations on loam and clay textured soils. These authors suggest that while leaf N differences attributed to soil compaction treatments were small, compaction is a significant factor associated with increased N recovery. Compaction may have prevented losses from the soil by reducing mineralization and increasing the probability of overall nutrient uptake in years to come.

The importance of understory on foliar N concentration was evident for both measurement periods. Weeds are hardy competitors for soil nutrients. Data collected over a number of years from loblolly pine for the LTSP in North Carolina show that foliar N was consistently higher for the understory absent treatment than for the understory present treatment (data not shown). Weeds reduce the amount of water and nutrients available to nearby trees, which are needed for growth. Less water implies reduced mineralization, which in turn influence the acquisition and the availability of other nutrients (Wilson 1994, Zabowski and others 1994).

Summary

The mechanism by which compacted soil in the present study supported better shortleaf pine growth than soil not compacted is not completely understood. Much of the better growth is likely due to better soil physical changes that caused better soil moisture conditions for growth during some part of the growing season. The better growth for trees in the compacted treatment is consistent with a LTSP study in California that has medium textured soils also. Controlling the understory increased leaf N and d.b.h. growth, but not height growth. Forest soils are complex with many components whose interactions can produce varied results. However, for this 8-year period, compaction has had a positive effect on shortleaf pine growth.

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HAY-SCENTED FERN (*DENNSTAEDTIA PUNCTILOBULA*) AND SUGAR MAPLE (*ACER SACCHARUM*) SEEDLING OCCURRENCE WITH VARYING SOIL ACIDITY IN PENNSYLVANIA

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Abstract.—Hay-scented fern is an invasive species that has increased in abundance over the last half of the twentieth century. Since this species may limit woody regeneration, it is important for forest managers to understand why its prevalence is increasing. Eighteen sugar maple stands across Pennsylvania were sampled to determine if soil chemistry and canopy density played a role in the abundance of hay-scented fern. It was found that percent canopy cover and soil O-horizon Ca/Al ratios were negatively related to percent fern cover. Percent canopy cover was related to soil A-horizon pH. In addition, soil A- and B-horizon pH levels were related to sugar maple seedling presence. These results indicate that soil chemistry should be considered as a factor controlling hay-scented fern abundance and sugar maple regeneration in Pennsylvania.

Hay-scented fern (*Dennstaedtia punctilobula* (Michx.) Moore) is a rhizomatous fern commonly found in deciduous forests, old fields and roadsides in its native range of the Eastern United States and Canada. This invasive species is capable of growing in a wide range of light and soil conditions (Conard 1908, Cody et al. 1977, Burnside 1988) and often develops into mono-dominant stands. Once established, hay-scented fern inhibits woody regeneration (Horsley and Marquis 1983, Bowersox and McCormick 1987, Kolb et al. 1990, Horsley 1993) by outcompeting woody species for light (Horsley 1986, Burnside 1988, Kolb et al. 1990, Horsley 1993, Hippensteel and Bowersox 1995) and rooting space (Hammen 1993, Lyon and Sharpe 1996).

Hay-scented fern has apparently increased in abundance over the last half of the 20th century. Several possible explanations for the dominance of hay-scented fern on forest floors have been proposed. White-tailed deer (*Odocoileus virginianus virginianus* (Boddaert)) selectively browse many forest understory species, such as sugar maple (*Acer saccharum* Marsh), red maple (*Acer rubrum* L.), white ash (*Fraxinus americana* L.), yellow poplar (*Liriodendron tulipifera* L.), and eastern hemlock (*Tsuga canadensis* L.). These species can be reduced in abundance or eliminated by deer browse (Tilghman 1989). Deer generally do not browse the less palatable hay-scented fern; thus hay-scented fern may be at an advantage over more palatable species where deer are abundant, and especially in canopy gaps (Horsley and Marquis 1983, Collins and Pickett 1988).

Another possible explanation for hay-scented fern spread is soil acidification caused by acid deposition. Pennsylvania soils have become increasingly acidic over the last half of the twentieth century (Robert 1994, Drohan and Sharpe 1997). Soil acidification results in base cation leaching, elevated aluminum concentrations, and an imbalance of Ca/Al in soil solutions and in the rhizosphere. This imbalance inhibits Ca and Mg uptake, reduces fine root growth, alters photosynthetic activity, and leads to nutrient imbalances in forest tree species (Ulrich 1983, Zoetl and Huettl 1986, Godbold et al. 1988, Shortle and Smith 1988, Wright et al. 1989, Joslin and Wolfe 1992). On forested sites with very acid soils, soil acidity/nutrient status may limit regeneration of acid sensitive species such as sugar maple and lead to increased abundance of more acid tolerant species of plants that appear to be more adaptable to an acidifying edaphic environment. The proliferation of hay-scented fern in extremely acid soils appears to indicate its adaptability to such conditions. The objective of this study was to determine possible relationships between hay-scented fern frond density, percent hay-scented fern cover, soil acidity and canopy density.

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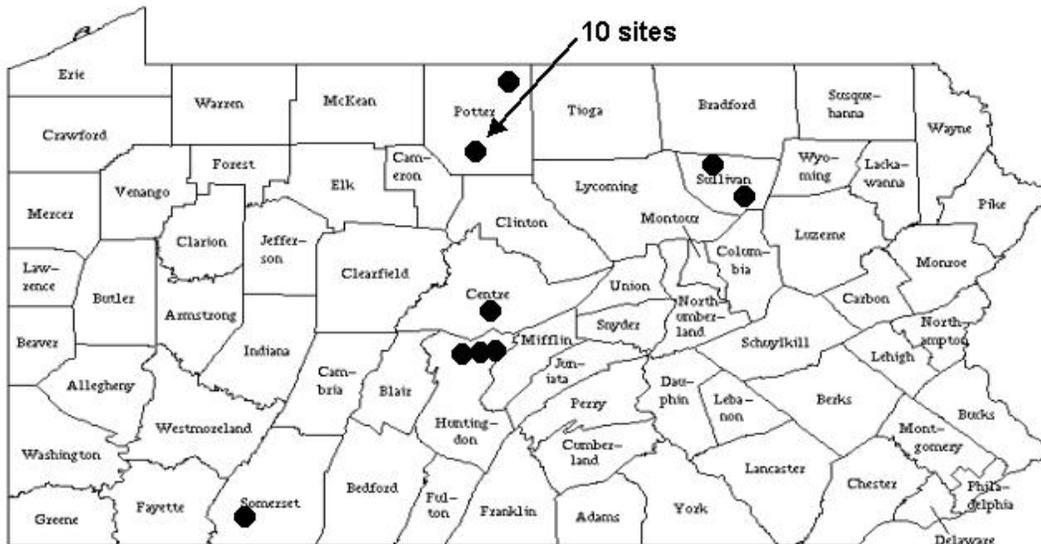


Figure 1.—General locations of the study sites.

Methods

Eighteen even-aged hardwood stands of predominantly sugar maple, located in Centre, Huntingdon, Potter, Somerset, and Sullivan Counties, Pennsylvania were selected for study (Figure 1). These stands were chosen to represent a gradient of soil acidity and Ca:Al values and the widest possible north-south distribution of sugar maple in Pennsylvania. Age of all stands was in the range of 80 to 90 years, and the stand size ranged from one to eight ha. Average dominant tree height for all stands (obtained through a subsample of at least five dominant trees per site) was 20.28 m and ranged from 13.20 m at site sixteen to 24.26 m at site five. Average dominant tree DBH for all stands (obtained from the same trees subsampled for height) was 36.25 cm and ranged from 25.03 cm for site sixteen to 50.58 cm at site seventeen. The stands were located in a variety of topographic locations, including stream bottoms, mid-slopes, and ridge tops in both the Valley and Ridge and Allegheny plateaus physiographic provinces.

Each stand was sampled to determine hay-scented fern percent cover, canopy density, sugar maple regeneration and soil chemistry. Random sampling of fern cover and sugar maple seedling occurrence was accomplished by throwing a hula-hoop from the center of each stand in a randomly selected direction. The 0.47-m² area described by the hoop when it landed was established as a sampling plot. Three such plots were established in each stand. Canopy density was determined from each plot location using a spherical densiometer. Fern fronds were counted and sugar maple seedling presence was assessed within the plots, and fern cover was visually estimated. Soil samples were collected from each plot from a small hand-excavated soil pit (40 cm deep). This depth ensured inclusion of the O, A, and B-horizons. Soil samples were extracted with 0.01 M SrCl₂ as a surrogate for plant available Al and Ca in each of the soil horizons (O, A, and B), and the molar Ca/Al ratio was calculated for each of the three horizons (Joslin and Wolfe, 1989). In addition, pH was determined in 1:1 water paste of air-dried soil. Linear and logistic regression techniques were used to analyze the data obtained.

Results and Discussion

The percent cover of hay-scented fern was strongly inversely related to percent canopy density of overstory sugar maple ($p \leq .001$) and O-horizon Ca/Al ratio ($p \leq .004$) in a linear regression model with a non-linear transformation. This linear regression analysis showed that canopy density and O-horizon Ca/Al ratio accounted for 71 percent of the variation in hay-scented fern cover (percent canopy density accounted for 53 percent of the variation and O-horizon Ca/Al ratio accounted for the remaining 18 percent of the variation in fern cover). Percent fern cover was negatively correlated with

Table 1.—Sugar maple seedling presence (1=present, 0=absent) and mean percent hay-scented fern cover, percent canopy density, A-, B-, and O-horizon soil pH, and A-, B-, and O-horizon Ca/Al ratio at each study site (ND – no data).

Site No	Sugar Maple Seedling Presence	Mean Percent Fern Cover	Mean Percent Canopy Density	Mean A Horizon pH	Mean B Horizon pH	Mean O Horizon pH	Mean A Horizon Ca/Al	Mean B Horizon Ca/Al	Mean O Horizon Ca/Al
1	0	0.25	0.90	4.74	5.12	5.54	22.56	0.33	147.11
2	1	0.00	0.97	5.18	5.10	4.90	627.15	129.13	1070.45
3	1	0.00	0.96	4.75	4.95	4.85	171.49	0.74	262.47
4	0	0.00	0.94	6.61	ND	ND	ND	ND	ND
5	1	0.00	0.94	5.13	5.02	4.57	2.78	5.11	206.70
6	1	0.00	0.91	4.68	5.24	4.45	106.00	331.15	2387.69
7	0	0.47	0.92	3.84	4.58	3.56	8.74	0.69	415.74
8	1	0.52	0.94	4.26	4.52	4.10	1.69	0.59	86.63
9	1	0.40	0.93	3.91	4.46	3.86	5.87	1.47	119.48
10	1	0.07	0.92	3.70	4.51	4.11	4.23	0.70	41.08
11	0	0.33	0.93	3.82	4.60	3.93	25.98	3.88	55.90
12	0	0.42	0.86	3.99	4.80	3.91	15.70	0.54	37.43
13	0	0.13	0.93	3.58	4.36	4.09	1.96	0.22	53.84
14	0	0.53	0.78	4.12	4.32	3.54	3.02	0.22	55.20
15	0	0.90	0.67	4.10	4.65	4.02	2.10	0.62	29.44
16	0	0.27	0.72	4.43	4.75	3.94	10.43	0.49	158.51
17	0	0.98	0.69	4.12	4.80	4.51	0.64	0.18	47.67
18	1	0.09	0.92	4.64	5.66	4.01	23.87	5.67	336.41

percent canopy density of sugar maple, which was expected because more sunlight was likely to reach the forest floor with reduced canopy density. Percent fern cover was also negatively correlated with O-horizon Ca/Al molar ratio. Percent canopy density and O-horizon Ca/Al data appear in Table 1.

In addition to the relationship between fern cover and O-horizon Ca/Al ratio, hay-scented fern frond density was found to be negatively related to the Ca:Al ratio in the A and B horizons (Figs. 2 and 3, respectively). A positive ($p \leq .057$) relationship was also present for percent canopy cover and A-horizon pH.

Sugar maple seedling occurrence, in relation to soil parameters and hay-scented fern occurrence, was analyzed by logistic regression. Results indicated that sugar maple seedlings were sixteen times less likely to occur on plots with hay-scented fern. However, sugar maple seedlings were four times more likely to occur with every unit increase in A-horizon soil pH and over six times more likely with every unit increase in B-horizon soil pH, within the pH ranges encountered in this study (pH range of 3.16-6.98 in the A-horizon and 4.22-5.91 in the B-horizon). Sugar maple seedling occurrence and corresponding B-horizon Ca/Al ratios are plotted in Figure 4. B-horizon Ca/Al ratio explained 55 percent of the variation in sugar maple seedling numbers on the study plots (Fig. 4).

Higher hay-scented fern densities were associated with reduced canopy density (which was correlated with low soil A-horizon pH levels) and low O-horizon Ca/Al ratios in sugar maple stands across Pennsylvania. Hay-scented fern appeared to be a superior competitor in the high light and low Ca/Al environments of extremely acid soils. Once established, hay-scented fern competes with other species for light and rooting space, which may further limit woody regeneration. These results indicate that soil acidity correction by liming may be required to regenerate sugar maple stands on strongly to extremely acid soils. The liming study of Wilmot et al. (1996) appears to support this hypothesis.

The data indicate that there is a strong association between soil acidity and both sugar maple canopy density and hay-scented fern abundance. This result is counter to the widely held notion (Marquis and

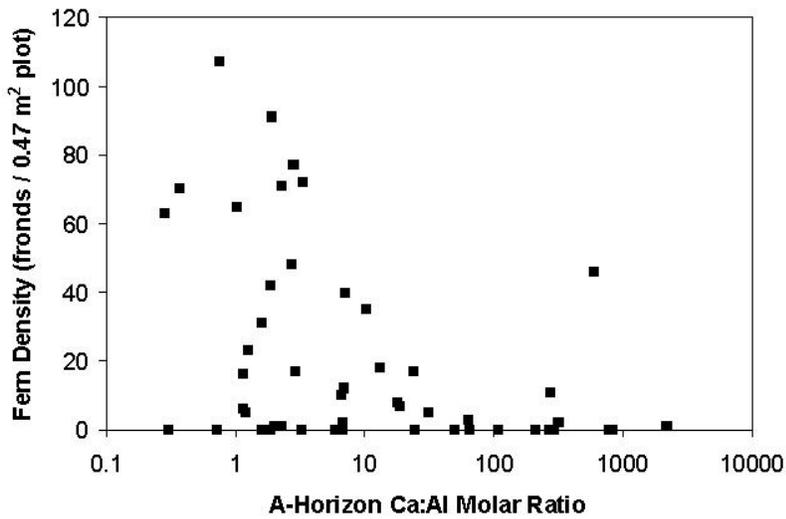


Figure 2.—The relationship between fern frond density and A-horizon Ca:Al molar ratio.

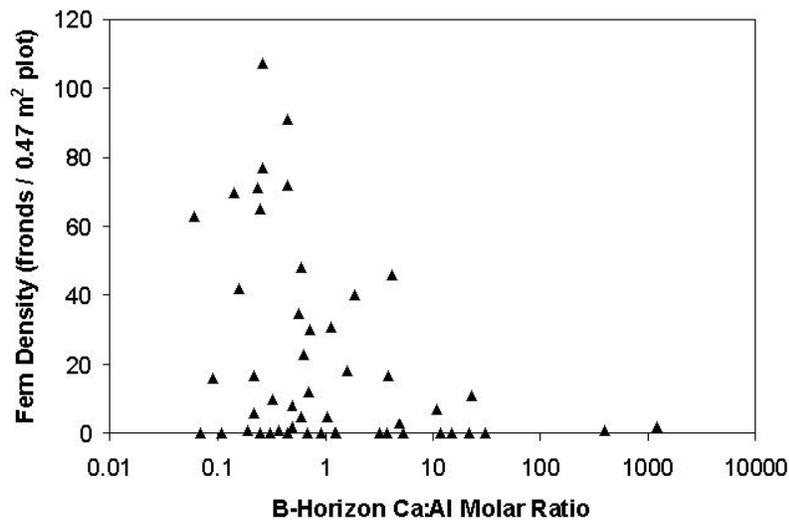


Figure 3.—The relationship between fern frond density and B-horizon Ca:Al molar ratio.

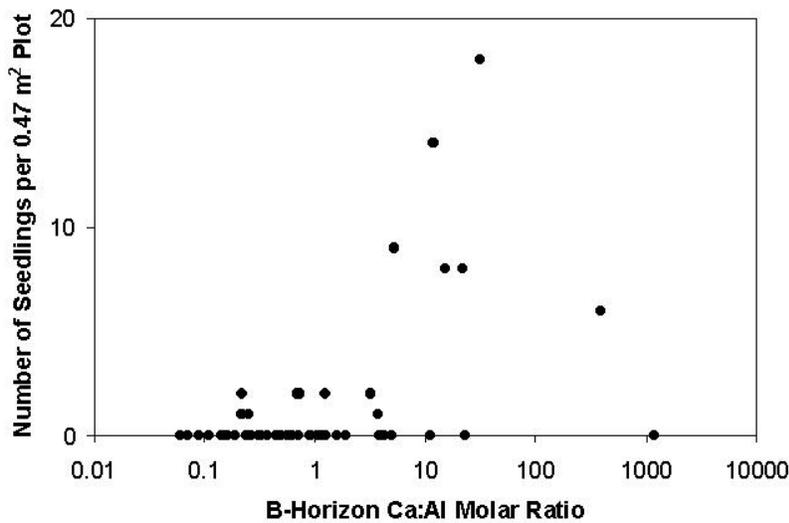


Figure 4.—Sugar maple seedling density versus B-horizon Ca:Al molar ratio.

Brenneman 1981, Horsley and Marquis 1983, Tilghman 1989) that hay-scented fern abundance is explained primarily by preferential browsing, which reduces competition from plants that are more palatable to white-tailed deer. It is also clear from the data presented that sugar maple seedling occurrence is associated with soil acidification on the sites that were studied. White-tailed deer numbers were not assessed in this study; however, the results represent data from 18 randomly selected sites across Pennsylvania, and we have no reason to believe background numbers of white-tailed deer differed in any more than a random way between sites. Further investigation into the relationship between hay-scented fern abundance, sugar maple regeneration, and edaphic conditions is needed.

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LANDOWNER PERSPECTIVES ON ATTRIBUTES OF THE WEST VIRGINIA FOREST STEWARDSHIP PROGRAM

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ABSTRACT.—We surveyed participants of the West Virginia Forest Stewardship Program to evaluate its status and effectiveness following a decade of program implementation. The questionnaire used permitted the respondent to answer freely: “what did you like most?” and “what did you like least?” about the Forest Stewardship Program during their participation in the program. With regard to the desirable attributes, 81% of survey respondents answered; only 65% filled in the blanks concerning the least desirable attributes. The most desirable attributes listed were: usefulness of the plan (23%), professional assistance and advice (20%), and educational aspects (19%). Respondents listed least desirable attributes as not enough follow-up (17%), not enough cost-sharing assistance (10%), and no dislikes (23%). While most comments were favorable, some program participants shared alternative perspectives.

Introduction

The West Virginia Forest Stewardship Program (FSP) has been in place for over 10 years. Five-year (Egan et al. 2001) and ten-year (Jennings 2003) assessments documenting the background of the participants and evaluation of the implementation rates of the recommended practices have been completed and have demonstrated the overall positive impact of this program administered by the West Virginia Division of Forestry (WVDOF). Likewise, high satisfaction ratings of FSP participants in states across the nation have been demonstrated (Esseks and Moulton 2000).

FSP is a federal program that provides cost-share dollars as an incentive for private forestland owners to formulate a management plan. Currently in West Virginia, woodland owners who enroll in the program pay 25% of the cost of having a licensed forester work with them and write an FSP plan; the remainder is paid with federal dollars under the administration of the State Forester, the Director of the WVDOF. This level of incentive has changed over the first decade of the program. During the early years of the program, some landowners were offered cost free plans.

Once a landowner enrolls in the program, the written management plan is the only link to the FSP, as other cost share dollars that might be accessed for plan implementation are available under different federal programs. Thus, the primary outcome of the FSP is to set the framework from which the private forestland owner can make decisions concerning the management of their properties.

Even though the FSP may receive high ratings for the service it provides, evaluations of various facets of the program are important to determine how well it is serving participants. As part of our 10-year assessment, we evaluated whether there were areas of the FSP that could be improved to further increase participants' satisfaction and enhance the service provided by the foresters and administrators of the FSP in West Virginia. To achieve this objective we asked open-ended questions to determine if there were areas in the FSP that should be highlighted and others that could be improved. This paper reports the results of participants' responses to questions about their most positive and negative perspectives of the FSP in West Virginia.

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Methods

A self-administered questionnaire-based survey was utilized to assess implementation rates of recommended practices following the initial 10 years of the FSP in West Virginia. The questionnaire paralleled the 5-year assessment of the program by Egan et al. (2001). Included in the questionnaire were questions dealing with demographics, satisfaction levels of all aspects of the program, motivation behind enrollment, implementation of recommended practices, and participation in other natural resource assistance programs. As part of the development of the questionnaire, West Virginia University Division of Forestry personnel, extension specialists, foresters from the USDA Forest Service and West Virginia Division of Forestry, and landowners from adjacent states made comments and suggestions on clarity and appropriateness of this survey instrument.

Our population included all private forestland owners listed in the WV FSP database as having been enrolled in the first 10 years of the program. At the time of our survey, 3,656 properties had received a management plan under the FSP. In order to avoid duplicate mailings, a single questionnaire was sent to all landowners who were listed more than once due to multiple properties enrolled in the FSP. Additional database filtering removed any participant who did not have an address listed leaving a mailing list that included 3,092 landowners.

Dillman's tailored design method was used for the data collection procedure (Dillman 2000). Questionnaire mailings were made in the following manner beginning in the first week of January 2003: a prequestionnaire announcement postcard, first questionnaire, reminder postcard, and second questionnaire. Addresses of respondents to the first questionnaire were removed from the mailing for the second questionnaire. Stamped return envelopes were sent with the questionnaires and FSP participants were asked whether or not they would like a short summary of the results; this was intended to provide an incentive to complete the questionnaire.

To evaluate how the WV FSP might better benefit the landowners enrolled in the program, we provided space for write-in responses. Two questions were directed questions, where the respondents had the opportunity to answer about: 1) what they liked most about the FSP, and 2) what they liked least about the FSP. A third opportunity for "free" expression was provided at the end of the questionnaire which included space to write in any additional comments they had about the program.

To facilitate analysis and presentation of the open-ended questions, we categorized these into groups of similar responses. One researcher preformed this categorization. Following the initial categorization of each of these three write-in areas, we again went back and reconfirmed that our categorization was stable and that there were minimal discrepancies in categorization. Some responses, particularly those of the general comments section, could have multiple categories. In categorizing these it was difficult to limit our subjectivity; therefore we classified these multiple types into the category that we felt was most strongly expressed by the respondent and this was by and large the first statement.

Results

The overall survey response rate was 63%, with 1,672 returned questionnaires from the total of 3,092 that were mailed. This response rate excludes the 436 mailings that were returned for insufficient or incorrect addresses. We examined nonresponse bias with respect to acreage and found no significant difference in average property size among respondents (200 ac) and nonrespondents (150 ac).

What Did You Like Most About The Program?

A total of 1,087 survey respondents answered the question concerning the most favorable attributes of the FSP. Four categories comprised 78% of the total responses for the most desirable attributes of the FSP (table 1). The plan itself was the top-ranking category and many respondents indicated that the plan gave a comprehensive perspective of their forested property. Program participants noted as a positive attribute the inventory portion of the plan, which provides a "detailed" list of the quantity of timber, the species present, and the location of different timber types on their properties. The set of

compiled maps included in the stewardship plans also were frequently listed as one of the most desirable attributes of the FSP.

Technical assistance ranked as the second most positive attribute of the FSP. Most respondents under this category listed as desirable attributes the opportunity to walk through their woods with a “professional” and to gain valuable insight into the value and variety of trees on the property, as well as being able to learn about potential management opportunities. In the words of one landowner, the “good plan made me aware of things I wasn’t originally aware of...”

The education/information category overlapped to a large extent with the first two categories, although we were not able to establish the direct source (the plan or foresters) of their educational experience. Landowners reported to have learned about specific opportunities such as how to create different wildlife habitat, timber stand improvement, and programs to support future management activities. They frequently mentioned the forester by name and cited as desirable “the personal attention to my needs from the forester.”

The fourth ranking category of “the best” of the FSP was what we have called “improving property”. In this category respondents cited reasons like “it provides goals and objectives to work towards” and the “opportunity to manage the land for the future” as the key benefits of the FSP.

All other categories totaled less than 10% of all responses. Despite this low percentage, cost sharing, tax incentives, and the on-the-ground practices implemented by the responding landowners were the single most important benefits of the program for their specific properties.

What Did You Like Least About the Program?

Fewer survey respondents (875) filled out this question regarding the least desirable attributes of the FSP than in the previous question (table 2). Moreover, nearly one in four (23%) who responded indicated there were no undesirable aspects that they could list.

Of those who did list undesirable attributes, 150 reported the fact that they did not receive any follow-up or contact with the forester or other administrators of the program following the initial plan or the initial visit by the forester who wrote the plan.

Table 2. —Number and percentages for responses to the question “What did you like least about the forest stewardship program.”

Category	Number	Percent
None	202	23
No follow-up	150	17
Other	86	10
Insufficient cost-share	85	10
Administrative barriers	68	8
Communication/Information	64	7
Inconsistent objectives	61	7
Inadequate plan	50	6
Unsatisfactory contacts	42	5
Management cost	26	3
Taxes	25	3
Management assistance	16	2
Total	875	100 ^a

^a Table entries do not add to 100 because of rounding error.

Table 1.—Number and percentages for responses to the question “What do you like most about the Forest Stewardship Program”.

Category	Number	Percent
Plan	249	23
Technical assistance	216	20
Education/Information	201	19
Improving property	187	17
Other	86	8
Cost sharing	66	6
Tax incentive	57	5
Implemented practices	25	2
Total	1087	100

All other categories each accounted for 10% or less of the total response for this question. Insufficient cost-share funding (10% response rate) was cited most frequently for either having insufficient funds (to help apply recommended management practices) or as having uncertainty with whether or not the funds would be available in a given year.

Importantly, 7% specified the most undesirable attribute as having objectives, either perceived or written, in the plans that were inconsistent with their own. Many expressed that the planwriters (or the

program in general) focused primarily on timber production and short-term economic gain. For example, one respondent reported the “emphasis on making the property into a timber producing business.”

Other categories, while not accounting for a large percentage of the total, reflected important concerns. Administrative barriers such as the amount of “paperwork”, time spent enrolling in the program, generic nature of some plans that appeared to be “boilerplate” plans, incorrect maps, and either too much or too little detail in the plans were included in this category. Management costs apart from the financial assistance available through FSP or other cost-share programs were listed by only 3% of the respondents.

Other Comments

At the end of the survey, respondents were asked, “Is there anything else that you would like to tell us about the Forest Stewardship Program?” The responses were extremely diverse among the 509 responses. Our “explanations” category (table 3) accounted for 23% of the total. In this category people expressed reasons why activities specified in plans had not been carried out. Many expressed the lack of time or other resources, or the fact that they do not live on the property.

Compliments were given to the program from 15% of those who filled out this section. These respondents felt strongly about the general “excellence” of the program or were more specific in giving credit to many of the WVDOP and consulting foresters whom they had worked with and associate with the program. Appreciation for the services rendered by these foresters was described as “excellent”, “outstanding”, or “very helpful”.

Several recommendations for improving the FSP were made and 68 respondents were included in this category. Most related to the comments in the “least desirable” section of the questionnaire, e.g., needing more financial cost-share, technical assistance, and plan follow-up.

Responses we labeled as “complaints” were found on 1 in 10 questionnaires. Complaints were often expressed as a matter of unfulfilled expectations such as the inability to receive any follow-up contact, or contact at all with respect to the implementation of the plan-writing phase. One expressed discomfort with the unannounced visit of the property by the foresters (“strangers”) writing the plan. Others were disappointed with the results of forest management activities—such as timber harvesting—subsequent to the plan-writing phase, but still associated with the program. Again, many were not satisfied with the quantity of financial and technical assistance available.

Discussion

Contrary to popular thoughts within the circle of natural resources professionals (e.g., Schmidt 2003), not all perceptions of foresters in our study were negative and suspicious. While some of the comments received concerned this issue, many more gave praise to the services rendered to respondents of this questionnaire.

As presented in this paper, the need for follow-up contacts, and additional technical and financial assistance are in demand by private forest owners. This corroborates with research by Magill and others (2004, in press) that of three types of assistance most frequently offered to forest landowners (financial,

Table 3. —Number and percentages by categories in general comments of FSP by questionnaire respondents.

Response	Number	Percent
Explanations	116	23
Compliments	77	15
Recommendations	68	13
Request assistance	60	12
Complaints	50	10
Philosophy	43	8
Activities	35	7
Change of ownership	16	3
No contact	12	2
Conflicting objectives	12	2
No plan/ Not in program	11	2
Waiting	9	2
Total	509	100 ^a

^a Table entries do not add to 100 because of rounding error.

technical, and educational), preference for direct technical assistance ranks highest. The opportunity to have a professional natural resources agent visit private forestland with the landowner is highly valued as suggested by the compliments paid to the WV FSP and by the recommendations given by the respondents of our questionnaire.

While most comments were favorable, many added insight into areas of the FSP that might be improved. Clearly, elicitation made by FSP participants claiming that the program, or specific foresters they worked with, did not take their objectives into account should be well heeded. Many studies have indicated that both forest owners and the general public—rural and urban—have opinions and objectives that diverge from those of natural resources professionals (Bliss et al. 1997; Egan et al. 1997). As Schmidt (2003) suggests, it is “once the [management] objectives have been settled on, the forester takes over.”

As with any program, the need for consistent information and service is crucial to the long-term stability of the program. Unfortunately, cost-share programs such as the Stewardship Incentives Program have been vulnerable to political and economic vagaries over the course of the FSP. Policies aimed at supporting management on private forests should maintain consistency in funding over long-term planning horizons. Given the program’s availability, most users taking advantage of the opportunity are pleased with the outcome; this can be beneficial to the long-term stability of FSP if conveyed to and understood by political decision-makers.

Acknowledgments

The authors are grateful for the administrative assistance of the West Virginia Division of Forestry and the funding made available through the West Virginia Forest Stewardship Program. Also helping with this manuscript were three anonymous reviewers who provided helpful comments and suggestions. The Director of the West Virginia Agricultural and Forestry Experiment Station approved publication of this manuscript; Scientific Article No. 2863.

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ESTABLISHMENT, GROWTH, AND IMPLEMENTATION OF THE FOREST STEWARDSHIP PROGRAM IN WEST VIRGINIA

Daniel J. Magill, David W. McGill, Shawn T. Grusbecky, and Brian M. Jennings[†]

ABSTRACT.—Over the past decade upwards of 3500 Forest Stewardship plans have been written in West Virginia, representing about 600,000 acres. We studied the history of the Forest Stewardship Program (FSP) in West Virginia to provide additional insight into management practices that have been recommended and those that have been implemented. Timber harvesting, timber stand improvement, and wildlife habitat improvement ranked consistently high among all recommendations in FSP plans. However, timber management was among the lowest of rankings for implemented practices. Private forest owner perceptions and reservations of post-harvest property conditions are likely a key factor affecting the implementation of this management activity. Apart from the high variation for industry foresters, no differences could be discerned between forester types for practice implementation rates on FSP properties.

The Forest Stewardship Program (FSP) in West Virginia provides a vital first link between a private woodland owner and a natural resource professional. Establishing this relationship is important, because more than 250,000 individuals privately own more than 90 percent of the 12 million forested acres in West Virginia. Since the inception of the FSP in West Virginia in 1990, more than 3500 landowners who control over 600,000 acres of woodlands in the state have enrolled in the program. This constitutes about 1.4 percent of all private forest owners in West Virginia and approximately 5 percent of the forested acreage.

West Virginia, like many other states, has a changing population of private forest owners caused by the both exurbanization—the population shift from urban to rural settings—and from parcelization (Egan and Luloff 2000, Jones et al. 2003, DeCoster 1998). Despite the problematic nature of these issues, the FSP has been a popular program with enrollment of acreage increasing significantly until 1997 when there was a drop off in participation. This decrease in enrollments, however, coincided with the termination of the Stewardship Incentive Program (SIP; Jennings et al. 2003).

One outcome of the WV FSP has been the increasing numbers of consulting foresters participating in the program. Over the first decade of the Stewardship Program (USDA Forest Service 2003), the number of consulting foresters attending the plan writers training sessions required by the West Virginia Division of Forestry (WVDOF) has increased dramatically to over 100 participating resource professionals. Currently, any forester who wants to write Stewardship plans in West Virginia is required to attend at least two of the training workshops each year. Involving private consultants with the program is one of the key components in increasing the number of woodland owners that participate in the Stewardship Program as they actively seek out private forest owners to enroll in the FSP.

Previous research has been conducted on implementation rates of forestry practices in the WV FSP (Egan 2000, Jennings 2003). In this paper, we investigate the role that various types of foresters play in the development of forest stewardship plans and some of the outcomes, or implementation, of the practices that they recommend in these plans.

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Methods

Information for the study was collected during the winter and spring of 2003. A seven-page mail survey questionnaire was used to collect the information. The sample population included all woodland owners who have a Forest Stewardship Management Plan in the state. Questionnaires were mailed to over 3500 landowners using multiple mailings based on the Dillman Method: pre-survey postcard, first mailing, follow-up postcard, and then a second mailing to increase landowner response rate (Dillman 2000).

Questions were asked about property size and ownership objectives, type of forester writing the plan, their time involved, the landowners satisfaction with the forester, and forestry practices recommended in the plan. Other questions included extent of recommendation implementation, landowner satisfaction, how did the landowner learn of the program and satisfaction with the program, extent of landowner participation with other forestry assistance programs (SIP, FIP, etc.), the extent of timber harvesting on the property, and landowner demographics. With respect to recommendations and implemented practices, respondents indicated the number and degree to which (strongly to somewhat) ten recommendations were written in their plans. These recommendations included: timber harvesting, timber stand improvement (TSI), wildlife habitat improvement, grapevine control, road construction, tree planting, water improvement, soil protection, visual quality, and recreation. In a subsequent question, respondents had the opportunity to identify which of these practices have been implemented and the extent of implementation (fully, almost complete, somewhat, none).

We used descriptive statistics and graphical methods to evaluate the degree to which management recommendations have been written into forest stewardship plans and the percentage of recommendations that have been implemented over the initial decade of the FSP. We combined the degree of recommendations and extent of implementation into two binary response variables: 1) recommended, not recommended and 2) implemented, not implemented. Recommendations and implementations were ranked to aid in result interpretations. Descriptive statistics and graphics were likewise used to describe the types of foresters who wrote FSP plans over the study period from 1990 to 2000.

Results and Discussion

Sample Response

The FSP database contained 3,092 relevant addresses. In total, 1,672 surveys were returned producing a response rate of 63 percent when the 436 “bad addresses” were eliminated from the population.

Description of Respondents

The average survey respondent reported owning 209 total acres for over 23 years of which 187 acres (89%) was enrolled in the FSP. This average acreage enrolled is slightly larger than that of the overall FSP population who average 171 acres per plan. Additionally, the average annual income for participants ranged from 45,000 to 60,000 dollars per year with an average age of 62 years old. More than 75% reported having some college or technical school training.

Foresters Writing Plans

With initially equal representation in writing stewardship plans, the proportion of private consultants grew sharply from the program's inception in 1990, and was higher than for WVDOF foresters and industry foresters for the remainder of the study period from 1990 to 2000 (fig. 1). The most dramatic increase in the number of private consultants developing plans occurred in 1993, the fourth year of the program in WV when private consultants wrote 68 percent of the plans for the year. During this first decade of the FSP in West Virginia, private consultants, WV DOF foresters, and private industry foresters have written 55, 34, and 11 percent of the forest stewardship plans, respectively. At the beginning of the FSP, you would expect WV DOF foresters to be most prevalent, since they would be the “first in the loop” during the onset of such a program. But as the program gathered, “steam”, more consulting and industry foresters became aware of the program and began to write FSP plans.

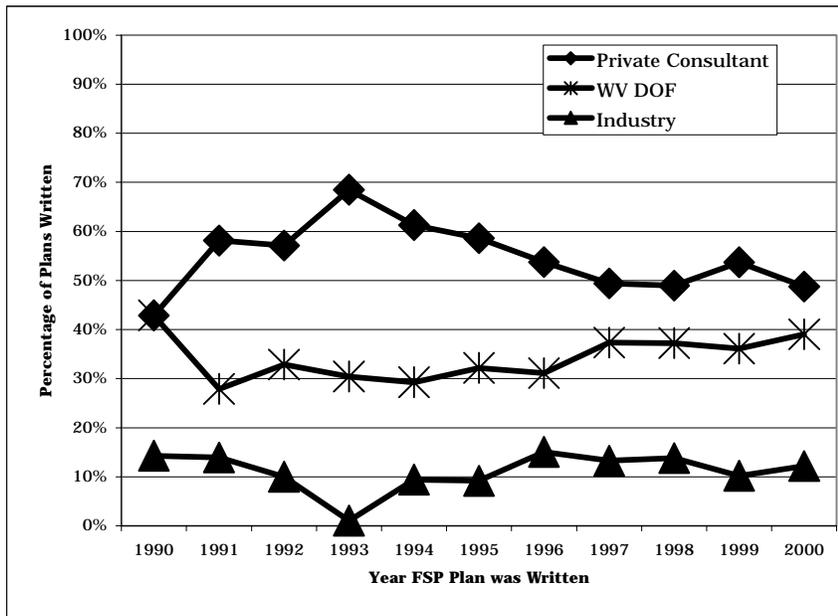


Figure 1.—Percentage of FSP plans by year written for three forester types: private consultants, service foresters of the West Virginia Division of Forestry, and industry.

The large proportion of plans written by private consultant foresters could be partly due to the training and opportunity provide by the WVDOF for this group of foresters to write plans. With the workload and time constraints placed on state service foresters, having private consultants available to write plans increases the opportunity for more landowners to obtain a FSP plan. We suggest that the indication in figure 1, that the number of plans written by private consultants is steadily declining could be due to the fluctuating number of plans written each year and the availability of funding from year to year.

Foresters writing management plans compares closely with similar research in Alabama (Zhang 1996). Over the ten-year period between 1986 and 1995, consulting foresters wrote 48 percent of Alabama’s forest management plans, followed by public assistance foresters (29%) and industry foresters (23%). The larger proportion of plans written by industry foresters in Alabama is probably because 20 percent of the land in that state is owned by forest industry. In contrast forest industry only owns about 7 percent of the land base in West Virginia (Griffith and Widmann 2003). Another reason could be that most of the industry land in Alabama is used to grow short rotation southern yellow pine, and not hardwoods as in West Virginia. The lower percentage of plans written by industry foresters in West Virginia is also likely due to the fact that several large companies that work with private forest owners have their own form of plan they produce. Hence these industry foresters do not necessarily need to have the cost-share dollars available through the FSP (our target population).

Forest Management Practices

Guidance to private forest owners on the part of the natural resources professionals that enroll in the FSP in West Virginia comes in large part through the recommendations discussed during the plan-writing process. These are incorporated into management plans for the owner to use in planning and implementing management activities. Of the ten main recommendations (above) listed in FSP plans, timber harvesting, TSI, and wildlife habitat improvement ranked above all others as the most frequently occurring in management plans (fig. 2). These rankings were relatively consistent in the first decade of the program. The least common recommendations were tree planting, water improvement, and recreation. Despite these low ranks and high variability by year, even water improvement, for example, was found in 57 to 75 percent of the stewardship plans. Hence, survey respondents indicated that a wide variety of recommendations were included in their plans.

Discussion and results of implemented practices is derived from the same practices recommended in plans and already discussed in the previous section of the paper. Management activities that were

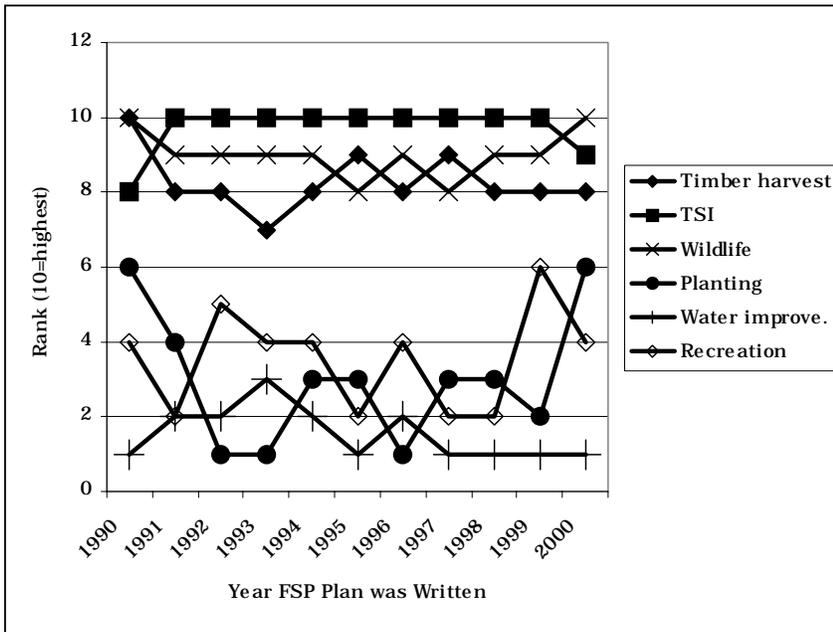


Figure 2.—Ranks of the three most frequent and three least frequent recommendations written into FSP management plans over the first ten years of the program in West Virginia. Practices that had intermediate rankings are not shown.

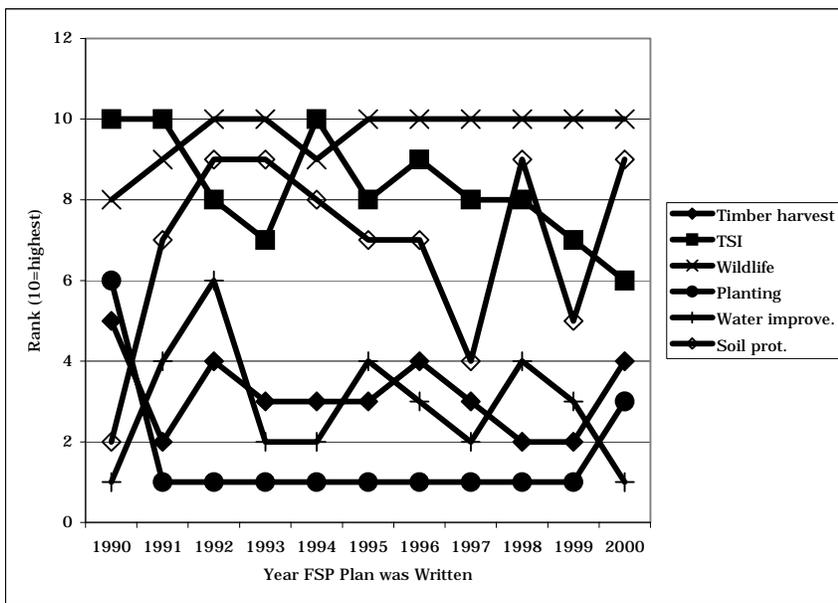


Figure 3.—Ranks of the three most frequent and three least frequent implemented practices occurring on FSP management properties over the first ten years of the program in West Virginia. Practices that had intermediate rankings are not shown.

implemented over the period showed a wider variation among rankings of specific practices recommended (fig. 3). With respect to figure 3, the relationship between implementation and the year a plan was written describes the rate a practice was implemented in a plan written in a specific year, which usually would be the same year an activity is recommended. Rankings for the implementation of management activities contrasted significantly with those seen for recommendations in the plans.

While timber harvesting was one of the most frequently cited recommendations, it ranked as one of the least frequently implemented practices. Why aren't private forest owners harvesting timber at a rate consistent with the proportion of plans in which recommendations for this practice exist? Despite the presence of these recommendations, landowners' perceptions of timber harvesting are diverse. Egan et al. (1997) have demonstrated that even among Tree Farmers (those with properties enrolled in the American Tree Farm System) there are differences in perceptions of the outcomes of timber harvesting.

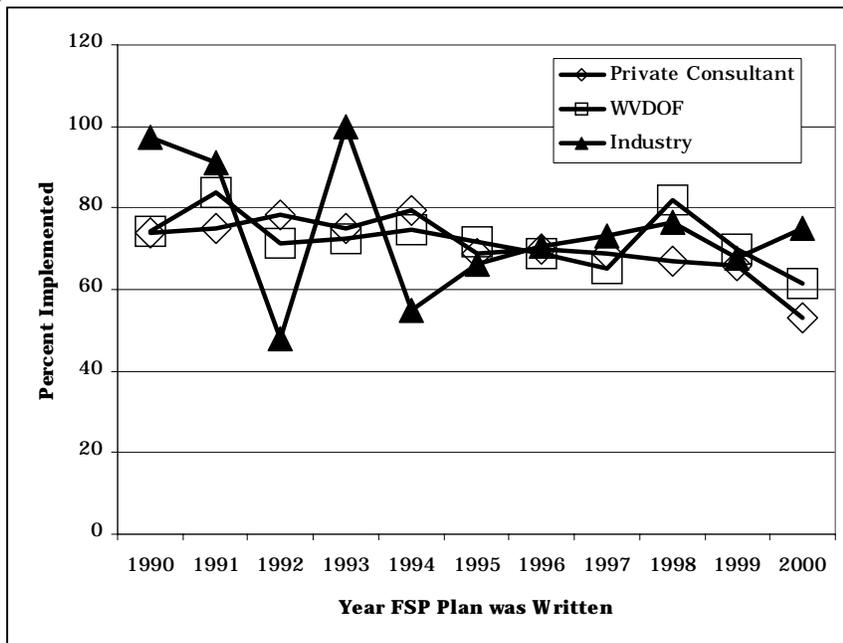


Figure 4.—Average percentage of forest management practices implemented on FSP properties during the first ten years of the program in West Virginia. Averages are separated by type of forester that wrote the FSP management plan.

Translating this to our population of FSP participants, we expect that there are similar reservations with post-harvest conditions of the respective properties. We also suggest that other reasons for low implementation of timber harvest recommendations could include lag times for improved growth, size, and value as well as for a delayed income source. Other reasons for low implementation of other practices could include schedule lag times (e.g. waiting for a required plant growth stage or property condition) lack of cost-share assistance for practices, landowner physical disability, or even forester follow up.

TSI was initially one of the higher ranking practices applied on FSP properties, but with time it decreased in rank. By 2000, the practice ranked 6th out of the ten practices. This is likely due to the limited time that landowners have between the time the plan was written until the time we conducted the survey (Jennings and McGill 2003). Typically, it may take a landowner at least a year to begin TSI work, especially if it has been recommended on a large portion of the property. One practice that showed high variation and ranked among the highest implemented practices was soil protection, which was also among the most recommended, but not in the top three ranking. It is not possible from our survey results to establish the on-the-ground treatments that were implemented to protect certain areas of the respondents' properties, although it is likely related to other activities that were implemented like road construction and the simultaneous implementation of seeding and other erosion control measures.

No clear differences among forester types were indicated in terms of the likelihood of someone having implemented a practice (fig. 4). Apart from the wide variation in implementation rates for industry foresters—due to the lower number of plans written by this group—implementation rates track evenly with one another over the ten-year period.

Conclusions

Implementation rates of forestry practices recommended to private forest owners participating in the FSP have been shown to be high in several studies in West Virginia and nationwide (Egan et al. 2000, Jennings 2003, Esseks and Moulton 2000). This evaluation of recommended and implemented practices through time corroborates studies that suggest that landowners objectives and interests are wide ranging (Fraser and Magill 2000) and that their priorities may depart from those of natural resources professionals (Kluender and Walkingstick 2000, Egan et al.1997).

While traditional forest management recommendations like timber harvesting, timber stand improvement (TSI), and wildlife habitat improvement ranked consistently high among all recommendations, wildlife management maintained the most consistent high ranking between 1990 and 2000. Even though implementation of stand improvements fluctuated over the study period, it still remained highly ranked in being conducted. The fluctuation in implementation rate for TSI can be related to lag times incurred as a result of activity scheduling in a plan. For example, the landowner was recommended to first install access into the management area before conducting TSI. Timber harvest activities were among the lowest ranked for implemented practices, which again could be associated with lag times. These contrasts point to the continued need to seek out ways to engage landowners in ways that promote both their own objectives and those that meet societies demands for wood, clean water, and the other amenities derived from the private forests of West Virginia.

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IDENTIFYING PRIORITY FOREST MANAGEMENT ISSUES IN WEST VIRGINIA—A SURVEY OF STATE SERVICE FORESTERS

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ABSTRACT.— West Virginia Division of Forestry (WVD OF) State Service foresters have a wide range of responsibility. We surveyed this group of foresters to illuminate the critical issues facing them and the WVD OF. Forty-eight of the sixty-six questionnaires were returned (73% response). Within the six issues categories in the questionnaire, top concerns listed by WVD OF foresters were: harvesting in poor weather conditions, harvesting with little regard for desired future conditions, prohibitive workers compensation rates, lack of landowner education, negative publicity from uninformed sources, and number of landowners implementing stewardship practices.

Introduction

West Virginia Division of Forestry (WVD OF) State service foresters have a wide range of responsibility—assisting at fire incidents, checking compliance of timber harvesting operations with best management practices and the Logging Sediment Control Act, and providing technical assistance on private forests. In this latter function, during the past decade, WVD OF Service Foresters were responsible for writing over a third of the forest management plans under the Forest Stewardship Program (FSP; Egan et al. 2001, Jennings 2003). Moreover, the WVD OF oversees all FSP activities at the state level. Along with these duties, some state service foresters coordinate the management of the eight state forests and coordinate special fire and watershed education programs.

Service foresters are state employees of the WVD OF and are closely linked with the communities they serve. They fit a niche that has responsibilities to answer to politicians, private landowners, advocacy groups, other foresters, and forest products industry. In these roles, the state service foresters have a unique opportunity to work with diverse clients, from loggers to private landowners and develop a unique understanding of the issues facing the forestry sector in the state.

Over the past decade, big changes have occurred in the forestry sector in West Virginia. New issues have emerged as well in the realm of forest management and along with new laws like the Logging Sediment Control Act of 1992, and have brought additional responsibilities and work loads to the state's service foresters. Several large mills have been established in the state during this period as well, adding to the demand for timber from private forests. In addition, several new transportation corridors have been opened, or are in the process of opening, increasing the "exurbanization" and "greening" of rural communities that has been shown to lead to changes in conservation perceptions in these communities (Egan and Luloff 2000, Johnson and Beale 1998, Jones et al. 2003).

Because of these recent changes in the forestry sector and the potential changes this could bring to the state with respect to forest resources management, we initiated a study to compile the opinions of our state service foresters on priority management issues. Our objectives included determining what WVD OF foresters view as the most relevant forestry issues confronting the forest sector in West Virginia today.

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Methods

In summer 2001, a two-part mail survey was conducted (Westfall 2001). The sample frame consisted of West Virginia Division of Forestry service foresters at the county, district, and state headquarters levels. Names and addresses were obtained from the 2001 WVDOF employee list. The employee list was checked with individual district foresters to assure its accuracy.

First, to “flesh out” issues facing the forestry sector in West Virginia, a letter was mailed to all county and district level service foresters (n=56) asking each of them to list five major problems facing the forest sector in West Virginia. The response rate was 32%.

Based on these initial responses, a questionnaire was developed with 48 problem statements. Respondents, this step including state level foresters and State Forest foresters (n=66), were asked to rank the level of severity of these problems on a scale of 1 (no problem) to 4 (severe problem) following recommendations by Tuckman (1999). A second letter with a questionnaire was sent to all nonrespondents three weeks after the first. Forty-eight of the sixty-six questionnaires were returned (73% response).

Survey questions reported in this paper concern six resource management categories:

- 1) Timber harvesting,
- 2) Forest management and planning,
- 3) Forest policy, legislation, and regulation,
- 4) Continuing and outreach education,
- 5) Public relations, and the
- 6) Forest Stewardship Program.

For comparing the magnitude of concern in discussion for various issues, we viewed average ranked responses with “scores” of greater than 3.0 as “of greatest concern” and those less than 2.0 as “of least concern”. We considered average scores greater than 3.5 as crucial issues.

Results and Discussion

The top seated issues with all foresters among the six, topic categories were prohibitive worker’s compensation rates (mean score=3.7), harvesting with little regard for desired future condition of the stand being harvested (3.6), negative publicity for forestry from uninformed sources (3.6), and the overuse of diameter limit cutting (3.5; table 1a and 1b). Eight other issues had scores that averaged between 3 and 3.5, indicating the importance of these as issues deserving attention of resource management agencies.

At least one issue from each of the six categories had average scores of 3 or higher. The forest management and planning category, consisting of issues involving various parts of silvicultural prescriptions and harvest planning, had the greatest number of issues with averages greater than 3.0. Despite the requirements of the West Virginia Logging Sediment Control Act which requires licensed and certified timber harvesting operators and the notification of intent to harvest timber (WVDOF 2002), according to WVDOF service foresters preharvest planning for silvicultural and road engineering purposes is lacking. It is likely that with increased attention to water quality from nonpoint sources, that these concerns will increasingly surface in both natural resources and political arenas.

Importantly, some issues were of very low concern to the WVDOF foresters, although none of the issues had average scores below 1.5. The lowest average scores were generally found in the Forest Stewardship Program (FSP) category where four issues resulted in average scores less than 2.0. Low scores for availability of foresters to write stewardship plans, access to information concerning the FSP, the quality of plan writer training sessions, and the number of training session required to permit a forester in qualifying as an FSP planwriter suggest that these issues are adequately managed in the eyes

Table 1a.—Statements and ratings from West Virginia Division of Forestry foresters. Forest management issue statements were generated from an initial questionnaire, and then put before the service foresters for rating. Ratings are based on a scale of 1 to 4 (1= *not a problem*, 4= *severe problem*).

Statements	Mean
<i>Timber harvesting</i>	
Operating when weather conditions are poor.	3.2
Increased size of equipment leads to more site damage	3.0
Small independent loggers going out of business	2.8
Damage to county roads by logging trucks	2.5
Shortfall of trained sawyers and equipment operators.	2.4
Timber regulation should be done by independent agency	1.7
Private industry sector is tied too closely to the WVDOF	1.7
<i>Forest Management and Planning</i>	
Harvesting with little regard for desired future conditions	3.6
Overuse of diameter-limit cutting	3.5
Lack of silvicultural prescriptions on private property.	3.4
Need for planned road system prior to harvesting	3.2
Need for preharvest regeneration assessment	3.0
Need for comprehensive preharvest management plan	2.7
Need for more clearcutting and less partial cutting.	2.4
Need to reevaluate minimum product size.	2.2
Need for protecting wildlife and their habitat.	1.9
<i>Forest Policy/Legislation/Regulations</i>	
Prohibitive workers compensation rates	3.7
Lack of penalties in BMP and LSCA violations	3.2
Lack of enforcement powers WVDOF	3.2
Non-payment of severance taxes and high estate taxes	2.8
Foresters enforcing LSCA laws differently	2.7
Timber industry does not appear fully committed to LSCA.	2.7
Companies buying/selling timber from unlicensed loggers.	2.6
Loggers are not paying workers compensation rates	2.6
Increase in timber trespass complaints.	2.6
Need for clearer BMP standards	2.3

of the WVDOF foresters. Indeed, we have heard many complimentary remarks from FSP administrators and from the FSP landowners concerning the high quality of the program in West Virginia (Jennings 2003).

Foresters reflecting on the state of the forest products sector in West Virginia simultaneously emphasized concern for future site productivity, as indicated by their emphasis on “desired future conditions”, and for the ability of timber harvesting firms to be able to function from a business standpoint. Workers compensation rates were identified to be a crucial issue, but whether this was out of interest of the inability of the “small, independent logger” to compete in the market or just a general awareness of the problem as an industry, it is difficult to determine. Currently, logging firms pay about 48 cents on the dollar for this insurance premium.

Some of the concerns expressed by the WVDOF foresters are being addressed by the forest sector in the form of best management practices, although currently the state’s Best Management Practice handbook focuses exclusively on methods for controlling sedimentation and erosion, and nothing on silvicultural opportunities (WVDOF 2002).

This survey corroborates the work of Fajvan and others (1997) showing very little evidence of silvicultural practices applied to the majority of the 100 recently harvested stands they sampled. Then, only clearcuts could be identified as having a specified silvicultural practice. Most of the stands they

Table 1b.—Statements and ratings from West Virginia Division of Forestry foresters. Forest management issue statements were generated from an initial questionnaire, and then put before the service foresters for rating. Ratings are based on a scale of 1 to 4 (1= *not a problem*, 4= *severe problem*).

Statements	Mean
<i>Continuing and Outreach Education</i>	
Lack of landowner education	3.32
WVU and WVDOF lack outreach programming	2.79
Need for industry sponsored outreach education	2.70
<i>Public Relations</i>	
Negative publicity for forestry from uninformed sources	3.57
Need to enhance image of loggers	3.30
Timber companies shifting responsibility to contract loggers	2.91
Timber industry not taking responsibility for logging practices	2.89
Lack of willingness by corporate forest products firms to dialogue with groups opposed to logging and good forestry practices.	2.55
<i>Forest Stewardship Program</i>	
Number of landowners implementing FSP recommendations.	3.36
Shortfall in personnel to check forestry practice implementation.	2.77
Lack of private landowner demand for plans.	2.21
Statewide opportunities for landowners with plans to learn more about the program.	2.21
Adequate funding to write plans.	2.02
Availability of foresters to write plans.	1.83
Access to information regarding the FSP.	1.70
Quality of plan writer training sessions.	1.66
Number of required plan writing training sessions.	1.66

sampled were cut using a diameter limit protocol, a protocol that does not necessarily evaluate the spatial distribution of residual stems nor the regeneration potential of the stand slated for harvest. WVDOF foresters reflect the urgency of this issue by indicating there is little regard for desired future condition of harvested stands and that there are “too many” diameter limit cuts.

WVDOF foresters manage private forests. These are the most unregulated of all forestlands in the state and make up the largest component of forestland in the state. In contrast, large industrial forests are typically under third party certification programs or self certifying programs like the Sustainable Forestry Initiative (SFI 2003). These programs bring attention to issues like sedimentation and erosion to the forefront of the attention of the industrial field foresters that manage these properties. However, on small, private forests, landowner contact with professional foresters is rare (Fraser and Magill 2000) and their management preferences often misunderstood (Bliss and others 1994, Jones and others 1995). Concern by the WVDOF foresters in this study for lack of landowner education points to the need for more attention in this area.

These responses from the WVDOF service foresters were made in a time period of increasing timber harvesting activity, stricter requirements to follow BMPs, and greater attention on compliance with the WV LSCA. Moreover, several recent years have had abundant fires, which further burdens available WVDOF resources. But because of the similarity of viewpoints held by the majority of administrative levels, the primary aims and operational direction of the agency will likely to focus resources (when they become available) towards these critical issues.

Acknowledgments

The authors wish to thank two anonymous reviewers who provided helpful comments and suggestions. The Director of the West Virginia Agricultural and Forestry Experiment Station approved publication of this manuscript; Scientific Article No. 2864.

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**DIVERSITY OF INCENTIVES FOR PRIVATE FOREST LANDOWNERS:
AN ASSESSMENT OF PROGRAMS IN INDIANA, USA**

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ABSTRACT.— Many programs provide incentives for non-industrial private forest (NIPF) owners. Some are initiated by government, while others are organized by private interest groups. Due to the complexity of this web of programs, it is not clear what the incentives of the programs really are. We focus on four specific programs that involve slightly different actors and represent different rule structures - a federal cost-share program, a state tax incentive program, a nationwide private stewardship program, and a local private conservation organization. We perform institutional analysis of the formal and informal rules of the programs based on literature review, discussions with officers and formal guidelines of the programs. This provides us the opportunity to classify different types of rule structures, and explain them in relation to the differences in goals and organizational structures of the programs.

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WILL LANDOWNERS ADOPT REFORESTATION PRACTICES? ANSWERS FROM THE MISSOURI RIVER FLOOD PLAIN

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ABSTRACT.—Reforestation of ecologically sensitive flood plain lands will depend greatly on private landowners. This paper develops several competing models that can help public agencies to predict landowner adoption of reforestation cost-share programs. Akaike's Information Criteria (AIC) is used to rank the models, based on data from a mail survey of flood plain landowners along the Missouri River. Results show that landowners who already have forested land, have already interacted with forestry agencies and are of middle age and/or family tenure status are the most likely to adopt. Such landowners may enroll up to 13 percent of Missouri River flood plain land. The challenge for forestry agencies will be to make that first contact with or plant that first tree on the land of the other, less likely landowners.

The bottomland forests in the Missouri River valley, along with its tributaries, are some of the richest and most diverse ecosystems in the world. These flood plains provide a treasure of economic and ecological values. Some of these values include mitigating the erosive nature of stream channel dynamics, improving water quality, protecting levees and other structural improvements, production of forest products, moderation of storm flow events, travel lanes for wildlife, and aquatic habitat [Malanson, 1995].

Since settlement by Europeans and pioneering Midwesterners, the extent of bottomland forests has been greatly reduced. The riparian forest corridor with its network of tributaries was severely fragmented as these forests were cleared for agricultural production and impacted by large flood control projects [Brinson et al., 1981; Turner et al., 1981; Malanson, 1995].

One problem in these flood plain forests is the recurring one of sustaining mature oaks or securing adequate regeneration in the understory. In both the public and private sectors, there is a growing interest in improving the understanding of riparian forest ecosystems and in developing management techniques that ensure the sustainability of this important resource. The U.S. Department of Agriculture's continuous Conservation Reserve Enhanced Program [Missouri Conservation Reserve Enhancement Program, 2000] focuses on riparian buffers. Riparian buffers have substantial economic value in reducing agricultural non-point source pollution in a Missouri Watershed [Qiu and Prato, 1998].

One key to reforestation of the flood plain is the private landowner who owns and farms the lands adjacent to the rivers and riparian areas. This land is particularly suited to an agroforestry system or practice. In Missouri private landowners own over 90 percent of the flood plain along the Missouri River [Missouri Resource Assessment Partnership, 2001]. At this time, there is much we do not know about their goals for ownership of this land, nor do we know whether they would be interested in adopting land management practices that incorporate trees in flood plain areas. Recent research has shown that farmers will elect to plant a buffer unless the net crop price is high or the land rental rate is low. The choice of buffer type, trees or grass, is affected by crop price, farm size, relative incentive payments, relative cost share rates, and amount of deer damage [Lynch and Brown, 2000]. The economics of restoring private lands to forests can only continue to gain in importance. Landowners can derive periodic income from timber production. In addition, annual income from hunting leases or carbon credits may also be available to some landowners [Stanturf et al., 2000].

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To improve our understanding of the factors that will influence landowners and managers to adopt reforestation practices in flood plain areas and to better understand landowners' knowledge, motivation and behavior, we developed a survey directed to flood plain landowners. Our goal was to develop and evaluate models that can help public land management agencies and others to predict which landowners will adopt reforestation practices and at what cost-share levels. This information will help these agencies to develop and target programs and determine what these programs will cost and how much land they will influence.

Methods

A "Behavior Survey" was designed, tested and administered to a group of Missouri River flood plain landowners. This survey included questions about how respondents currently manage their land, what forest management practices (if any) they employ, whether they have a management plan and, if so who helped them develop it, and whether at various cost-share levels they would enroll in a flood plain reforestation program. In addition there were questions about long- and short-term goals and opinions of the forestry services received from different government agencies. There were also a set of demographic questions on age, gender and income. This set also included questions on land tenancy and land characteristics [Treiman and Dwyer, 2002].

Flood plain land outside the levee in thirteen counties bordering the Missouri River was chosen as the study area, representing 49 percent of the Missouri River's flood plain in Missouri [Missouri Department of Conservation, 2001]. Using available GIS coverages [Missouri Resource Assessment Partnership, 2001] we estimated the overall ratio of flood plain land to land owned. A complete list of flood plain landowners, private, corporate and public, in these 13 counties was developed from courthouse records and from aerial photographs. The list included the names of six-hundred and thirty-three names of private landowners. Of these, five-hundred twenty qualified as current landowners with land outside the levee. About one-half of these, 260, received the "Behavior Survey" and the other half received a separate "Knowledge Survey", designed to address other issues not addressed in this paper [Treiman and Dwyer, 2002]. The survey was administered using techniques and methods described by Dillman [2000]. In this paper, we shall consider the results of the "Behavior Survey".

We developed a set of *a priori* logistic models that might explain a landowner's decision to enroll in a hypothetical flood plain reforestation cost-share program. These models include demographic, knowledge and current behavior variables. (See Table 1.) The *a priori* models were developed to help design a future cost-share program and to predict who would enroll and how many acres would be enrolled. The resource management agency can also use the results to help target the program either towards the most likely enrollees or to try to help turn unlikely landowners into likely ones.

The variables of interest were those that might be easily identifiable by, or particularly useful to, program managers and planners. These included knowledge of forestry/land management and behavior: such as having a forest management plan, managing for wildlife or timber, or having trees on their land. Also included were a set of demographic variables: age, gender, income, tenancy and size of ownership that may help program managers target their work. Program managers and planners, as well as county-level state forestry agents have access to much of this data in Missouri, whether from county courthouse land ownership records and maps, lists of state forestry agency landowner contacts, records of other cost-share programs and so on.

We began with a demographic model, which included only age, gender and income, and then developed further *a priori* models to test whether the more forestry-specific parameters could help decision-makers design a better reforestation program. Age and gender might affect long-term commitment to the land [Olmstead and McCurdy, 1989]. Both Ervin and Ervin [1982] and Cooper and Keim [1996] have found that "orientation to farming" and experience affect policy adoption. Income, or the size of their ownership, might affect the landowner's ability to meet the landowner's part of the cost-share [Ervin and Ervin, 1982; Olmstead and McCurdy, 1989]. Length of tenancy and

Table 1.—A priori models with their underlying hypotheses and notation. These models were developed to explain and predict landowners’ adoption of the proposed reforestation cost-share programs. The models were designed to provide potentially useful input to decision makers considering and designing such programs.

Hypothesis	Model Structure
1. Demographic Model: Age, Gender, and Income	$\gamma(x) = \beta_0 + \beta_1A + \beta_2G + \beta_3I$
2. Forest Plan and Demographic	$\gamma(x) = \beta_0 + \beta_1F + \beta_2A + \beta_3G + \beta_4I$
3. Having Trees and Demographic	$\gamma(x) = \beta_0 + \beta_1H + \beta_2A + \beta_3G + \beta_4I$
4. Current Management and Demographic	$\gamma(x) = \beta_0 + \beta_1C_t + \beta_2C_w + \beta_3A + \beta_4G + \beta_5I$
5. Length of Tenancy, Having Trees and Demographic	$\gamma(x) = \beta_0 + \beta_1T_1 + \beta_2T_2 + \beta_3T_3 + \beta_4T_4 + \beta_5H + \beta_6A + \beta_7G + \beta_8I$
6. Length of Tenancy, Having Trees, Forest Plan and Demographic	$\gamma(x) = \beta_0 + \beta_1T_1 + \beta_2T_2 + \beta_3T_3 + \beta_4T_4 + \beta_5H + \beta_6F + \beta_7A + \beta_8G + \beta_9I$
7. Length of Tenancy, Having Trees, Forest Plan, Size of Ownership and Demographic	$\gamma(x) = \beta_0 + \beta_1T_1 + \beta_2T_2 + \beta_3T_3 + \beta_4T_4 + \beta_5H + \beta_6F + \beta_7S + \beta_8A + \beta_9G + \beta_{10}I$
8. Size of Ownership and Demographic	$\gamma(x) = \beta_0 + \beta_1S + \beta_2A + \beta_3G + \beta_4I$
9. Having Trees and Income	$\gamma(x) = \beta_0 + \beta_1H + \beta_2I$
10. Having Trees and Age	$\gamma(x) = \beta_0 + \beta_1H + \beta_2A$
11. Having Trees	$\gamma(x) = \beta_0 + \beta_1H$
12. Having Trees, Forest Plan and Age	$\gamma(x) = \beta_0 + \beta_1F + \beta_2H + \beta_3A$
13. Having Trees, (Future) Tenancy and Age	$\gamma(x) = \beta_0 + \beta_1H + \beta_2T_4 + \beta_3A$
14. Having Trees and Forest Plan	$\gamma(x) = \beta_0 + \beta_1H + \beta_2F$

Where the cumulative probability of enrollment at cost level j , is:

$$P(Y \leq j | \mathbf{x}) = e^{\gamma(x)} / (1 + e^{\gamma(x)}),$$

and Y , the observed response (yes/no) to the hypothetical program at cost j , depends on the vector \mathbf{x} , which contains a subset of the variables:

- A = Age, ordinal variable (1 to 5)
- G = Gender, categorical variable (1, 0) where 1 = “Male”
- I = Income, ordinal variable (1 to 4)
- F = Has Forest management plan, categorical variable (1, 0), where 1 = “Yes”
- H = Having trees on land now, categorical variable from the landowners response (1, 0), where 1 = “Yes”
- C_t = Managing for timber, categorical variable (1, 0) where 1 = “Yes”
- C_w = Managing for wildlife, categorical variable (1, 0) where 1 = “Yes”
- T_1 = Tenancy of current owner, ordinal variable (1 to 4) where 1 = <5 years, 2 = 5 to 15, 3 = 15 to 25 and 4 = >25 years
- T_2 = Tenancy of family, ordinal variable (1 to 5) where 1 = <5 years, 2 = 5 to 15, 3 = 15 to 25, 4 = 25 to 50, and 5 = >50 years
- T_3 = Live on this land, categorical variable (1, 0) where 1 = “Yes”
- T_4 = Likelihood of future tenancy of family, ordinal variable (1 to 4) where 1 = “Very unlikely”, 2 = “Unlikely”, 3 = “Likely”, and 4 = “Very likely”
- S = Size of ownership, ordinal variable (1 to 4) where 1 = <520 acres, 2 = 20 to 80, 3 = 80 to 160, 4 = 160 to 320, and 5 = >320 acres

and β_0 is a vector of intercepts for each of the four cost-share levels, coded as ordinal variables (1 to 4) where 1=\$50/acre cost to the landowner, 2=\$125, 3=\$175 and 4=\$250.

residence (whether absentee or resident) might also affect commitment to a cost-share program [Lynch et al., 2002]. If a landowner already has a forest management plan, or already manages the land for timber or wildlife, this may indicate a greater willingness to commit to a long-term reforestation program [Marty et al., 1988]. Finally, models were evaluated to determine which combinations of parameters would provide the best predictions. We limited our analysis to the 14 models we judged most useful and for which we had a sufficiently large dataset to allow comparison across models using Akaike’s Information Criteria (AIC).

The inferences that we wish to make from our survey depend on having a meaningful model. An information-theoretic approach consists of the science and objective based development of *a priori* models. This set of candidate models allows researchers to avoid data dredging, over-fitting and the rush to hypothesis testing [Burnham and Anderson, 1998]. AIC has developed into a tool to estimate the best approximating model (from the candidate set) and allows the researcher to explicitly look at model selection uncertainty [Akaike, 1973; Burnham and Anderson, 1998].

Following Anderson et al. [2000] and Burnham and Anderson [1998], we proceed from multiple working hypotheses, through multiple testing to seek the hypothesis that loses “as little information as possible about the truth” [Anderson et al. 2000]. To achieve this we used Akaike’s Information Criteria [Akaike, 1973], adjusted for small sample sizes (AIC_c) as the basis for ranking models. The response variable in these models is the cost-share level (cost/acre to the landowner) coded as a polychotomous ordered variable. Several of the independent variables are also coded as ordinal categorical variables [McCullagh and Nelder, 2001].

Each respondent was asked, in addition to all the other items, whether they would enroll in a cost share program at each of four different levels (i.e. four questions per respondent), using the following language:

“Hardwood bottomland forest restoration involves the planting of specially selected and grown trees such as black walnut, swamp white oak, bur oak and others. These species have potential for **high commercial timber value** on a 60-80 year rotation and also provide **annual value** as the source of other forest products, such as nuts. These trees also provide food and habitat for wildlife and help soil conservation. The trees are planted on raised beds if necessary, fertilized and use fiber-based mats to control weeds. There is also the opportunity to interplant other annual **revenue** producing crops, such as red top clover, with the trees.

To encourage this forest restoration, the State of Missouri is considering developing different programs to encourage bottomland owners to participate. These potential programs all plant the same trees. They differ only in the cost share between you and the state.

Please read the four potential programs and indicate whether or not you would enroll in each one, and, if so, how many acres you would enroll.”

A) Professional advice, planning and trees are provided free of charge. Tree planting and maintenance are paid on a cost share of 10% (you)/90% (state). Your estimated cost: \$50 per acre. Would you enroll? Yes....• No...•

B) Professional advice, planning and trees are provided free of charge. Tree planting and maintenance are paid on a cost share of 25% (you)/75% (state). Your estimated cost: \$125 per acre. Would you enroll? Yes....• No...•

C) Professional advice, planning and trees are provided free of charge. Tree planting and maintenance are paid on a cost share of 35% (you)/65% (state). Your estimated cost: \$175 per acre. Would you enroll? Yes....• No...•

D) Professional advice, planning and trees are provided free of charge. Tree planting and maintenance are paid on a cost share of 50% (you)/50% (state). Your estimated cost: \$250 per acre. Would you enroll? Yes....• No...•

The four levels were \$50, \$125, \$175 and \$250 cost /acre to the landowner (i.e. at the \$50 level the landowner pays less per acre than at the \$125 level) with the remainder of the cost being born by the program. The cost levels were based on costs observed at a flood plain reforestation trial along the Missouri River using root propagation method seedlings, mounding and weed control [Dey et al, 2001; Treiman personal communication]. We would expect more landowners to enroll at successively

Table 3.—The maximum likelihood parameter estimates of Model #10: $\gamma(x) = \beta_0 + \beta_1H + \beta_2A$, where H=Have Trees and A=Age.

Parameter ^{1,2}	Estimate	Standard Error	Wald Chi-Square	Pr>ChiSq
β_0 Cost-share=\$250	-3.96	0.75	27.83	<.0001
β_0 Cost-share=\$125	-2.97	0.67	19.64	<.0001
β_0 Cost-share=\$50	-1.77	0.61	8.39	0.003
β_1 Have trees	1.45	0.63	5.25	0.021
β_2 Age 20-35	-0.57	0.80	0.50	0.479
β_2 Age 36-50	1.11	0.51	4.77	0.028
β_2 Age 51-65	0.28	0.40	0.48	0.485

¹No estimate was produced for cost-share level \$175. All respondents who would adopt at that level also reported that they would adopt at \$250.

²The parameter for Age 65 and over is a linear combination of the parameters for the other age variables (dummies). No responses were observed for Age less than 20 (i.e. no responding landowners in our mailing were younger than 20).

higher levels of cost-share assistance. Analysis was based on a logit function, using these four levels. This produces four separate “intercepts” or constants, one for each cost-share level. Parameter estimates and AIC values were thus calculated by using PROC LOGISTIC [SAS Institute, 2001].

Results

The response rate for the “Behavior” surveys was 51.2 percent after two mailings [Treiman and Dwyer, 2002], for a total N=133. Model ranking methods, however, required that only those surveys in which the respondents chose to answer *all* the questions which were involved in all 14 of the competing models be included in the analysis, yielding N=81. A smaller AIC_c statistic is more indicative of a parsimonious model. The results of the AIC_c ranking are listed below in Table 2. This table also reports the distance between the AIC_c and the minimum AIC_c (ΔAIC_c) and the Akaike weights (T) of the AIC_c statistics, representing the predictive likelihood between models. In our model construct, models with AIC_c that are more than four units lower than other models are preferred [Burnham and Anderson, 1998].

In Table 2, the five highest ranked models (models #10, 13, 11, 12 and 14) were within <4 ΔAIC_c units of the each other [see Burnham and Anderson, 1998, page 123; Buckland et al., 1997]. These models must also be considered competing models because of the uncertainty in estimates of model precision (that is, the small differences in AIC_c are not enough to say with certainty that one model outperforms another). All five of these models rely on whether the land already has trees. Age is contained in three of the models. Two models include the presence of a management plan. The reported likelihood of continued (future) tenancy appears in one model.

The highest ranked model (model #10) relies only on whether the land already has trees (defined as answering “Yes” to having forest, timber or wood lots on flood plain land) and the landowner’s age, described by an ordinal categorical variable. Coding age in this way allowed for the exploration of non-linear effects of age upon adoption. The parameter estimates for this model are found in Table 3.

Table 3 shows the differing intercepts ($\beta_{0 \text{ cost-level}}$) for each of the cost-share levels. These intercepts decrease from the lowest cost-share level, \$50/acre cost to the landowner, to the highest, \$250/acre. This matches our expectation that lower cost to the landowner will result in acres enrolled. Note that all landowners who responded that they would enroll at a cost of \$175/acre to themselves would also enroll at \$250/acre.

The parameter estimate for “Having trees” ($\beta_{1 \text{ have trees}}$) is positive. Landowners who already have some forested land, wood lots, interplantings, or other trees on their flood plain land are more likely to

Table 2—Model selection results and ranking, using the Akaike Information Criteria adjusted for small sample sizes (AIC_c). K is the number of parameters in each model (including the constant for each of the 4 cost levels). ΔAIC_c represents the difference of each model from the model with the lowest AIC_c and T is the Akaike weight, calculated as

$\exp(-1/2\Delta_i) / \sum_{r=1}^{14} \exp(-1/2\Delta_r)$ for each model i in the set of 14 models. These weights represent the likelihood that a model is the best model in the set of models. The model number refers to the more complete description of the model found in Table 1.

Hypothesis	Model #	K	AIC _c	ΔAIC _c	T
Having Trees and Age	10	4	167.393	-	0.326
Having Trees, (Future) Tenancy and Age	13	5	167.555	0.16	0.300
Having Trees	11	1	168.803	1.41	0.161
Having Trees, Forest Plan and Age	12	5	169.461	2.07	0.116
Having Trees and Forest Plan	14	4	170.619	3.23	0.065
Having Trees and Income	9	4	173.869	6.48	0.014
Having Trees and Demographic	3	8	175.981	8.59	0.013
Demographic Model: Age, Gender, and Income	1	7	179.838	12.44	0.004
Current Management and Demographic	4	9	181.574	14.18	0.001
Forest Plan and Demographic	2	8	182.062	14.67	0.000
Length of Tenancy, Having Trees and Demographic	5	19	187.223	19.83	0.000
Length of Tenancy, Having Trees, Forest Plan and Demographic	6	20	187.835	20.44	0.000
Length of Tenancy, Having Trees, Forest Plan, Size of Ownership and Demographic	7	24	198.081	30.69	0.000
Size of Ownership and Demographic	8	22	198.498	31.10	0.000

enroll. The parameter estimates for the effects of age ($\beta_{2 \text{ age range}}$) show that “middle-aged” landowners (ages 36-50) are the most likely to enroll. Being younger than 35 actually has a negative effect.

Discussion

Elsewhere, Treiman and Dwyer [2002] used data from both the “Knowledge” and “Behavior” surveys to predict that at the lowest cost-share level (\$50/acre to the landowner), up to 13 percent of Missouri River flood plain land would be enrolled in a reforestation cost-share program. A total enrollment of 8,600 acres, at cost of \$1.7 million to the sponsoring agency, was predicted. But this ranking of competing models allows decision makers to see just who those potential enrollees are, and, more importantly, who they are not.

The only variable to appear in all five competing models (models #10, 11, 12, 13 and 14) is whether or not the landowner’s flood plain land already has forest, timber or wood lots on flood plain land. These owners have trees on *some* of their land but may be interested in enrolling in the hypothetical cost-share program in order to reforest other land. Having trees on the land is important for two reasons. First, it may indicate that the landowner already has an interest in forestry, even if this interest is only passive. Second, it may indicate that the landowner is not solely interested in row crops.

In Treiman and Dwyer’s separate “Knowledge” survey, row crops were identified by the average survey respondent as both their most important short-term and long-term goal, with an average ranking of 4.1 on a scale of 1 (Not Important) to 5 (Very Important). However, landowners with trees ranked row crops lower (3.9), than those without trees on their flood plain land (4.4), although this result is not significant at the 95% level [Treiman and Dwyer, 2002].

Whether or not the landowner has developed a forest management plan was the only variable related to current management practices included in any of the competing models. Having a forest management plan increased the likelihood of adoption across all cost-share levels. However, only 9 percent of Missouri River flood plain landowners report having such a plan [Treiman and Dwyer, 2002]. These landowners hold about 4,570 acres of the 71,400 private Missouri River flood plain acres outside the

levee. For the most part these landowners have, when developing their plan, worked with State, Federal or University-Extension foresters, with over 63 percent reporting such contact. In the “Knowledge” survey, those landowners also reported a favorable experience when working with those foresters, ranking the quality of forestry service received at an average of 3.5 on a scale of 1 (Poor) to 5 (Excellent) [Treiman and Dwyer, 2002].

Age is the only variable from the original demographic model to remain in the set of competing models. Neither income nor gender proved to be particularly useful as a predictor of adoption. The landowner’s age affects the likelihood of adoption of reforestation practices in a non-linear manner, with landowners between 36 and 50 being more likely to adopt than either younger or older landowners. Another demographic variable, the reported likelihood that the land would remain in the respondent’s family into the future, also appears in one of the competing models. The more likely the respondent thought that the land would remain in the family, the more likely they were to say that they would adopt the cost-share program.

Data on all of these predictive variables are available to program managers and planners, as well as county-level state forestry agents. County-level foresters know which land in their counties is already in trees. Program managers know who already has a management plan (in Missouri these plans are co-signed by the state forestry agency). Landowner age is, of course, confidential but could be “eye-balled” by the forester.

None of the other variables included in the original 14 models proved to be of much predictive value. It did not seem to matter how landowners were currently managing their land. Neither did the size of their ownership nor the length of their tenancy enter into any of the final five competing models.

Conclusions

The landowner most likely to enroll his/her land in even the most generous flood plain reforestation cost-share program is the landowner who already has trees and who has already worked closely enough with forestry agencies to have developed a forest management plan. The likeliest landowner is also middle-aged, perhaps with enough time to have settled down but also with enough of his/her lifetime remaining to commit to the long-term nature of forestry. This landowner also believes that his/her land will stay in his/her family into the future, so that his/her family will be there to enjoy the long-term benefits of reforestation as a bequest.

The hypothetical cost-share programs we have described to the survey respondents turn out to be most attractive to those who are or should already be the best clients of state, federal or university-extension forestry agencies. With the passage of the 2002 Farm Bill, significant new funding will be forthcoming to provide technical and financial assistance to landowners [United States Department of Agriculture, 2002]. This new funding would be most effective if directed towards these most likely adopters.

The only variable in the five competing models that is in any way under the control of state, federal or university-extension forestry agencies is the forest management plan. This is the opening by which these agencies can get people involved in making forest management decisions about their property. The challenge for forestry agencies interested in sustaining mature flood plain forests and securing adequate regeneration (natural or artificial) in the flood plain will be to go beyond the 13 percent of land that we predict would be enrolled in these hypothetical cost-share programs. The results of this survey show that state agencies must make that first, positive contact with the “unlikely” landowners, transforming them into “likely” landowners. This can lead to the development of a forest management plan and enrollment in a cost-share program. If that first tree is planted (or preserved) on row crop land, it can help turn that unlikely adopter into a likely adopter. Dedicating agency staff and resources to these contacts is vital, as is training staff to make that first contact a success. The questions remains, however; if unlikely landowners are converted into likely ones, where will state agencies find the funding and personnel to expand their cost-share programs?

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COMPOSITION AND DEVELOPMENT OF NON-TREE VEGETATION AND ITS RELATIONSHIP WITH TREE REGENERATION IN MIXED-OAK FOREST STANDS

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ABSTRACT.— As part of a large-scale study of oak regeneration in Pennsylvania, non-tree vegetation was measured on 46 mixed-oak stands. Blueberry, hayscented fern, mountain-laurel, and huckleberry were, by percentage cover, the four most abundant non-tree species. After harvest, grasses, sedges, forbs, blackberry, and hayscented fern had pronounced expansion in non-herbicide treated stands. In the herbicide-treated stands, both hayscented fern and mountain-laurel abundance were significantly reduced after herbicide application. Regeneration density, survivorship, and growth rate were examined in different non-tree vegetation types. White oak regeneration was most abundant on subplots having moderate blueberry or huckleberry cover. Abundant chestnut oak tended to occur on plots with heavy cover of blueberry or huckleberry. High levels of red maple regeneration were commonly associated with moderate cover of hayscented fern.

Introduction

In the mixed-oak forests of Pennsylvania, dense ground covers of blueberry (*Vaccinium spp.*), hayscented fern (*Dennstaedtia punctilobula* (Michx.) Moore), and other non-tree species can interfere with the development of oak advance regeneration (Allen and Bowersox 1989, Steiner and Joyce 1999, Horsley et al. 1992). Hayscented fern has been classified as a competitor species because of its ability to respond aggressively to sudden resource availability with vegetative expansion of rhizomes and sexual reproduction (Groninger and McCormick 1991, Hughes and Fahey 1991). Blueberry, huckleberry (*Gaylussacia baccata* Wang.), and mountain-laurel (*Kalmia latifolia* L.) are common associates in oak forest understories in this region, and they differ considerably in how they influence regeneration and stand structure. This is partially due to the difference in height achieved by mountain-laurel compared to the relative low height of blueberry and huckleberry. Mountain-laurel, in particular, has created concerns regarding management for desirable tree seedlings in hardwood stands because of its aggressive vegetative growth habit (Moser et al. 1996). Although mountain-laurel has little effect on regeneration establishment, it does suppress the growth of small seedlings (Waterman et al. 1995). In light of the regeneration problem facing forest managers in Pennsylvania, there is a need to better understand the distribution and dynamics of competing species (McWilliams et al. 1995). This paper focuses on the distribution and development of the non-tree vegetation at a landscape scale and its association with the establishment, survivorship, and growth rate of advanced and post-harvest tree seedling regeneration.

Study Areas

The study includes 46 mixed-oak stands on a total area of 2069 acres across the Appalachian Plateau and Ridge and Valley physiographic provinces of Pennsylvania. Stand area varies from 15 to 80 acres with an average size of 45 acres. Prior to harvest, oaks (*Quercus spp.*) were the dominant species in all the stands included in this study. Soils in both provinces are derived from sandstone, siltstone, or shale and are typically well-drained and support moderately productive forests. Stand elevations range from 1,000 ft in the Ridge and Valley province to 2,400 ft on the Appalachian Plateau. Mean annual precipitation ranges from 38 to 45 inches and frost-free periods range from 140 to 160 days, both vary with elevation and topography (Cuff et al. 1989).

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Table 1.— Non-tree vegetation average percentage cover, occurrence frequency (percentage of subplots), and heavy-cover frequency (percentage of subplots with > 30 percent cover by species) in 46 mixed-oak forest stands prior to harvest.

Species	Avg. Cover (%)	Freq. (%)	>30 Freq (%)
Blueberry (<i>Vaccinium spp.</i>)	10.3	71.9	12.1
Bracken fern (<i>Pteridium aquilinum</i>)	1.2	12.4	0.8
Forbs	0.2	39.5	0.1
Grass	0.7	17.6	0.7
Hayscented fern (<i>Dennstaedtia punctilobula</i>)	9.4	25.9	13.8
Huckleberry (<i>Gaylussacia baccata</i>)	5.1	21.8	7.3
Mountain-laurel (<i>Kalmia latifolia</i>)	8.8	29.3	12.5
Sedge	0.2	11.7	0.1
Teaberry (<i>Gaultheria procumbens</i>)	0.2	35.7	0.0
Witch-hazel (<i>Hamamelis virginiana</i>)	2.1	19.0	2.7

Methods

Field measurements in the 46 mixed-oak stands were performed during 1996 – 2002. All the stands were measured one year prior to harvest (clearcuts or shelterwoods), and 30 stands have been re-measured one year after harvesting. Depending on stand area, 15 to 30 twentieth-acre permanent plots (26.3 ft. radius) were systematically installed in a square grid to sample across the entire stand. Four permanent milacre subplots (3.72 ft. radius), were established within each plot. On each subplot, percentage cover of non-tree vegetation was estimated (in 5 percent increments) by species or species group, as was tree regeneration density and height by species. In total, 4830 pre-harvest subplots and 2898 one-year-after harvest subplots were included in this study.

Herbicide treatment was applied on 13 of the 46 stands, where hayscented fern densities appeared likely to inhibit tree regeneration. Six herbicide stands were re-measured one year post-harvest. Herbicide treatments were based upon the forester's management objectives for each stand and were not experimentally controlled. Stands were treated with herbicide if hayscented fern densities appeared likely to inhibit regeneration. OUST (sulfometuron methyl) or an ACCORD/OUST mix was applied to dense hayscented fern at various rates depending on fern density. The primary objective of the treatments was to establish and/or release desirable regeneration.

To assure a fair comparison of vegetation change before and after harvest, only 30 stands that were surveyed both before and after harvest are used. Those 30 stands were divided into two groups, non-herbicide and herbicide. The non-herbicide group included 24 stands (2273 subplots), and the herbicide group included 6 stands (625 subplots).

For the purpose of classifying, percentage cover of non-tree vegetation was grouped into four classes: none, low (1 to 10 percent cover), moderate (11 to 30 percent), and heavy (over 30 percent). The heavy class reflects the threshold level of competing vegetation considered problematic by Marquis (1994). Species frequency was obtained by counting all the subplots containing a given species and then dividing by the total number of subplots across all stands. Non-tree vegetation average percentage cover, occurrence frequency, and heavy-cover frequency (subplots with over 30 percent of cover by species) before harvest was calculated across all subplots. Regression analysis was performed between post-harvest non-tree cover and regeneration survivorship and percent growth. Percent growth is the relative change of cumulative height $((\text{post-pre})/\text{pre} \times 100)$, where cumulative height was the total height of all the stems for species or species group on a subplot.

Results

Pre-harvest Species Composition and Distribution

Species with occurrence frequency over 10 percent are listed in Table 1. Blueberry was the most abundant understory vegetation species both in percentage of cover and occurrence frequency. On

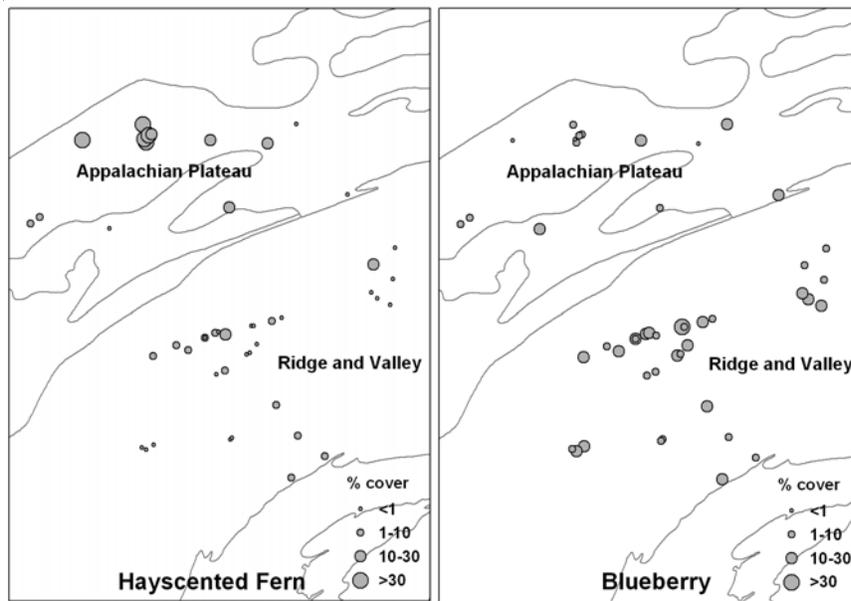


Figure 1.—Regional distribution of hayscented fern and blueberry average percentage cover prior to harvest.

average, blueberry covered 10.3 percent of the mixed-oak forest floor, and it was present in 71.9 percent of all the subplots. Hayscented fern was the second most abundant species (9.4 percent cover). Its occurrence tended to be the most concentrated; over half of the subplots in which it was present had above 30 percent cover. Mountain-laurel and huckleberry were the next two most abundant non-tree species by cover (8.8 and 5.1 percent, respectively). Heavy cover of hayscented fern, mountain-laurel, and blueberry all occurred on over 10 percent of all the subplots. Although low in coverage, forbs and teaberry (*Gaultheria procubens* L.) were the second and third most common species or species groups by frequency. They tend to occur “anywhere” but nowhere in great abundance.

Generally, prior to harvest, stands on the Appalachian Plateau had a higher percentage of hayscented fern cover than did stands in Ridge and Valley region (fig.1). However, there were three stands on the Plateau that had low fern cover and two stands in the Ridge and Valley that had moderate fern cover. Blueberry, on the other hand, was more abundant in the Ridge and Valley region than on the Plateau, but the difference was not as remarkable as for hayscented fern.

Vegetation Changes Pre- vs. Post-harvest

A comparison between non-tree vegetation before and after harvest is shown in Table 2. As mentioned, the stands in which herbicide was applied had high density of hayscented fern, so coverage by this species is an obvious and expected distinction between the two stand categories. Blueberry had the most abundant average cover in the non-herbicide group.

Considerable changes were observed in the non-herbicide stands one year post-harvest. Before harvest, blueberry, mountain-laurel, and huckleberry were the most abundant non-tree species, having average cover of 5 percent or more. One year post-harvest, however, only blueberry and hayscented fern retained over 5 percent cover. Meanwhile, hayscented fern occurrence frequency had a dramatic increase of almost three-fold compare to pre-harvest. Other herbaceous species and species groups, including forbs, grasses, sedges, bracken fern, and blackberry (*Rubus alleghaniensis*) also increased in cover and occurrence frequency. For mountain-laurel and huckleberry, the frequency of heavy-cover subplots was reduced one year after harvest.

Table 2.—Average non-tree vegetation percentage cover, occurrence frequency (percentage of subplots), and heavy-cover frequency (percentage of subplots with > 30 percent cover by species) before and one year after harvest in herbicide and non-herbicide stands.

Species	Avg. Cover (%)		Freq. (%)		Heavy-cover Freq. (%)	
	Pre-	Post-	Pre-	Post-	Pre-	Post-
Non-herbicide treatment						
Blackberry	0.0	0.1	0.4	9.9	0.0	0.0
Blueberry	14.0	10.8	64.8	75.6	17.9	13.2
Bracken fern	0.2	0.6	2.1	5.7	0.1	0.7
Forbs	0.3	2.2	2.8	61.3	0.0	2.1
Grass	0.3	3.2	2.7	29.1	0.1	3.9
Hayscented fern	5.1	5.3	11.1	29.6	7.1	6.7
Huckleberry	7.6	3.1	26.3	21.3	11.4	3.8
Mountain-laurel	8.5	2.8	25.9	24.5	11.7	2.8
Sedge	0.3	1.7	2.5	17.0	0.1	2.1
Teaberry	0.1	0.1	0.9	32.2	0.0	0.0
Witch-hazel	2.7	1.0	11.0	17.0	3.6	1.0
Herbicide treatment						
Blackberry	0.0	3.8	0.2	18.6	0.0	4.8
Blueberry	6.4	3.8	36.6	52.0	6.4	1.6
Bracken fern	1.7	0.3	10.7	10.1	1.9	0.3
Forbs	0.1	0.4	1.3	49.9	0.2	0.0
Grass	1.2	1.2	8.0	31.7	1.6	1.0
Hayscented fern	28.3	2.3	54.9	33.6	41.8	2.4
Huckleberry	0.5	0.6	2.2	11.7	1.0	0.6
Mountain-laurel	14.3	3.4	30.4	28.6	21.4	2.1
Sedge	0.3	0.5	2.7	22.1	0.0	0.0
Teaberry	0.1	0.3	1.6	25.4	0.0	0.0
Witch-hazel	2.6	1.1	7.0	10.9	3.8	1.8

In the herbicide stands, hayscented fern was reduced dramatically both in terms of cover (28.3 to 2.3 percent) and occurrence frequency (54.9 to 33.6 percent). In addition, subplots with a problematic level of fern cover were almost eliminated, which indicated that the herbicide treatments were successful. The herbicide treated stands also had a significant reduction of mountain-laurel, both in cover (14.3 to 3.4 percent) and heavy-cover frequency (21.4 to 2.1 percent).

Non-tree Vegetation Abundance vs. Regeneration Density

Interesting associations between non-tree vegetation density class and advanced regeneration density of tree seedlings were found across the study sites (fig. 2). The four most abundant non-tree species (i.e., blueberry, hayscented fern, mountain-laurel, and huckleberry) and four regeneration species (i.e., red maple (*Acer rubrum* L.), northern red oak (*Quercus rubra* L.), white oak (*Quercus alba* L.), and chestnut oak (*Quercus prinus* L.)) are presented for comparison.

Red maple regeneration was most abundant in association with low cover of mountain-laurel and moderate cover of hayscented fern, and regeneration was least abundant where coverage of both those species was heavy. Surprisingly, northern red oak was most abundant on plots with moderate hayscented fern cover and occurred at low abundance where hayscented fern was absent. Red oak regeneration density decreased as the cover of blueberry and huckleberry thickened. White oak was most abundant on plots with moderate cover of huckleberry and blueberry, and chestnut oak was most abundant on plots with moderate to high cover of those species.

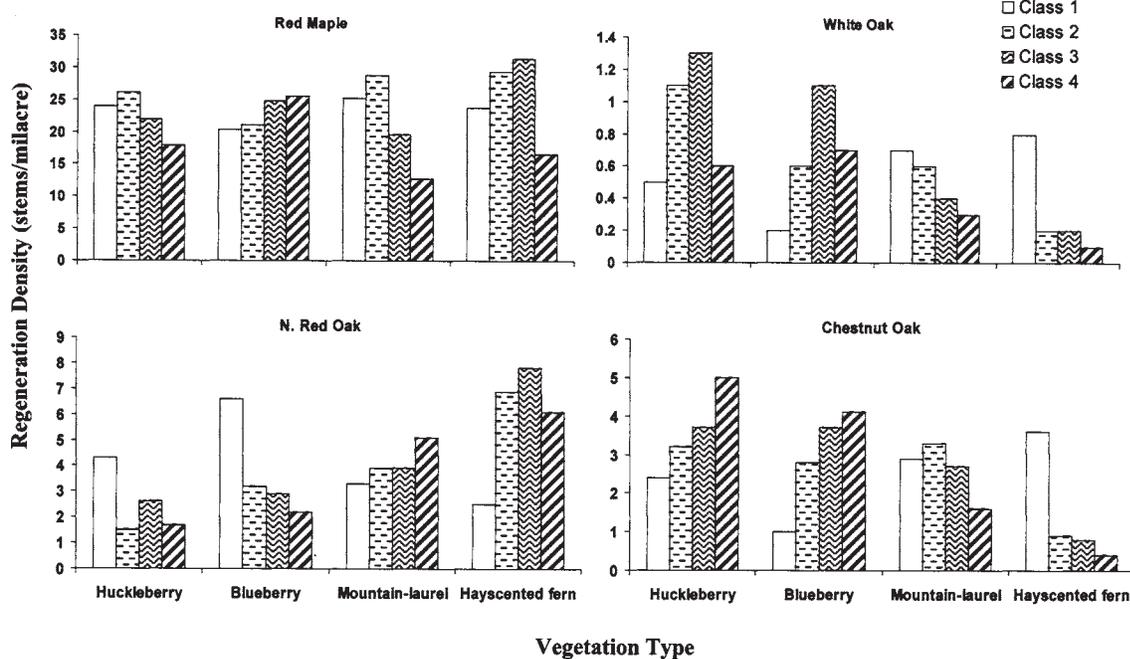


Figure 2.—Advanced regeneration density of four tree species under different cover classes (class 1: no cover; class 2: 1-10 percent; class 3: 11-30 percent; class 4: >30 percent) of four non-tree vegetation types.

No statistically significant relationships between vegetation type and coverage and regeneration survival rates were found. Nor were there any significant relationships between vegetation type and regeneration growth rate except for the relationship between red maple and blueberry (fig. 3). Percentage growth in cumulative height of red maple decreased as the blueberry cover increased ($p = 0.003$). If a stand had an average blueberry cover approximately over 10 percent, then the percent growth for red maple would be less than zero, which implies that red maple lost its density and cumulative height under those conditions.

Discussion and Conclusions

Blueberry was the most frequent non-tree species in the understory across all stands in the study area, and it was particularly common in the Ridge and Valley ecological province. Plots with moderate cover of blueberry and huckleberry had the most abundant white oak regeneration, and plots with heavy cover of blueberry and huckleberry had the most abundant chestnut oak regeneration. Rogers (1974) pointed out that heath communities dominated by blueberry and huckleberry have an affinity for infertile sites with well-drained acidic soils. The affinity of some oaks for similar environmental conditions may at least partially explain why regeneration of white oak and chestnut oaks was associated with blueberry and huckleberry. Blueberry and huckleberry had an average height of about two feet on our surveyed plots. Moderate to heavy cover of this low shrub layer might have reduced deer predation of acorns and small seedlings, therefore facilitating the establishment of oak regeneration.

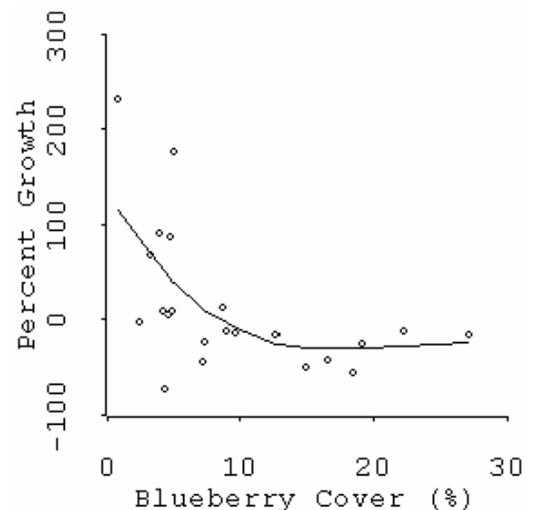


Figure 3.—Relationship between red maple percent growth in cumulative height ((post-pre)/pre*100) and average blueberry percentage cover in each stand.

Competition from hayscented fern has been identified as an important factor contributing to the regeneration problem in Pennsylvania (McWilliams et al. 1995). The abundance of hayscented fern in mixed-oak forest understories appears to suppress desirable tree seedlings by decreasing light quantity and quality beneath the herbaceous layer (Horsely 1993, George and Bazzaz 1999). In our study, herbicide treatment was an effective control of hayscented fern where it was used. But if herbicide was not used and hayscented fern was present, it spread both in cover and frequency following harvest. In addition, moderate levels of fern cover were associated with dense red maple regeneration. Similar association between fern cover and red maple basal area was also found in our recent research (unpublished). The combination of a dense fern cover and abundant red maple in both the overstory and understory will likely be detrimental to oak regeneration.

The co-occurrence of moderate levels of hayscented fern and high northern red oak regeneration is difficult to interpret. One possible explanation was that northern red oak can escape detection by deer under fern that helped its establishment, but neither white oak nor chestnut oak tends to occur where fern is common, and the maple doesn't matter because it is not a preferred deer food.

The following are our conclusions:

- In the mixed-oak forest in Pennsylvania, blueberry, hayscented fern, mountain-laurel, and huckleberry were the four most abundant species by cover.
- Hayscented fern was more abundant on the Appalachian plateau, while blueberry was more abundant in the Ridge and Valley ecological province.
- Red maple was most abundant on subplots with moderate levels of hayscented cover, while hayscented fern was negatively associated with both white oak and chestnut oak. Higher levels of blueberry and huckleberry were associated with higher density of oak regeneration.
- No significant relationships between non-tree species or species groups and regeneration survivorship and growth rate were identified, except there was a negative relationship between red maple growth rate and blueberry cover percentage.

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EFFECTS OF CANOPY AND HERBACEOUS COVERAGE ON REGENERATION IN A FOREST COMMUNITY IRRIGATED WITH TREATED WASTEWATER IN CENTRAL PENNSYLVANIA

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ABSTRACT.—Forest communities on Pennsylvania State Gamelands 176 (Toftrees) have been irrigated weekly with 5 cm of treated municipal wastewater since 1983. The understory of the irrigated forest communities was dominated by herbaceous vegetation and lacked tree regeneration. Our objective was to determine the effect of the existing canopy coverage and understory vegetation on natural and artificial regeneration.

Canopy coverage and the presence of herbaceous vegetation did not affect establishment of natural regeneration. After two growing seasons, the density of seedlings from natural regeneration was <1 seedling m⁻². However, artificial regeneration did respond to canopy treatments. Planted seedlings of five tree species had the greatest survival and height growth on open canopy treatments. Removal of herbaceous vegetation also improved the survival of artificial regeneration.

The results of this study determined that natural regeneration did not occur in sufficient numbers within the irrigated forest community. However, artificial regeneration can be used to augment these numbers. Creation of small, open canopy areas within the existing irrigated forest communities, treatment of the understory with herbicide, and planting a selection of species capable of growing under the irrigation conditions will ensure a future forest community.

Introduction

The Pennsylvania State University has been irrigating its treated municipal wastewater onto forest and agricultural land since 1983. The existence of a healthy forest community has become important to the success of a year-round wastewater application program, especially in the winter months when agricultural soils tend to freeze. Five cm of treated wastewater have been irrigated weekly onto forest and agricultural lands at Pennsylvania State Gamelands 176 (Toftrees). During this period of irrigation, an alteration of species composition and forest stand structure had occurred. The irrigated forest communities had low overstory and understory densities, lacked regeneration of tree species, and the understory was dominated by herbaceous plants, indicating that the natural regeneration processes were not occurring (Larrick and Bowersox 1999).

As trees die in a forest community, small openings, or gaps, in the canopy are created. Windstorms also create larger gaps by blowing down one or more adjacent canopy trees. When the surrounding trees do not fill the gaps in the canopy, the understory plants become the beneficiaries of light, water, nutrients, and growing space, thus starting the gap-phase stage of stand development (Watt 1947, Bormann and Likens 1979, Hibbs 1982). In most forest communities, established tree seedlings and saplings (advanced regeneration) captured the increased resources within the gap and increased their growth (Oliver and Larson 1996). However, the irrigated forest community lacked this gap-phase dynamic. Advanced regeneration was not established so that when gaps were created, herbaceous vegetation captured the resources and dominated the site. The objective of this study was to determine the effect of the existing canopy and understory herbaceous vegetation on natural regeneration processes. In

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addition, seedlings were planted to determine species survival and growth under the growth the irrigated conditions.

Study Areas

The study site was located on Pennsylvania State Gamelands 176 (Toftrees), approximately 2 km north of University Park, PA. The Toftrees area was leased by the university from the Pennsylvania Game Commission in 1983 to expand its municipal wastewater irrigation system. The Toftrees area is located in the Ridge and Valley region of central Pennsylvania (Lull 1968). The average annual precipitation is 95 cm (Braker 1981). Site quality was average, and if occupied by an even-aged mixed oak stand, the site index would be 20 m at 50 years (Larrick and Bowersox 1999). Soils present on the site were primarily Morrison (Ultic, Hapludalf) and Hagerstown (Typic, Hapludalf).

Materials and Methods

Twelve 0.20 ha circular plots were located in the irrigated forest communities. The DBH and species of the overstory trees were recorded. Three canopy treatments were randomly assigned to each of the plots (four replicates per canopy treatment). The canopy treatments were closed canopy, partial canopy, and open canopy. Closed canopy treatment plots had no overstory trees removed; partial canopy treatment plots had enough overstory trees removed to obtain a 70-75% stocking level; and the open canopy treatments had all overstory trees removed. Canopy treatments occurred in April 2000.

Within each 0.20 ha canopy treatment plot, four 10 x 10 m understory treatment plots were established. Each subplot was assigned one of two understory treatments (herbicide, control). Two plots were treated with herbicide and two were untreated. The herbicide treatment was 2% glyphosate applied in May. The understory subplots were enclosed with wire fencing (2 m in height) to exclude white-tailed deer (*Odocoileus virginianus*) foraging on the vegetation.

In May 2000, five species were planted as seedlings within each of the understory treatment plots. The species included quaking aspen (*Populus tremuloides* Michx.), red maple (*Acer rubrum* L.), black gum (*Nyssa sylvatica* Marsh.), river birch (*Betula nigra* L.), and Norway spruce (*Picea abies* (L.) Karst.). A total of 1,440 seedlings were planted (six seedlings per species per understory treatment plot). Height growth of seedlings was recorded in August.

Natural regeneration was monitored on three 2 x 2 m areas within each understory treatment plot. Seedlings were inventoried in August by species and size. Also, the percent ground cover and aboveground biomass of herbaceous vegetation were inventoried on the understory plots not treated with herbicide.

Soil samples were collected from each understory treatment plot in October. The samples were sent to the Pennsylvania State University Agricultural Analytical Laboratory for analysis. The samples were analyzed for pH, phosphorus, cation exchange capacity (CEC), the base saturation of potassium, magnesium, and calcium, total nitrogen, and nitrate-nitrogen. Measurements of all variables were taken for two growing seasons (2000 and 2001), however, only the findings of the second growing season are presented for brevity.

Statistical Analysis

Analysis of variance (ANOVA) was used to determine if significant differences existed among canopy treatments for mean basal area and density before and after canopy treatments. The effects of canopy and understory treatments and their interaction on height growth of the artificial regeneration were also determined using ANOVA. Main effects and interactions were significant at the $p < 0.05$ level. Tukey's mean separations were used to determine differences among treatments and were significant at the $p < 0.05$ level.

Survival of artificial regeneration was modeled using binary logistic regression. The independent variables included species, and canopy and understory treatments. Model I included the variable

Table 1.—Mean tree density and basal area before (Fall 1999) and after (Spring 2000) canopy removal treatments on Pennsylvania State Gamelands 176 (Toftrees).

	Density trees ha ⁻¹		Basal Area m ² ha ⁻¹	
	<i>Before</i>	<i>After</i>	<i>Before</i>	<i>After</i>
Closed canopy	366 ab	366 a	22 ab	22 a
Partial canopy	374 a	293 b	24 a	18 b
Open canopy	326 b	0 c	18 b	0 c

species. Model II also included canopy treatment. Model III included all three variables of species, canopy and understory treatments. The dependent variable was survival (0=dead; 1=alive). Each independent variable had a reference or control. Quaking aspen was the reference for species because it had been studied within the irrigated community (Jacobs 1997, Kelso 1999). Closed canopy was used as the reference for canopy treatment because it reflected the current conditions of the irrigated forest community. Similarly, plots not treated with herbicide were used as a reference because of the exiting herbaceous vegetation that dominated the understory.

Results

Prior to tree removal, the mean stem density and basal area of the canopy treatment plots was similar (table 1). The partial canopy treatment had the greatest stem density with 374 trees ha⁻¹, but was not significantly different from the closed (366 trees ha⁻¹) or the open canopy treatments (326 trees ha⁻¹). The partial canopy treatment plots had a greater mean basal area prior to treatment with 24 m² ha⁻¹, and was not significantly different from the closed canopy (22 m² ha⁻¹). The open canopy treatment plots had the least amount of basal area with 18 m² ha⁻¹ and were significantly different from the partial canopy treatments, but not the closed canopy treatments.

Following overstory tree removal, there were significant differences in the mean stem density and basal area among canopy treatments (table 1). The stem density and basal area of the closed canopy treatments remained the same, whereas the stem density of partial canopy treatment plots was reduced to a mean of 293 trees ha⁻¹ and the open canopy had no trees remaining on these plots. The average basal area of the partial canopy plots was reduced by 6 m² ha⁻¹ to 18 m² ha⁻¹, and the open canopy had a basal area of zero.

Mean relative density of the irrigated forest after canopy treatments was represented by the species of the white oak group [(white oak (*Quercus alba* L.), chestnut oak (*Quercus prinus* L.)) at 37 percent, red maple and the species of the red oak group [northern red oak (*Quercus rubra* L.), black oak (*Quercus velutina* Lam., and scarlet oak (*Quercus coccinea* Muenchh.)] both at 25 percent. All other species [including quaking aspen, black cherry (*Prunus serotina* Ehrh.), and eastern white pine (*Pinus strobus* L.)] each represented less than 4 percent of the relative density.

Overall the number of stems recorded on the natural regeneration plots was very low and we were unable to perform a statistical analysis. The number of tree stems recorded was 194 across all natural regeneration subplots (<1 seedling m⁻²). Eighty percent of the recorded seedlings were on understory plots that were treated with herbicide, and 75 percent of the seedlings were recorded on the open canopy treatment plots. Red maple was the most abundant species, followed by black cherry. However, all inventoried seedlings were less than 25 cm in height.

The ground coverage and aboveground herbaceous biomass were significantly different among the three canopy treatments (table 2). Mean ground coverage on closed canopy treatments was 60 percent. On

Table 2.—Mean ground coverage aboveground herbaceous biomass during the second (2001) growing seasons by canopy treatment on Pennsylvania State Gamelands 176 (Toftrees).

	Ground Coverage	Biomass
	percent	g m ⁻²
Closed canopy	60 c	80 c
Partial canopy	79 b	139 b
Open canopy	100 a	496 a

Table 3.—Logistic regression models¹ for the survival of planted seedlings after the second growing season (2001) on Pennsylvania State Gamelands 176 (Toftrees).

Dependent Variable: Alive =1; Dead=0	Model I		Model II		Model III	
	<i>b</i>	<i>e^b</i>	<i>b</i>	<i>e^b</i>	<i>b</i>	<i>e^b</i>
Species (reference=quaking aspen)						
Red maple	0.6*	1.9	0.9*	2.4	1.0*	2.8
Black gum	-0.1*	0.9	0.1	1.1	0.2	1.2
River birch	-1.4*	0.2	-1.3*	0.3	-1.3*	0.2
Norway spruce	2.2*	8.9	2.5*	12.1	2.7*	14.1
Canopy (reference=closed)						
Partial			0.5*	1.6	0.5*	1.7
Open			1.5*	4.3	1.6*	4.9
Understory (reference=no herbicide)						
Herbicide					1.3*	3.6
Intercept		0.5*		-0.4		-1.2*
-2 Log likelihood (d.f.)		1,113 (4)		1,050 (6)		938 (7)
Model Chi-square (d.f.)				63 (2)*		67 (1)*

¹Coefficients designated by (b) and odds ratio by (*e^b*).

partial canopy treatment plots mean ground coverage was 79 percent. The open canopy treatments had the greatest mean ground coverage, 100 percent. The open canopy treatment plots had a mean aboveground biomass of 496 g m⁻², which was 3.5 times greater than the mean biomass produced on the partial canopy treatments (139 g m⁻²), and 6 times greater than the closed canopy treatment plots (80 g m⁻²).

A total of 691 seedlings were alive at the end of the second growing season. Seedling survival at the end of the second growing season was greatest on the open canopy plots and the herbicide subplots for all species planted. Model I demonstrated the relationship between seedling survival and species (table 3). Survival of all four species was significantly different from the quaking aspen seedlings. Seedlings of red maple and Norway spruce were more likely to survive than the quaking aspen. Black gum and river birch seedlings had lower survival than quaking aspen seedlings.

In Model II, the addition of the independent variable of canopy treatment changed the species analysis. Black gum seedling survival was no longer significantly different from quaking aspen seedling survival, and the coefficient became positive. Black gum seedling survival was greater than quaking aspen seedlings on two of three canopy treatment plots, which accounted for the change in signs of the coefficient. The open canopy treatment plots had seedling survival 4.3 times greater than the closed canopy. Seedlings on the partial canopy plots were 1.6 times more likely to survive than seedlings on closed canopy plots. The addition of understory to the Model III did not change the significance of the other independent variables, and was significant. The use of herbicides increased seedling survival by a factor of 3.6.

At the end of the second growing season, a response to canopy treatment was observed for all planted species (table 4). Some of the seedlings experienced dieback from the terminal bud, which accounted for negative height growth. Quaking aspen seedlings on the open canopy treatment plots had a mean height growth of 117 cm, which was significantly different from seedlings on the partial (31 cm) and the closed (17 cm) canopy treatments. River birch seedlings had the second greatest mean height growth on the open canopy plots with 41 cm and was significantly different from seedling height growth on the closed canopy (-19 cm) treatment plots, but not from the partial canopy (-3 cm) treatments. Seedlings of red maple and black gum were similar in their height growth responses to the canopy treatments. Both species had the greatest response on the open canopy treatments, a slight height growth response on the partial canopy treatments, and negative height growth on the closed

Table 4.—Mean height growth (cm) of planted seedlings by species, canopy and understory treatment after the second growing season (2001) on Pennsylvania State Gamelands 176 (Toftrees).

	Quaking aspen	Red maple	Black gum	River birch	Norway spruce
Canopy					
Closed	17 b	5 b	-2 b	19 b	0 b
Partial	31 b	2 b	1 b	3 ab	5 b
Open	116 a	28 a	17 a	41 a	11 a
Understory					
No Herbicide	64 a	7 a	5 a	1 a	6 a
Herbicide	72 a	10 a	7 a	17 a	5 a

Table 5.—Mean values for soil variable based on canopy treatment for the second growing season (2001) on Pennsylvania State Gamelands 176 (Toftrees).

	pH	P	CEC	K	Mg	Ca	Total-N	Nitrate-N
	g m ⁻²	meq 100 g ⁻¹	percent saturation	percent	ppm			
Canopy								
Closed	7.5 a	39.8 ab	20.7 a	1.4 b	28.8 a	69.8 a	0.48 a	51.4 a
Partial	7.4 ab	38.1 b	17.6 b	1.4 b	27.4 a	71.2 a	0.32 b	34.6 b
Open	7.6 a	49.1 a	20.8 a	2.4 a	29.4 a	68.2 a	0.52 a	60.7 a

canopy treatments. Red maple seedlings had a mean height growth of 28 cm on the open canopy and were significantly different from the partial (2 cm) and closed canopy (-5 cm) treatment plots. Black gum seedlings had 17 cm of height growth in the open canopy and were significantly different from the partial (1 cm) and open (-2 cm) canopy treatment plots. Finally, Norway spruce seedlings had the lowest mean response on the open canopy treatment (11 cm), and was significantly different from seedlings on the partial (5 cm) and closed (0 cm) canopy treatments. After the second growing season, there was no significant difference in the response of planted seedlings to understory treatments or to the interaction of canopy by understory interaction.

Soil analysis demonstrated the high nutrient content of the irrigated soils (table 5). During the second growing season, there were significant differences among canopy treatments for six of the eight measured soil variables. The partial canopy treatment plots had mean soil variables that were significantly different from the closed and open canopy treatment plots. In all cases where there was a significant difference for the tested variable, the partial canopy treatment plots had the lowest mean value. The soil pH values for the irrigated soils ranged between 7.4-7.6 and the base saturation was 100 on the canopy treatment plots.

Discussion

The ability of the Pennsylvania State University to irrigate its treated municipal wastewater year-round depended on the existence a forest community. However, the future of the forest community on State Gamelands 176 (Toftrees) was not certain. The yearlong application of 5 cm of treated wastewater to the existing forest community had altered the structure of the forest community, replacing woody species in the understory with herbaceous vegetation. The forest community has entered the gap-phase stage of stand development, but few tree stems have been established to replace the overstory. The low stem density of tree seedlings inventoried across all treatments indicated that a herbaceous community eventually would replace these stands.

The negligible number of tree stems inventoried on all treatments indicated that there was a failure in the natural regeneration process (seed production, storage, and germination, and seedling establishment). The necessary conditions for seedling establishment were provided for most tree species found in the overstory. For example, the optimal conditions for quaking aspen germination were full

sunlight and bare mineral seedbed (Perala 1990), which were represented on the open canopy/ herbicide subplots. However, no quaking aspen seedlings were inventoried as natural regeneration. The germination requirements for black cherry were also provided (partial or closed canopy and undisturbed leaf litter) (Marquis 1990) and a few stems were inventoried, but none were established seedlings. Acorns of oak species have greater rates of germination when they have been stored in the soil or covered with a layer of litter (Auchmoody et al. 1994, Lhotka and Zaczek 2002). These conditions were established and again a few stems were inventoried, but none were established seedlings. The exact cause of the failure of natural regeneration has not yet been determined. Removal of the canopy and herbaceous understory did not produce natural regeneration of tree species in sufficient numbers.

The influence of canopy gaps on natural regeneration has been identified as an important mechanism in the regeneration process, as gaps provide increased light, water, and nutrients to stems in the understory. However, some researchers have demonstrated that other factors in addition to canopy gaps influence the regeneration process. Breckage et al. (2000) reported canopy gaps (0.03 ha) made in a southern Appalachian forest did not promote seedlings recruitment, and that disturbance to the forest floor and the presence of understory vegetation were more important factors for seedling recruitment.

The size of canopy gaps has also been investigated. Hibbs (1982) reported larger canopy openings in a hemlock-hardwood forest in Massachusetts favored shade intolerants, whereas medium openings favored shade tolerant species. The results of this study did not demonstrate any differences between the open and partially open canopy treatments for the recruitment of species dependent on shade tolerance.

Planted seedlings of certain species survived and grew under the irrigated conditions. Differences in planted seedling survival among species were attributed to two factors: tolerance of shade and quality of planting stock. Quaking aspen and river birch were species intolerant of shade and, therefore, had low survival on the closed and partial canopy treatments. Conversely red maple, black gum, and Norway spruce were classified as either intermediate or tolerant of shade, and had higher survival rates on all canopy treatment plots. The low survival of both quaking aspen and river birch seedlings was also attributed to inferior planting stock of these species. Seedlings of quaking aspen had very small aboveground stems and the root systems of the river birch were insufficient to support their large aboveground stems. Survival of quaking aspen was similar to previous studies.

Height growth of red maple, black gum, and Norway spruce seedlings did not respond to the canopy treatments in the first growing season. These seedlings depended on their root-reserves and were unresponsive to canopy treatment. In comparison, the height growth of quaking aspen and river birch seedlings was immediately responsive to the high light conditions on the open canopy treatment plots. By the end of the second growing season, all planted tree species had their greatest height growth on the open canopy treatment plots, indicating that full-sunlight was an important factor in seedling growth. Two-year height growth of quaking aspen seedlings was similar to other studies that I have completed on the irrigated forest communities (Ahlsvede et al. 1999, Kelso 1999).

The soil analysis of the irrigated forest community was consistent with what other researchers have reported (Larrick and Bowersox 1999). The soil pH reflected the high amount of the base cations present in the irrigated soil. Positively charged base cations replaced hydrogen ions on binding sites in the soil matrix and increased the soil pH. The elevated values of the soil nutrients on the irrigated forest may be the cause for the failure of natural regeneration. The tree species on the irrigated forest communities have been adapted to low soil nutrients and the irrigated conditions were not suitable for the regeneration process for these species.

Creation of small canopy openings in the overstory and application of herbicide did not establish natural regeneration in adequate numbers to produce a fully-stocked forest community. The addition of seedlings from artificial regeneration can be used to supplement natural regeneration. Open canopy treatment plots would not only promote the greatest height growth of planted seedlings, but also aboveground herbaceous biomass production. A partial canopy removal treatment would temper the

growth of both artificial regeneration and aboveground herbaceous community. Application of herbicides gives managers the opportunity to create open canopy conditions, which allow maximum growth for all planted species, and at the same time reduce the herbaceous vegetation.

Winter irrigation of the forest community has become imperative to wastewater application. However, winter application has the negative effect of causing damage to small diameter stems from ice accumulation. A management goal has to include protection of any young, woody stems from winter irrigation when the temperatures are below freezing until they have reached a height and diameter to withstand the accumulation of ice.

An important implication of this research was the response of an upland forest community to human disturbance, specifically the addition of highly nutritious wastewater over a short period of time. Future research may be directed at investigating the response of species to changes in the environmental variables (precipitation, edaphic conditions) under which they evolved. Some scientists have predicted that global climate change will affect many ecological regions in the future. The upland forest community on the study site, which had adapted to the climate of central Pennsylvania for centuries, was changed over a short period of time. The response of the forest community has not been positive. Regeneration has not been occurring on the irrigated forest community and herbaceous vegetation has become dominant in the understory. Artificial regeneration continues to be an alternative to establishing a forest community on State Gamelands 176 (Toftrees).

Acknowledgments

We would like to thank the Pennsylvania State University Office of Physical Plant for funding this research, the Pennsylvania Game Commission for their support, and all of the field assistants who helped on this research project.

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FOREST VEGETATION IN HAMILTON COUNTY, OHIO: A CLUSTER ANALYSIS AND ORDINATION STUDY

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ABSTRACT. —Twenty mature, relatively undisturbed forests in Hamilton County, Ohio were systematically sampled and subjected to cluster analysis. Based on those results eight forest types were identified. These included: silver maple-cottonwood-green ash or black willow on alluvial floodplains; pin oak-red maple in depressions of floodplains; oak-hickory on well-drained slopes >25%; oak-maple-ash on dissected till plains or ridge and slope complexes; mixed mesophytic with tulip poplar on well-drained till plains; mixed mesophytic without tulip poplar on loess or colluvium; maple-ash-beech on residuum, colluvium or well-drained till plains of Illinoian or Wisconsin origin; and beech-maple on lacustrine and loess deposits. Geology, especially the glacial history, soils, topography, and disturbance history appear to influence tree species community patterns.

Introduction

Dr. E. Lucy Braun began her classic studies of forest ecology in southwestern Ohio, in particular, Hamilton County or the Cincinnati Region (Braun 1916, 1936, 1950). Geologically, Hamilton County is at or near the terminus of the Nebraskan, Kansan, Illinoian, and Wisconsin glacial advances (Ray 1974). Phytogeographically, the county is in Braun's (1950) Western Mesophytic Forest Region, but near the junction of her Mixed Mesophytic, Oak-Hickory and Beech-Maple Forest Regions or Kuchler's (1964) Oak-Hickory, Mixed Mesophytic, and Beech-Maple Forests.

Braun (1916) and Diehl (1933) considered topography and soils as two of the most important factors in determining forest types and distribution in Hamilton County. However, bedrock geology and the glacial deposits may have a major influence on local vegetation patterns (Braun 1950, Forsyth 1970).

The majority of the land area of Hamilton County was a part of the original Symmes Purchase of 1788. Following European settlement, changes in the landscape began to appear, first as a result of land clearing and agriculture and later because of urban expansion. Existing forests were reduced in size and fragmented as a series of isolated islands on the landscape. Today, forest remnants occur in parks, nature preserves, and old estates where they have received protection. In recent times some of these remnants have been the basis of ecological studies by Hoyer *et al.* (1978), Bryant (1987) and Swanson and Vankat (2000). In a similar way, Levenson (1980, 1981) sampled the parks and woodlots of metropolitan Milwaukee, Wisconsin as biogeographic islands within an agro-urban setting.

The primary goals of the present study were to continue to characterize the mature forest remnants in Hamilton County, Ohio (Bryant 1987) as part of an evaluation of urban forests and to determine the nature, patterns and environmental relationships of the remnants.

The Setting

Hamilton County is located in the extreme southwestern corner of Ohio. The 107,350 ha county is bounded on the west by southeastern Indiana and on the south by the Ohio River and adjacent northern Kentucky. Fenneman (1916) placed Hamilton County in the Till Plains section of the Central Lowland Physiographic Province. However, he found this area to be less typical of the Till Plains of Ohio and Indiana because of its proximity to important drainage lines. Brockman (1998) included portions of Hamilton County in the Outer Bluegrass Region of the Interior Low Plateau and the Illinoian Till Plain of the Central Lowland Province.

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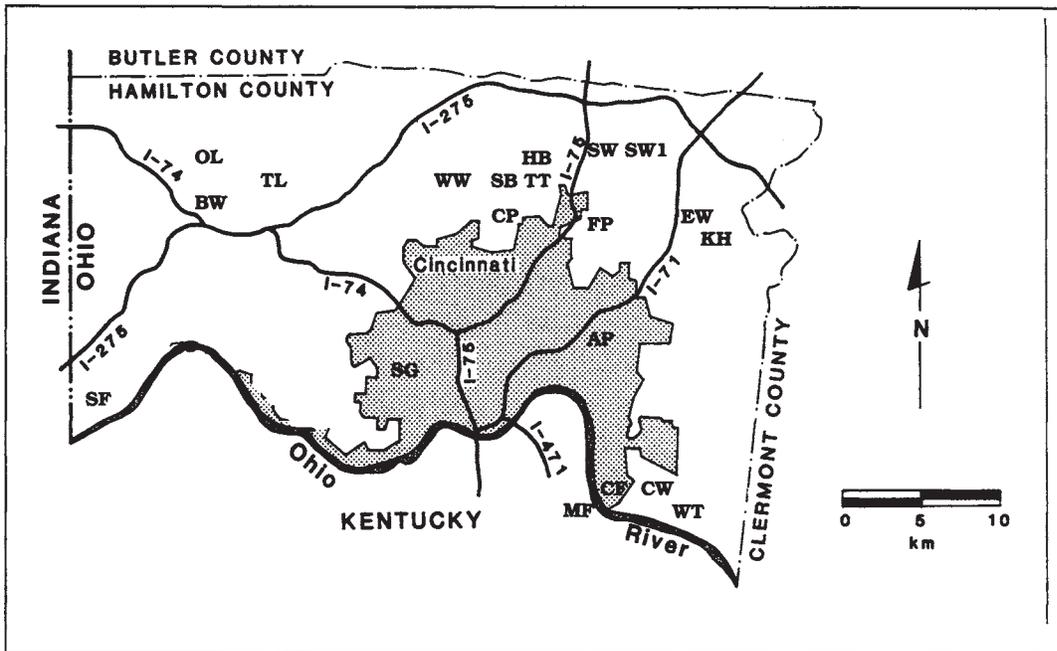


Figure 1.—The location of study sites in Hamilton County, Ohio. Letters indicate the study sites sampled except for Melbourne Forest which is in Campbell County, Kentucky. See Table 1 for abbreviations.

The local physiographic features of importance are uplands, slopes, floodplains, and terraces (Braun 1916, Diehl 1933). Braun equated these features to the soils of the area. Lerch *et al.* (1982) listed prominent topographic features of the county as gently rolling glacial uplands, steep hillsides along the major streams, extensive glacial river terraces and outwash plains, and floodplains. The topographic relief ranges from 139 m to 293 m. Stream dissection has produced many steep-sided valleys on some of the glacial deposits in the county where underlying Ordovician bedrock has been exposed (Forsyth 1970). Soils in the county reflect the glacial and erosional history of southwestern Ohio (Lerch *et al.* 1982).

The climate of Hamilton County is of the continental type with cold winters and hot summers. Annual precipitation is about 101.6 cm with over half of that falling during the growing season.

Description of Study Sites

A total of nineteen sites were selected for sampling from throughout the Hamilton County and one in Campbell County, Kentucky across the Ohio River from southeastern Hamilton County. The sites were selected to provide a representative sample of forests, geology, and soils of the region (fig. 1). Eleven sites were located in Hamilton County Parks and three were in Cincinnati City Parks. The remaining sites located in a Cincinnati Recreation Commission Preserves, in a cemetery, on an older private estate, one was a University preserve, one on the property of the Cincinnati Water Works, and one was on a convent grounds. Many other sites were visited, but were not sampled for various reasons. Mt. Airy Forest, a park of over 400 ha near the center of Cincinnati, was not sampled because it was formerly pasture land that had been reforested by plantings (Braun and Jones 1926). Other sites were generally too small to obtain a random sample or too disturbed.

The forest of Ault Park (AP), California Woods Nature Preserve (CW), Caldwell Park (CP), Winton Woods (WW), Bowles Woods (BW), and Melbourne Forest (MF) were old-growth forests previously described by Bryant (1987). Emery Woods (EW) was thoroughly described by Swanson and Vankat (2000).

California Floodplain (CF) is located on the Ohio River Floodplain. Soils are Genesee-Urban Land complex and are loamy alluvium.

Shawnee Floodplain (SF) has been described as a back swamp area of Shawnee Lookout Park. Jules silt loam on the site is deep, nearly level and well-drained (Lerch *et al.* 1982).

Oak Leaf (OL) is a forest in the Miami-Whitewater Forest. The Bonnell silt loam covers steep (25 – 35%) slopes. These well-drained slopes border the Illinoian till plain.

Timber Lakes (TL) is also located in Miami-Whitewater. Soils are primarily Eden silty clay loam on 25 – 40% slopes. Cincinnati silt loam is a loessial soil over Illinoian till (Lerch *et al.* 1982).

Spring Grove (SG) is within Spring Grove Cemetery where it has been preserved since 1880 (Linder 1995). Soils in the ravine include Urban terraces (Lerch *et al.* 1982).

Kroger Hill (KH) is found on well-drained Illinoian till plain where the rolling land is pitted with sinkholes. Soils are Rossmoyne silt loams which are loess over underlying Illinoian glacial till.

Trillium Trails (TT) is within Winton Woods Park. Soils are residual on hillsides and include the Eden silty clay loam which formed from interbedded limestones and soft calcareous shales.

Sharon Woods (SW) is a gorge area of Sharon Woods Park. This area is part of the dissected Wisconsin till plain. The soil complex includes Miami silt loam, Eden silt loam, and Russell-Urban Land.

Sharon Woods (SW1) occurs within the Park just north of the gorge area. This forest is on Wisconsin till plain with little or no dissection. Soil is the Miami-Hennepin.

Hazelwood Botanical (HB) is within the Hazelwood Botanical Preserve managed by the University of Cincinnati. This forest was initially studied by Segelken in 1929. This woods is located on the Illinoian till plain of northern Hamilton County. Soils include the Rossmoyne silt loam. Not long after this forest was systematically sampled, it was destroyed by a tornado.

Spring Beauty Dell (SB) is located with Winton Woods Park. This forest is an old slack water terrace and lacustrine sediments from the Markland silty clay loam (Lerch *et al.* 1982).

French Park (FP) is the site of a former Girl Scout camp. Colluvium covers the lower slopes of the hillsides. This is Pate silty clay loam. Bonnell silt loam is also present. This is a loessial soil over underlying Illinoian glacial till (Lerch *et al.* 1982).

Withrow Woods (WT) is located off 5-mile Road in southeastern Hamilton County. This includes a deep ravine and was part of an old glacial terrace of Illinoian age (Fenneman 1916). Upland soils include the Cincinnati silt loam and the Rossmoyne silt loams Eden soils are prominent on the steep slopes.

Methods

All forests sampled exceeded the minimum size of 3.8 ha that Levenson (1981) considered to be the smallest size at which a mature, mesic forest can perpetuate its interior conditions while sustaining limited random perturbations (Loucks 1980). All sites were sampled in 0.04 ha circular plots spaced at 30 meter intervals throughout the forest interior. All trees ≥ 10 cm at diameter-breast-height (DBH) were measured in each plot. The number of plots taken per site depended on the size of the forest and/or its homogeneity or heterogeneity. The number of plots sampled/stand ranged from 10 to 45.

Data for each forest were analyzed to relative frequency, relative density, and relative dominance or relative basal area. These values were summed to produce an importance value for each species

Table 1.—Tree density (trees/ha), basal area (m²/ha), species diversity (H'), and the number of tree species in sample for forests in Hamilton County, Ohio

Forest	Trees/ha	m ² /ha	H'	Species Richness or Number of Species
Ault Park (AP) ¹	313	34.27	2.83	20
Bowles Woods (BW) ¹	279	32.19	1.94	15
Caldwell Park (CP) ¹	199	26.07	3.35	23
California Floodplain (CF)	175	43.92	0.99	6
California Woods (CW) ¹	324	39.16	3.11	20
Emery Woods (EW) ²	401	42.60	3.68	32
French Park (FP)	287	29.93	2.51	15
Hazelwood Botanical Preserve (HB)	191	28.28	1.99	12
Kroger Hills (KH)	251	31.63	2.01	19
Melbourne Forest (MF) ¹	396	30.19	2.98	17
Oak Leaf (OL)	274	25.71	1.85	10
Sharon Woods Gorge (SW)	294	31.85	2.61	19
Sharon Woods Upland (SW1)	294	27.99	1.78	11
Shawnee Floodplain (SF)	435	16.34	1.00	4
Spring Beauty Dell (SB)	340	28.87	1.82	15
Spring Grove (SG)	314	29.12	2.48	16
Timber Lake (TL)	299	26.77	2.56	15
Trillium Trails (TT)	277	31.01	1.96	12
Winton Woods (WW) ¹	240	28.85	2.18	15
Withrow Woods (WT)	306	37.38	2.97	23

¹After Bryant (1987)

²After Swanson and Vankat (2000)

(Appendix 1). Because of the presence of the invasive Amur honeysuckle (*Lonicera maackii*) throughout the parks (Luken and Thieret 1996), shrubs, saplings and seedlings were not included in this survey. Density (trees/ha), basal area (m²/ha) and species diversity (Shannon Index or H') were determined for each forest.

Stand relationships were summarized using both the Unweighted Pair Group Mean Average (UPGMA) clustering technique (Kovach 1998) and the Bray-Curtis ordination procedure (McCune and Medford 1999).

Results

There was a total of 47 tree species in the forests samples in Hamilton County, Ohio and environs. Only two species, *Ailanthus altissima* (tree-of-heaven) and *Maclura pomifera* (Osage orange) were not native to the area. We interpreted this to indicate that the forest samples, at least at the tree level, were not greatly disturbed. Tree density, basal area, species diversity, and tree species richness are shown in Table 1. All upland sites sampled fit within the ranges for tree density (trees/ha) and basal area (m²/ha) reported by Parker (1989) and Martin 1992) for mature mesic forests.

Twelve species of trees were present \geq 50% of the stands. American beech (*Fagus grandifolia*) and white ash (*Fraxinus americana*) were the most widely distributed species, occurring in 90% of the forests sampled. Sugar maple (*Acer saccharum*), red oak (*Quercus rubra*), and black cherry (*Prunus serotina*) occurred in 85% [17 of 20] of the stands; slippery elm (*Ulmus rubra*), 80%; bitternut hickory (*Carya cordiformis*), 70%; black walnut (*Juglans nigra*) and hackberry (*Celtis occidentalis*), 65%; shagbark hickory (*C. ovata*) and blue ash (*F. quadrangulata*), 55%; and white oak (*Q. alba*), 50%. These species tend to be indicative of the mesic nature of the county and most stands. A number of other species are more site-specific and restricted in their distribution.

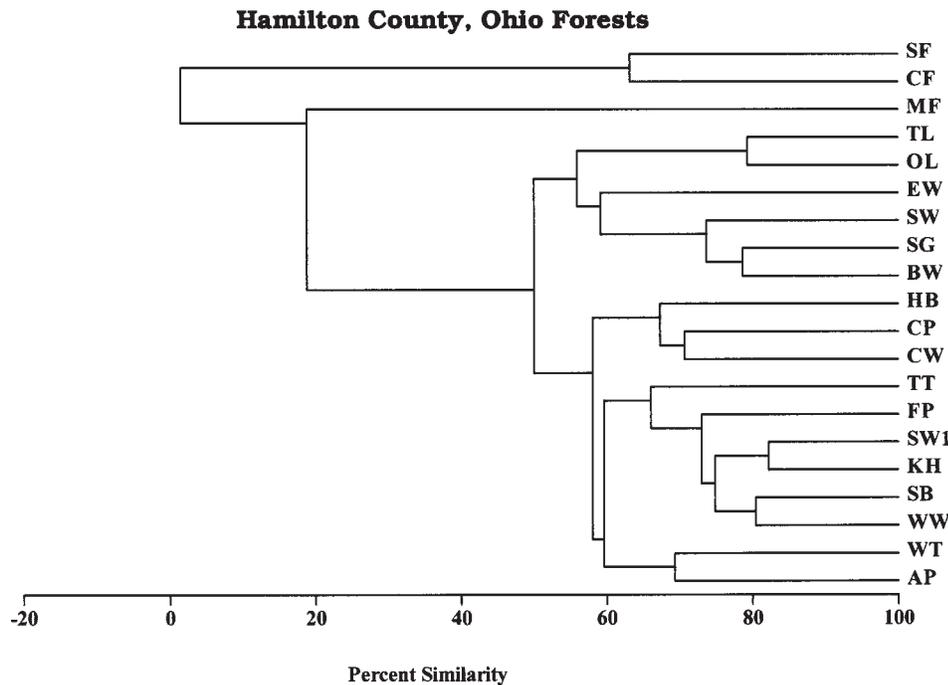


Figure 2.—Cluster analysis dendrogram of the forest sampled in Hamilton County, Ohio.

Twenty-four species accounted for $\geq 5\%$ of the stand basal area in at least one forest. Sugar maple ranked first with 85% [17 of 20 stands] followed by white ash at 60%. American beech and red oak were next with 45%. Tulip poplar (*Liriodendron tulipifera*) at 25% was the only other species to account for as much as 5% of the total basal area in four or more of the forests. This suggested differences in site quality across the county.

Based on the cluster analysis (fig. 2), eight groupings were recognized; however, the mesic associations showed overlap and were the most difficult to distinguish cleanly. Pin Oak-Red Maple (MF) occupied depressions of the Ohio River floodplain and the Silver Maple-Cottonwood-Green Ash or Black Willow (CF and SF) were on the larger floodplains. Mixed Mesophytic forests with the indicator species of this type, *Aesculus octandra* (Yellow Buckeye) and *Tilia heterophylla* (White Basswood) (Braun 1950) sorted out. However, there appeared to be two types of mixed mesophytic forests: those with *Liriodendron tulipifera* (Tulip Poplar) (CW, CP, HB) and those in which Tulip Poplar is rare or absent (WT, AP). Oak-Hickory forests (OL, TL) were at the drier end of the moisture gradient. Beech-Maple forests (WW, SB) also represented a distinct type within the mesic grouping. This forest type was mainly on sites underlain with a fragipan. Oak-Ash-Maple forests (SW, SG, BW, EW) and Maple-Ash-Beech forests (FP, KH, TT, SW1) were intermingled with the Mixed Mesophytic and Beech-Maple assemblages and did not separate out easily. For that reason, we considered the latter two species assemblages as Western Mesophytic after Braun (1950) and Gordon (1966, 1969).

On the ordination (fig. 3), Axis 1 appeared to be a moisture gradient with the floodplain forests and depression forest at the wet end and the Oak-Hickory) at the drier end. All the other forests were intermediate across the moisture gradient. Axis 2 was interpreted to represent a topographic gradient from depressions to floodplains to rolling flats to ravines to slopes and steep slopes. Based on this interpretation, the forests tend to sort out as Depression forests of Pin Oak-Red Maple; Floodplain forests of Silver Maple-Cottonwood-Green Ash or Black Willow; Beech-Maple on the rolling flats or terraces; Mixed Mesophytic in ravines; Western Mesophytic on sites with intermediate slope and moisture conditions; and, Oak-Hickory on exposed hillsides and steeper slopes. Disturbance history of the forests undoubtedly plays a role in this interpretation.

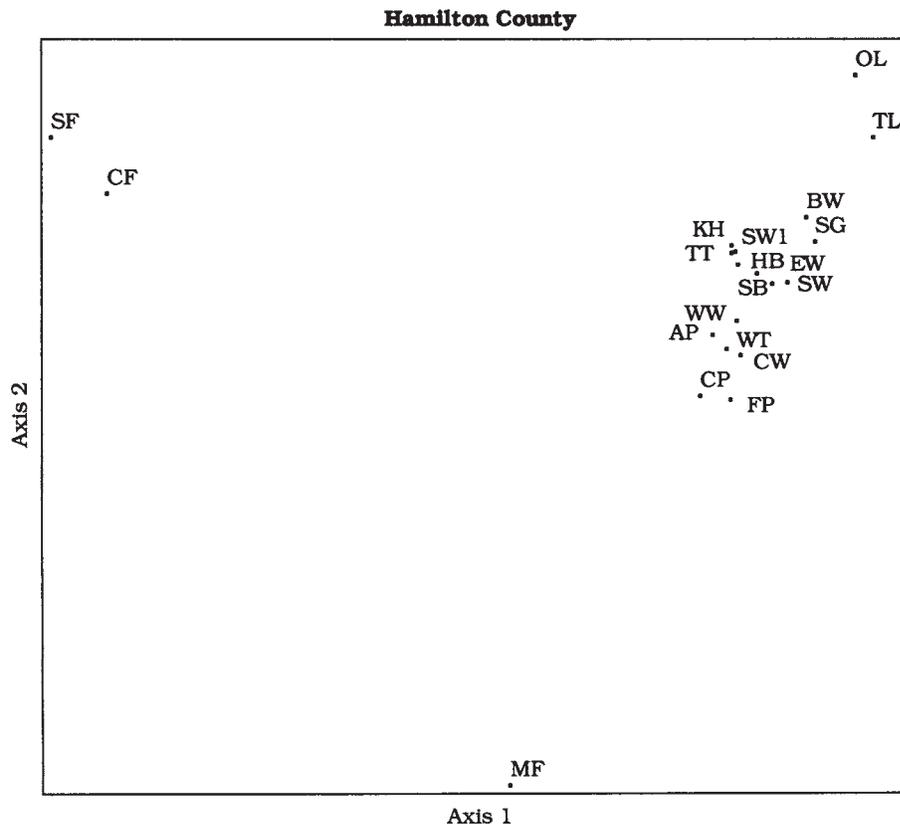


Figure 3.—Ordination of the twenty forests sampled in Hamilton County, Ohio. Axis 1 is a moisture gradient and Axis 2 is a topographic gradient.

Discussion

Even though the forests of Hamilton County, Ohio have been widely studied (i.e., Braun 1916, 1936, 1950; Bryant 1987; Diehl 1933; Hoyer et al. 1978; Swanson and Vankat 2000), they are still of great interest. Much of that interest results from the glacial history of the county plus the diversity of soil types and landforms present. Braun's (1950) inclusion of the forests of the county as part of the Western Mesophytic Forest region and the well-known urban history (Wade 1959) of Cincinnati add to that interest.

Diehl (1933) recognized six forest types in Hamilton County, Ohio. These were: Pin Oak-Red Maple; Beech; and Beech-Maple on uplands and terraces; Mixed Mesophytic and Oak-Ash-Maple on slopes; and Willow-Cottonwood-Silver Maple on recent alluvium of floodplains. Diehl (1933) correlated these forest types to soils and/or slopes. Bryant (1987) reported four types of old-growth forest remnants including Oak-Hickory, Beech-Maple, Depression Forests, and Mixed Mesophytic. A number of these forest types are site specific.

In the Cincinnati region, the forests of the floodplain are two types: the depression forest and "typical" floodplain forest (Braun 1916). She reported the depression forest to be uncommon and limited to undrained situations on the floodplain. The dominants of the depression forest include pin oak (*Quercus palustris*), red maple (*Acer rubrum*), and swamp white oak (*Q. bicolor*) (Braun 1916). She noted that the depression forests of the uplands, i.e. on the Illinoian Till Plain, are similar to those of the floodplains. Diehl (1933) recognized this upland type, however, the Pin Oak-Red Maple type is more common east of Hamilton County (Braun 1916, 1936).

Many forests on the Ohio River floodplain have also been destroyed or seriously disturbed by man. Braun (1916) listed the following species as “typical” of floodplains: willows (*Salix nigra* and *S. alba*), cottonwood (*Populus deltoids*), white elm (*Ulmus Americana*), silver maple (*Acer saccharinum*), boxelder (*A. negundo*) and sycamore (*Platanus occidentalis*). These vary in abundance in different parts of the association. Early descriptions of these floodplain sites in Cincinnati (Glazer 1999) note that, “along the [Ohio] river edge was a marshy beach bottom spotted with an assortment of sycamores, cottonwood, and water maples...This low-lying riverfront area extended approximately eight feet to the north, where a short but very steep slope marked the line of a higher plateau where beech, yellow poplar, and hickory trees receded into the distance”. Those early descriptions resemble the California floodplain. The presence of black willow (*S. nigra*) on the Shawnee Lookout floodplain indicates site a back swamp area where alluvial silt deposition is a common event (Potter 1996). The low basal area of the SF site reflects its successional nature.

Mixed mesophytic forests with the indicator species, yellow buckeye (*Aesculus octandra*) and white basswood (*Tilia heterophylla*) (Braun 1950) were found in a number of dissected areas and ravines. Braun (1950) reported that dissected areas of Illinoian glaciation and adjacent northern Bluegrass are contiguous at the east with typical mixed mesophytic forests of the Mixed Mesophytic Forest region. This dissected zone affords favorable sites for mixed mesophytic forest to extend westward as a narrow band (Braun 1950). Forsyth (1970) found that mixed mesophytic forests in Hamilton County are confined to some of the deep, steep-sided valleys produced by stream dissection. In these valleys, where a great variety of moisture conditions support many tree species, a tree association best described as mixed mesophytic occurs (Forsyth 1970). These associations tend to correlate with the occurrence of Kansan drift in Ohio (Forsyth 1970). Mixed mesophytic forests also occur on Kansan deposits in adjacent northern Kentucky (Bryant 1978). The presence or absence of tulip poplar (*Liriodendron tulipifera*) in these mixed mesophytic associations may be related to sand content of the soils, although this was not determined.

Beech-Maple forests have been reported for Hamilton County (Diehl 1933, Bryant 1987); however, Braun (1950) suggested that her Beech-Maple forest region was entirely within the area covered by the last or Wisconsin ice sheet, one that just entered the county. Vankat *et al.* (1975) considered Hueston Woods in southwest Ohio, but north of Hamilton County, to be at the southern extent of the Beech-Maple region. Bryant (1987) described Beech-Maple south of the terminus of the Wisconsin in Hamilton County on soils of mixed glacial origins and underlain with a fragipan. Lindsey *et al.* (1965) also reported Beech-Maple on Illinoian deposits well south of the Wisconsin glacier border in Indiana. The high importance of white ash (*Fraxinus americana*) in the Beech-Maple stands of Hamilton County is unlike those stands to the north and may be a reason that the Beech-Maple stands ordinate with western and mixed mesophytic stands.

Oak-Hickory forests were primarily in the western portion of Hamilton County, especially on slopes and hillsides where erosion had removed much of the glacial materials. White oak (*Quercus alba*), black oak (*Q. velutina*), shagbark hickory (*Carya ovata*), and pignut hickory (*C. glabra*) were predominant in the Oak-Hickory stands. Schmelz and Lindsey (1970) placed red oak (*Q. rubra*) and bitternut hickory (*C. cordiformis*) with their upland mesic group not with Oak-Hickory. We agree with that interpretation. However, sugar maple (*Acer saccharum*) was an important subcanopy member and was gaining importance in the canopy. Over much of the eastern North America, sugar maple has been found to be increasing in importance, especially where fire cycles have been halted (Loucks 1970).

Forests that we first recognized as Oak-Ash-Maple or Maple-Ash-Beech were lumped together as the Western Mesophytic type. Gordon (1966, 1969) mapped much of Hamilton County as mixed mesophytic-western type after Braun's (1950) recognition of the Western Mesophytic Forest region. She noted that the major vegetation types occurring in the region form a complex mosaic that is the result of present influences as well as past influences operative within recent enough time that their effects on vegetation have not yet been eliminated. She recognized that the underlying bedrock had a major influence on the vegetation and that was evidence of the lack of “complete climate” control.

Braun (1950) considered the Western Mesophytic Forest region to be a transitional region. Under similar environmental conditions in adjacent southeastern Indiana, Lindsey *et al.* (1965) identified a western mesophytic forest type in which *Fagus-Quercus-Acer-Carya* and *Fraxinus* showed a tendency to mix rather than segregate. Later, Schmelz and Lindsey (1970) identified the following forest types for Indiana: Beech-Maple, Oak-Hickory, Mixed Woods, Western Mesophytic, and Lowland-Depressional. In another paper, Lindsey and Schmelz (1970) listed Beech-Maple, Oak-Hickory, Lowland-Depressional and Mixed Woods for Indiana. In the latter paper, they combined the Western Mesophytic and Mixed Woods types. The transitional nature of forests of southern Indiana and also those of Hamilton County is further supported by earlier forest surveys. Bryant (1987) analyzed the 1788 Symmes Purchase survey, which includes most of present-day Hamilton County. In this paper, Bryant (1987) found that species in five genera: *Quercus* (20.12%), *Acer* (18.43%), *Carya* (14.90%), *Fraxinus* (14.13%), and *Fagus* (12.75%), accounted for over 80% of the trees recorded. Species of *Juglans*, *Aesculus*, *Ulmus* and *Celtis* accounted for another 12.75% of the trees. Additionally, Drake (1815) listed beech, white oak, sugar tree, walnut, hickory, and ash as the most numerous trees in the Miami River area of southwestern Ohio. He included other important timber trees, namely, wild cherry, yellow poplar, and blue ash. Those two early surveys add support to the mesic-transitional nature of the original forests that were recognized by Braun in 1950 and is still recognizable today.

Braun (1950) found that a mosaic of unlike climaxes tended to characterize the Western Mesophytic region similar to our findings for Hamilton County, Ohio. Braun considered the region to be a tension zone where the compensating effects of local environments permit unlike climaxes to exist close to one another. It is not possible that all of these are in equilibrium with the regional climate and that their presence is indicative of past events (Braun 1950). We agree that locally Hamilton County is a tension zone and that only where environmental conditions are distinct (site specific) do clear-cut community types occur. The strong relationship between site environment and stand composition is best observed at the ends of the various environmental gradients or determined by other specific environmental features.

Acknowledgments

This research was partially supported by a Faculty Fellowship from Saint Peter's College to MEH.

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INFLUENCES OF THE EVERGREEN UNDERSTORY LAYER ON FOREST VEGETATION COMMUNITIES OF THE CENTRAL APPALACHIAN HIGHLANDS

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ABSTRACT.—Evergreen understory communities dominated by *Rhododendron maximum* and/or *Kalmia latifolia* may exert significant controls on regeneration of overstory trees and nutrient retention in central Appalachian forests, but the regional distribution and ecological importance of these communities are poorly documented. We employed leaf-off satellite remote sensing combined with field sampling and quantitative analyses to assess patterns of *R. maximum* and *K. latifolia* distribution. We compared topographically mediated distributions of these species in the warm and dry Ridge and Valley physiographic province and on the cooler and wetter Allegheny Plateau. The spatial extent of each understory community (either *R. maximum* or *K. latifolia*) was estimated with greater than 80 percent accuracy using leaf-off Landsat data combined with topographic data. Using clustering analysis and non-metric multidimensional scaling (NMS) with plot survey data, we found that *R. maximum* and *K. latifolia* varied from each other in their species associations, and that topographic exposure, relative wetness, and intensity of gypsy moth defoliation were the chief environmental variables explaining differences in distribution between the two study areas. ANOVA demonstrated significant differences in mid-story volume of canopy tree species where evergreen understory species were prominent, indicating that these communities substantially influence overall forest structure, stand dynamics, and regeneration.

Introduction

In the eastern United States, the Appalachian highlands region contains perhaps the most extensive contiguous area of forests, since the high amount of topographic relief has precluded their large-scale transformation into agricultural or urban land uses (Robison 1960). These forest ecosystems provide a number of societal services on the regional scale (Daily et al. 1997), including the maintenance of air and water quality, sediment trapping, flood attenuation, and nutrient storage (Perry 1998, Aber et al. 2000). Moreover, on the global scale these forests also act as a carbon sink capable of attenuating climatic changes caused by increases in atmospheric carbon (Schlesinger 1977, Sedjo 1992, Currie et al. 2003). Finally, forest ecosystems are valuable for providing habitat and refuge for wildlife, and are valued for their recreational and aesthetic characteristics. The future sustainability of these societal services depends on the welfare and successful management of forest ecosystems, which in turn entails a thorough understanding of the distribution of individual species and associations of species in these forest communities.

The broad-scale relevance of the evergreen understory species rosebay rhododendron (*Rhododendron maximum*) and mountain laurel (*Kalmia latifolia*) to Appalachian forests at landscape and larger scales has not been addressed extensively in the literature, despite their wide geographic extent and prominent role in forest ecology where present. *R. maximum* (RMAX) and *K. latifolia* (KLAT) occurring in forest understories in nearly pure stands or mixed in varying proportions have been mapped on the landscape scale in the southern Appalachians, but in the central Appalachian Mountains only plot level analyses of their reproductive behavior and ecological relationships have been performed (Plocher and Carvel 1987, Cooper and McGraw 1988). Their locations are correlated with topography, with RMAX thriving in protected, mesic locations and KLAT more often located on exposed, upslope sites. The different landscape positions inhabited by these two species are characterized by differences in solar radiation, cold air drainage, soil moisture, summer and winter

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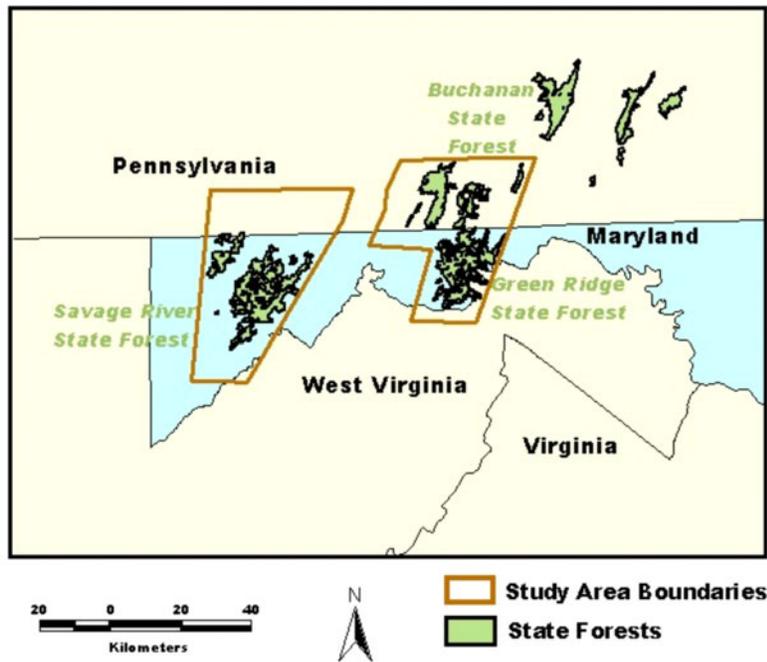


Figure 1.—Area map of present study showing extent of state forest lands.

temperature regimes, and atmospheric drying potentials (Davidson 1966, Monk et al. 1985, Lipscomb and Nilsen 1990a, 1990b, Muller 1991, Dobbs 1995). Existing works suggests that RMAX and KLAT are important component species of Appalachian forests because they:

- 1) Have the potential for slowing mineral cycling (McGinty 1972, Thomas and Grigal 1976, Monk et al. 1985), and therefore have water quality maintenance implications in the event of disturbance. RMAX has been referred to as a keystone species in this regard (Yeakley et al. 1994);
- 2) Inhibit canopy tree regeneration when present in dense stands (Minkler 1941, Phillips and Murdy 1985, Clinton *et al.* 1994, Waterman *et al.* 1995, Clinton and Vose 1996, Baker and Van Lear 1998, Nilsen *et al.* 1999, Walker *et al.* 1999, Beckage *et al.* 2000, Nilsen *et al.* 2001, Lei *et al.* 2002);
- 3) Provide forage and refuge for wildlife (Romancier 1970, Gates and Harman 1980, Thackston *et al.* 1982, Johnson *et al.* 1995, Spear 1998, USDA 2002);
- 4) Are significant components of Appalachian forests with regard to carbon sequestration and nutrient budgets (McGinty 1972, Thomas and Grigal 1976, Monk *et al.* 1985);
- 5) Enhance the beauty and hence inspirational value of Appalachian forests (Hollenhorst *et al.* 1993);
- 6) Comprise part of the cultural and ethno-botanical heritage of the Appalachian region, traditionally indicated for a number of medical uses (Sargent 1893, Uphof 1968) and used for craft items (e.g. one common name attributed to KLAT is ‘spoonwood’) (USDA 2002).

Study Areas

The spatial extent and potential ecological influence of the evergreen understory species RMAX and KLAT were assessed in two representative study areas of the central Appalachian highlands (Figure 1). One study area, encompassing the Green Ridge and Buchanan State Forests in Maryland and Pennsylvania, is located in the warmer and drier Ridge and Valley physiographic province, while the other is located on the cooler and wetter Allegheny Plateau (Savage River, Potomac, and Forbes State Forests in MD and PA). A considerable extent of these study areas is designated as public forest or game lands, and consequently experience less intensive development and logging pressure than adjacent private lands. Because of differences in topography and climate, as well as land use history, the forest community composition and structure of the forest communities in these two study areas are dissimilar

from each other, but are representative of a broad range of environmental conditions typically found in the central Appalachian region.

Ridge and Valley Study Area

The Green Ridge State Forest (GRSF) area in Maryland was heavily cut for timber, tanbark, and hoop poles between 1879 and 1910, and wildfires were common in the regenerating forest for a number of years thereafter (Mash 1996). In the early and middle 20th century, much of the area that is now the GRSF was planted in fruit (primarily apple) orchards (Mash 1996). This intense land use history within the GRSF has influenced current forest composition, which is largely devoid of KLAT and RMAX. The current forest of the ridge and valley study area is composed primarily of deciduous trees, mixed with a lesser amount of coniferous trees in the canopy. Our data show that chestnut oak (*Quercus prinus*) comprises 24 percent of the total basal area, followed by red oak (*Quercus rubra*) at 16 percent, white oak (*Quercus alba*) at 13.5 percent, scarlet oak (*Quercus coccinea*) at 7.5 percent, and black oak (*Quercus velutina*) at 6.5 percent (Chastain, unpublished data). Various pines (*Pinus virginiana*, *strobus*, *rigida*, and *pungens*) make up 9 percent of the overall basal area in this study area. These forests are largely 50 to 75 years in age (Maryland Continuous Forest Inventory, unpublished data).

Yearly average minimum and maximum temperatures for this study area are 2.3 and 15.8°C, with temperatures rarely exceeding 32°C in the summer months, but regularly dipping below -10°C in the winter (Stone and Matthews 1974). Elevation ranges from 123 to 845 meters, with steep northeast-southwest trending ridges in the GRSF and a larger contiguous upland area in the Martin Hill section of the BSF. The annual precipitation range was 896 – 1297 mm (averaging 1023 mm) for the period 1983-1998 (Lynch, pers. comm.).

Allegheny Plateau Study Area

The Allegheny Plateau study area is considered part of the mixed mesophytic forest by Braun (1950). As with the ridge and valley, this area was largely cut in the 20th century, but has regenerated naturally in the interim. The forest canopy in this study area is primarily deciduous with localized areas dominated by eastern hemlock (*Tsuga canadensis*). Red oak (*Quercus rubra*) comprises 25 percent of the total basal area, followed by red maple (*Acer rubrum*) at 18 percent and chestnut oak (*Quercus prinus*) at 17 percent (Chastain, unpublished data). Hemlock accounts for 5.5 percent of the total basal area. American chestnut was an important component of these forests previous to its extirpation in the early to mid 20th century due to the blight caused by the fungus *Cryphonectria parasitica* (Braun 1950). The elevation range for the Allegheny Plateau is 304-986 meters, with a humid continental climate characterized by severe winters and mild summers. The annual precipitation range between 1983 and 1998 was 913 – 1490 mm, with an average of 1216 mm (Lynch, pers. comm.).

Methods

The spatial extent and distribution of RMAX- and/or KLAT-dominated evergreen understory communities in each study area was first mapped using satellite imagery combined topographic information (Chastain and Townsend in review). A forest community classification was developed from the floristic (plot) data and ordination was used to assess patterns of species composition and examine relationships with hypothesized environmental gradients. Finally, an analysis of variance (ANOVA) was performed to examine differences in forest vertical structure between stands with an evergreen understory layer and those without. Hypotheses tested during these analyses include:

- 1) Multispectral remote sensing image data combined with topographic information can be successfully applied to accurately map the RMAX and KLAT-dominated evergreen understory communities on the landscape and regional scales.
- 2) Forest composition and structure are related to the presence of the evergreen understory layer, and these characteristics are different from forests that do not contain an evergreen understory. That is, canopy tree regeneration is inhibited by dense evergreen understories, and is manifested in a simplified structure where KLAT and RMAX are present.

- 3) Due to differences in topographic patterns, moisture and disturbance regimes, and land use histories between the two study areas, different environmental factors control the pattern of forest communities in general and evergreen understory communities specifically. Therefore, KLAT and RMAX-dominated communities should be distinct compared to other community types, and vary from each other in their distribution in ordination space between the two study areas.

Vegetation Sampling Design

The vegetation composition and structure data were obtained through the establishment of 213 vegetation survey plots between 1999 and 2002. The plot layout followed Townsend and Walsh (2001), with two crossing 60 meter transect tapes and five sample points located at the end of the transects and at their intersection. The orientation of the first transect line was pre-determined randomly. This plot design was chosen because it has the dual utility of efficiently supporting vegetation community analysis and also has been determined to be favorable for remote sensing vegetation studies (Grosenbaugh 1952, Lindsey et al. 1958, Justice and Townshend 1981). Basal area (BA) was estimated at each of the sample points using a metric factor-2 Bitterlich prism. Tree height measures (top and bottom of leaf canopy) were taken for three canopy and subcanopy trees at each sample point using a laser rangefinder. Heights (top and bottom of leaf canopy) were noted for all shrub/sapling strata species present with greater than 15 percent coverage. Cover estimates were made for the canopy, subcanopy, shrub/sapling, and herb layers as a whole and by species. Finally, leaf area index (LAI) was estimated at these plots using hemispherical photography acquired just above the height of the evergreen understory vegetation where present and at a height of 1 meter where no evergreen understory was present. Out of the 213 vegetation plots visited, 105 are located on the Allegheny Plateau and 108 in the Ridge and Valley province. Observations obtained within these plots were found to be comparable to data acquired in over 800 Maryland continuous forest inventory (CFI) plots established in these two study areas, so the field data from which this study is based are considered representative of regional forest characteristics.

Because of the linear nature of some evergreen understory patches (i.e. along streams or ridges) and logistical concerns, plots representing the evergreen understory were limited to one 60 meter transect with three sample points - one at each end and one in the middle. So that all data points were consistent and all analyses were comparable, plots having two perpendicular transects were analyzed using only one randomly-selected transect.

Evergreen Understory Community Mapping

The maps used to represent the spatial extent and distribution of evergreen understory communities in the two study areas were developed from supervised and decision tree classifications of leaf-off (March 31, 2000) Landsat Enhanced Thematic Mapper (ETM) imagery and topographic data derived from a digital elevation model (DEM). Field samples were used as training data for the remote sensing maps. This work indicated that topographic data improved the mapping of evergreen understory communities over Landsat data alone by spatially differentiating RMAX and KLAT by their disparate ecological niches, and is described in detail by Chastain and Townsend (in press).

Forest Community Classification

A cluster analysis using Ward's minimum variance method (Lance and Williams 1967) was performed to distinguish discrete forest vegetation classes (communities) in the vegetation survey data. Dissimilarity among plots was computed using the Sorenson statistic (Bray and Curtis 1957) based on an importance value (IV), which was calculated as a sum of the relative basal area (RBA) and percent coverage of the individual species. The IV was used to positively weight the evergreen understory shrub species whose basal area was often low when computed using the Bitterlich variable plot method. Ward's method was then implemented in the SAS statistical software, producing an agglomerative hierarchical classification that could be used to distinguish groups of plots dominated by KLAT, RMAX, or neither in the subsequent analyses.

Table 1.—Descriptions, computations, and references for the various topographic indices used in the classifications performed for the Ridge and Valley and Allegheny Plateau study areas.

Name	Acronym	Computation	Reference
Transformation	BEERS	$\cos(\text{aspect} - 45) + 1$	Beers <i>et al.</i> 1966
Relative Slope Position	RSP	$\text{dist to bottom} / (\text{dist to top} + \text{dist to bottom}) * 100$	
Topographic Convergence Index	TCI	$\ln(\alpha / \tan\beta)$	Beven and Kirkby 1979
Terrain Shape Index	TSI	Z'/R where Z' = mean elevation of plot and R = plot radius	McNab 1989
Land Form Index	LFI	$\text{sum of slope observations} / (N * 100)$	McNab 1993
Terrain Relative Moisture Index	TRMI	$\text{slope position} + \text{curvature} + \text{slope angle} + \text{slope aspect}$	Parker 1982

Ordination

Nonmetric multidimensional scaling (NMS) ordination was performed using the software PC-Ord (McCune and Mefford 1999) on the species IV scores obtained for the 213 plots. The analyses were conducted using data from both study areas together and separately to determine the extent to which composition varied according to different gradients within each area. Ordination arranges species and samples along one or more dimensions so that similar species and samples are closer together and dissimilar ones are farther apart (Jongman *et al.* 1995). Ordination is often used in ecology to describe the strongest patterns or gradients of species composition. Statistically, ordination behaves as a data reduction technique that collapses large amounts of information containing many variables (*i.e.*, all the species present in a data set) into a smaller number of composite (synthetic) variables that comprise the majority of the variation in the data set. Ordination assumes that regular patterns of species co-occurrence exist, but, unlike discrete classification techniques, the data are retained in a continuous form.

Species are hypothesized to respond to a number of topographic, disturbance, or resource gradients which can be considered to control the gradients of composition described by the ordination. Correlation and regression are therefore used to relate ordination scores to environmental variables (species abundance is not used because of the large number of zero IV values found in a vegetation data set). Table 1 describes the topographic variables we analyzed with respect to the NMS ordination. In addition, the average annual precipitation (cm) between 1983 and 1998 (Lynch, *pers. comm.*) on the vegetation plots, percent rock cover, and the number of gypsy moth defoliations were also examined. All of the environmental variables examined were derived from GIS data layers except rock cover, which was observed during field visits.

Forest Structure (ANOVA)

In the southern Appalachians, it has been observed that areas containing dense evergreen understories are structurally simpler than surrounding forests (essentially two-tiered forests consisting of the tree canopy and shrub layer) due to the inhibition of tree regeneration (Hedman and Van Lear 1995, Baker and Van Lear 1998). In addition, KLAT understory communities in Massachusetts were found to favor poorer sites characterized by lower BA and leaf area (Wilson and O'Keefe 1983). In this research, ANOVA was used to test the hypothesis that differences existed between the volume of the mid-story forest strata in areas with an evergreen understory present in contrast to areas where it is absent ('control' plots). The volume of the mid-story was calculated geometrically by averaging the depth of the vegetative matter in three mid-story individuals on each of the three sample points per plot and determining its coverage from the percent coverage data estimated at each plot. This method was followed for each stratum to construct a simple model of forest structure for each plot (Figure 2) in which the canopy, subcanopy, and shrub/sapling layer are represented by volumetric shapes. Similarly,

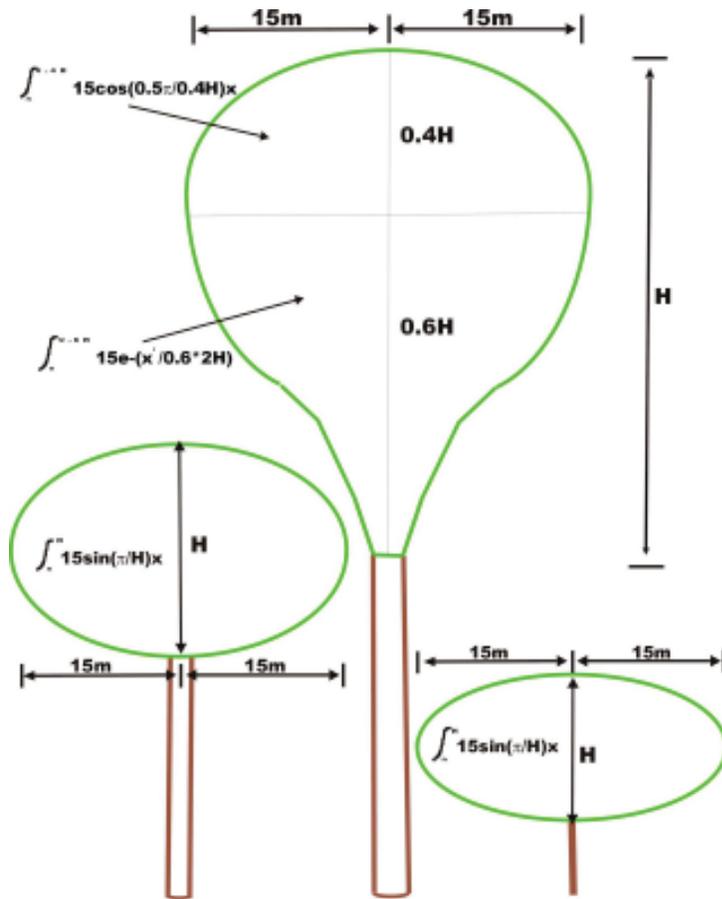


Figure 2.—Schematic drawing of the simplified three-strata model of forest structure as developed for this research.

ANOVA was used to test whether the presence of KLAT or RMAX was associated with lower BA and LAI in these two central Appalachian study areas.

Results

Spatial Extent of the Evergreen Understory Community

As reported in Chastain and Townsend (in review), the forest maps created from the Landsat ETM and topographic data classifications (Figure 3) achieved overall accuracies of 87.1 percent in the Ridge and Valley study area and 82.9 percent on the Allegheny Plateau. Overall accuracy was improved in the Plateau region by incorporating topographic information into the classification process, but inclusion of topographic information increased confusion among winter-green vegetation types in the Ridge and Valley study area because of the intense land-use history in this area (Chastain and Townsend in review). The evergreen understory made up a much greater proportion of the forested area

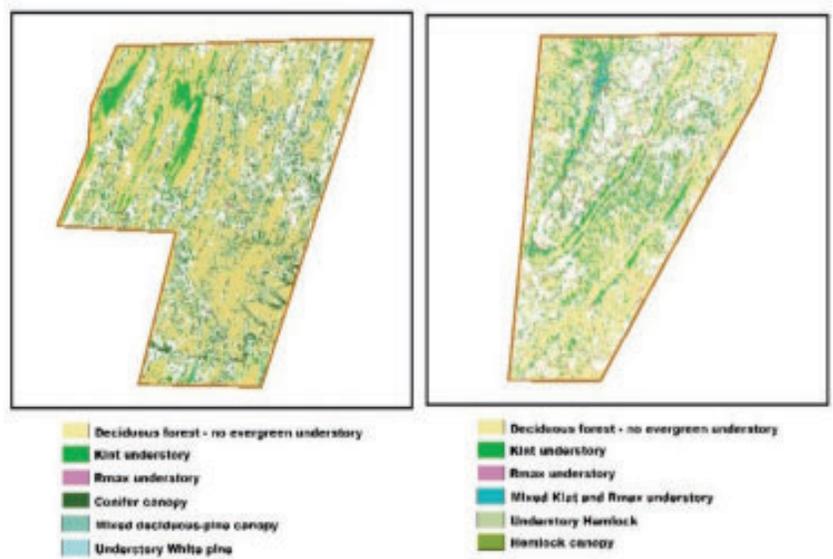


Figure 3.—Maps resulting from the Landsat ETM and topographic data classifications of the Ridge and Valley (left) and Allegheny Plateau (right) study areas.

Table 2.—Results from the hierarchical forest community classification.

Group	Type	Number of Plots		
		Total	Ridge/Valley	Plateau
Oak Forests	chestnut oak dominated	22	17	5
	white oak dominated	8	8	0
	red oak dominated	3	0	3
	mixed oak	20	13	7
Dry Oak Forests		22	22	0
Red Maple/Oak Forests	red maple with red oak	19	6	13
	red maple with chestnut oak	8	0	8
Conifer Forests		6	6	0
Sugar Maple Forests		21	7	14
Kalmia Understory Forests	KLAT with chestnut oak canopy	37	26	11
	KLAT with maple-oak canopy	18	0	18
	KLAT-RMAX with maple/oak canopy	11	0	11
	KLAT w/ red oak canopy	4	0	4
Rhododendron Understory Forests		10	3	7
Hemlock Forests		4	0	4
Total Plots		213	108	105
Evergreen Understory Plots		80		

of the Plateau study area (26.6 percent) compared to the Ridge and Valley study area (6.1 percent) (Figure 3). KLAT-dominated communities were for the most part the only evergreen community present in the Ridge and Valley province, possibly because RMAX's environmental niche is severely constrained by the presence of limestone bedrock in the moister valley locations of this physiographic province, and RMAX seedlings are inhibited by the presence of calcium (Romancier 1970). As a result, RMAX occurs only in steep, narrow stream valleys cut into the sandstone and siltstone ridges in this study area.

Forest Community Classification

The clustering analysis identified 8 major species associations, and a larger number of types based on variations in dominant and overstory species (Table 2). Oak species were dominant or co-dominant in the canopy in 172 out of the 213 plots (with and without evergreen understory layer presence). All 22 of the plots that were classified as 'dry oak' (containing *Q. coccinea* and *Q. velutina*) occurred in the Ridge and Valley province. A total of 80 evergreen understory plots were classified from the plot data, with 29 in the Ridge and Valley, and 51 in the Allegheny Plateau study area. These community classifications were used in subsequent analysis to assess their separability in ordination species-space.

Ordination Results

Nonmetric multidimensional scaling (NMS) showed clear separation among species and vegetation classes (Figure 4). In addition, joint plots with the environmental variables (where the length and direction of the environmental vector is scaled to the strength and direction of the correlation of that variable with the ordination axis) indicate discernable and readily interpretable relationships between species and environmental gradients. First, patterns of species assemblages were distinct between the two physiographic provinces (Figure 4A). In addition, KLAT and RMAX dominated plots are clearly distinct from other vegetation types and themselves in the ordination (Figure 4B). Notably, the Ridge and Valley KLAT plots and the Plateau KLAT plots inhabit distinct space on this plot, with KLAT presence strongly related to gypsy moth defoliation (measured as the cumulative number of years

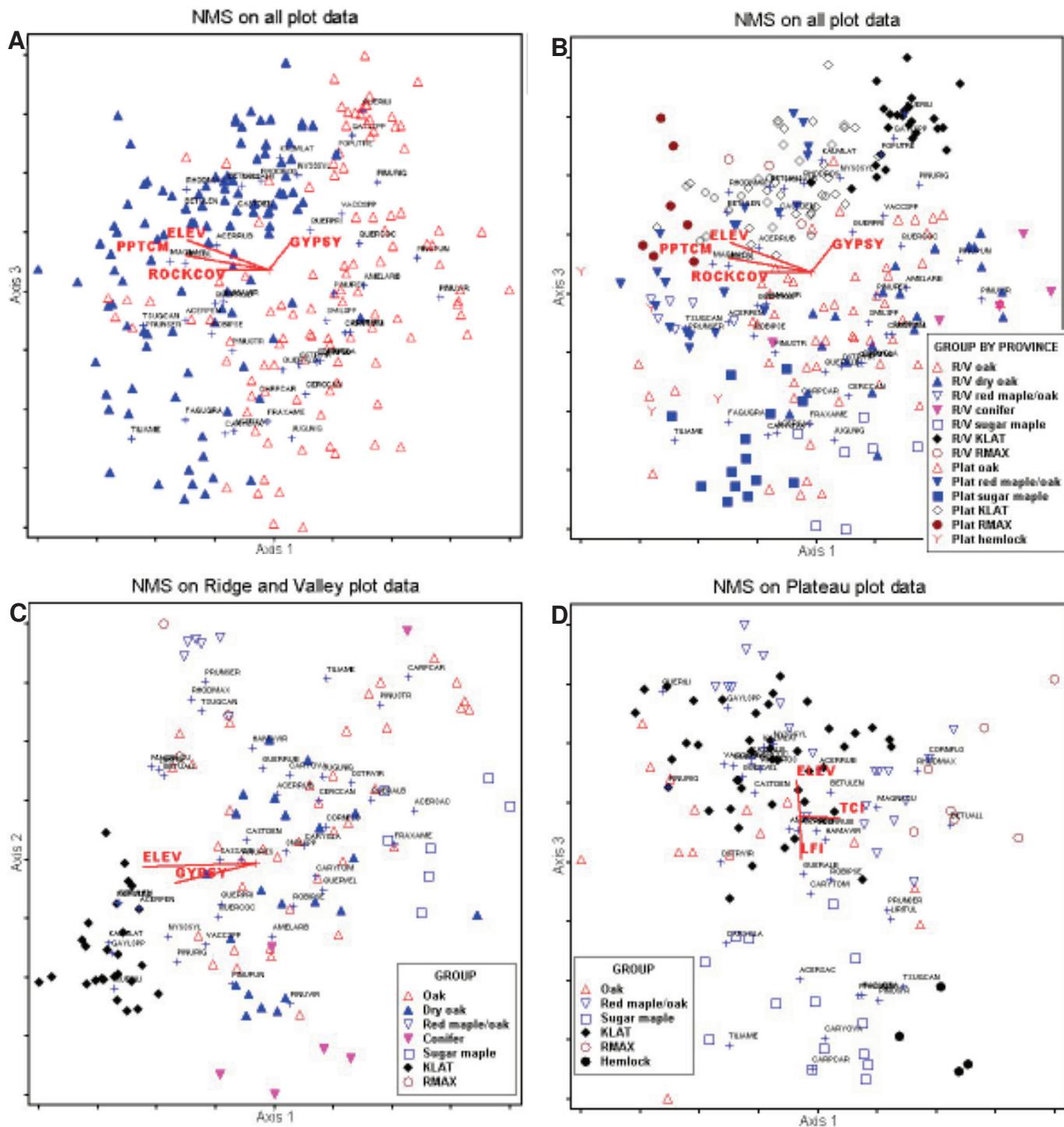


Figure 4.—Joint plots of four separate NMS ordinations of species IV scores and environmental variables. Ordination results of data from all 213 plots symbolized to represent the two study areas are shown in A), with the Plateau plots symbolized as dark triangles, and the Ridge and Valley plots as light triangles. All 213 plots symbolized to indicate the group identified in the community classification (see Table 2) are shown in B), ordination results from the Ridge and Valley plot data are shown in C), and ordination results from Allegheny Plateau plot data are shown in D).

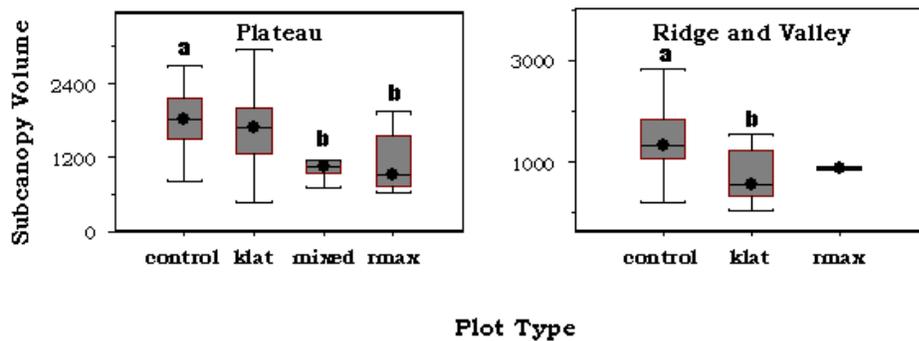


Figure 5.—Box and whisker plots of subcanopy foliar volume for the Allegheny Plateau and Ridge and Valley study areas. The y-axis is a relative measure of cubic meters per 900 meters square, which is the area of a 30-m Landsat TM pixel. Different letters indicate significant differences between classes.

defoliated) in the Ridge and Valley. Examining the Ridge and Valley alone (Figure 4C), the gypsy moth disturbance gradient appears strongly related to the KLAT plots (R with ordination axis 3 = -0.52). The NMS ordination on the Plateau plot data alone suggests the overall importance of topographic gradients in this study area (Figure 4D), as the history of gypsy moth disturbance and land use is less intense. The TCI (wetness index) gradient increases in the direction of the plots containing the moisture-loving RMAX, and the slope gradient decreases in that direction. Similarly, LFI (a gradient of exposed to protected sites) decreases in the direction of the drier KLAT plots. Pearson correlations for TCI with ordination axis 3 = -0.41 and for LFI with axis 1 = -0.42.

ANOVA Results

The ANOVA results indicated that the presence of an evergreen understory was significantly related to a paucity of trees in the midstory (subcanopy) as measured by subcanopy volume in some, but not all understory types in the two study areas (Figure 5). In the Ridge and Valley province, ANOVA results indicated a significant difference in subcanopy volume by plot type ($F = 16.75$, $P < 0.0001$), but while Tukey's multiple comparison test revealed that KLAT plots had a significantly lower subcanopy volume than 'Control' (no evergreen understory) plots, RMAX sites were not significantly different from either the KLAT or 'control' plots. However, the RMAX plots in this province were limited in spatial extent and concentrated in narrow riparian areas where patch width was insufficient to exclude regeneration. ANOVA also indicated that KLAT plots had significantly lower LAI compared to the control and RMAX plots in the Ridge and Valley province ($F = 6.76$, $P = 0.0018$). Further, live BA was significantly lower on KLAT plots compared to both the control and RMAX sites ($F = 18.85$, $P < 0.0001$). Therefore, KLAT communities are demonstrably more open than is typical of other forest types in the region.

Results from the ANOVA on the Allegheny Plateau indicated that the subcanopy volume of the mixed and RMAX plots were significantly lower than the subcanopy volume in the control plots ($F = 4.9$, $P = 0.0034$). Surprisingly, the KLAT plots and control plots were not significantly different with respect to subcanopy volume (Figure 5). Although not clear from the data, KLAT communities may be patchier on the Plateau. The ANOVA results also indicated that KLAT plots had a significantly lower average LAI than RMAX plots ($F = 5.64$, $P = 0.0014$), but was not significantly lower than the average LAI on the control plots. No significant BA differences were noted between the Plateau plot types.

Discussion

The composition and structure of forests in the central Appalachian highlands have developed as a result of climate, topography, land use, and disturbance. However, disturbance and land use tend to complicate the association between vegetation patterns and the direct and resource gradients that influence forest composition. The successful mapping of the evergreen understory communities in these

two adjacent but dissimilar study areas in the central Appalachian highlands has yielded a clearer understanding of their spatial extent and pattern of occurrence. Ordination of plot data showed that indirect gradients related to environmental history were important for characterizing the distribution of KLAT and RMAX. In the Ridge and Valley, gypsy moth defoliation opened up the subcanopy light environment permitting the development of dense stands of *Kalmia*, whereas on the Plateau, climate and terrain-related gradients distinguished KLAT and RMAX from each other and forests without an evergreen understory layer. Finally, the data indicate that the evergreen understory significantly affects the structural characteristics of forest stands containing dense KLAT or RMAX. Although plot level studies performed in other locations in the Appalachians confirm this result, this research provides data to assess on a regional scale the spatial implications of understory communities on overall forest structure, stand dynamics, and regeneration. The dynamics of KLAT and RMAX are therefore a central component to central Appalachian forest communities.

The overarching goal of this research is to examine the prevalence, ecological impacts and potential future influence of dense evergreen understory coverage in the central Appalachian highland forests. Further work to address this entails the examination of the temporal dynamics of KLAT- and RMAX-dominated understory communities over a 16 year period using remote sensing imagery and assessing via simulation models the importance of these understory communities with respect to carbon sequestration and nutrient dynamics on a landscape scale. Ultimately this research will provide the basis for informed management decisions and adaptive management strategies in Appalachian forests.

Acknowledgments

The authors thank Jodie Thompson, Jack Geary, Brian Sturtevant, and others for providing field support, and Clayton Kingdon for lab support. This research was supported by EPA grant RA26598-01 to S.W. Seagle and P.A. Townsend. This paper is Scientific Series Contribution Number 3708-AL, University of Maryland Center of Environmental Science.

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TEMPORAL CHANGES IN SPRING GROUND-FLORA COMMUNITIES ACROSS RIPARIAN AREAS IN A NORTH-CENTRAL OHIO OLD-GROWTH FOREST

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ABSTRACT.—Spring ground-flora composition and structure were sampled along riparian areas in Johnson Woods State Nature Preserve, an old-growth oak-maple-beech forest in north-central Ohio. Seven transects that bisected the stream valley were surveyed during early and late spring. Ground-flora communities differed between early and late spring and on landforms during each survey. Early spring floodplains were dominated by *Floerkea proserpinacoides* and *Ranunculus septentrionalis*, while late spring floodplains were dominated by *Geum vernum* and *Leersia virginica*. Early spring uplands were dominated by *Anemone quinquefolia* and *Cardamine concatenate*, while late spring uplands were dominated by *Parthenocissus quinquefolia* and *Podophyllum peltatum*. Principal components analysis showed a complex environmental gradient from stream channel edge into the surrounding uplands strongly related to soil characteristics and landform. The variability observed between early and late spring surveys suggests the need for multiple surveys in both spring and summer in order to gain a complete record of ground-flora communities.

Riparian areas are complex ecotones between aquatic and terrestrial ecosystems creating unique gradients that influence the structure and composition of plant communities. Riparian areas fulfill several important functions including regulating nutrient and energy flow between the terrestrial and aquatic systems (Gregory et al. 1991), providing unique habitat with high species richness and diversity (Goebel et al. 2003c), offering corridors for wildlife movement (Hodges and Kremetz 1996), and creating aesthetic appeal (Kline et al. 2000). In many locations these unique systems have been destroyed and their functions disrupted. As a result, ecological, as well as economical, problems have arisen including poor water quality, stream bank erosion, and a decline in populations of flora and fauna.

Where these problems are especially severe, restoration of the riparian area is often selected as a solution. Existing methods that describe riparian restoration are vague and generic (Lee et al. 2000) leaving those who wish to restore riparian areas at a loss for specific guidance. In order to generate better information for restorationists, a reference ecosystem on which to base objectives and goals of a restoration is needed (Pickett and Parker 1994; Aronson et al. 1995). Reference ecosystems provide information on structure, composition, and underlying dynamics that drive sustainable ecosystem functions.

In Ohio, many of the riparian areas have undergone some form of disturbance from agriculture, logging, or urbanization. Therefore, locating undisturbed ecosystems to study for reference conditions is difficult. While scarce, isolated tracts of old-growth forest exist that reflect minimal anthropogenic disturbance. Many of these old-growth forests have small streams or rivers running through them with intact native riparian areas that may be studied as reference ecosystems. The limited number of riparian reference areas that have been identified and studied in Ohio focus mainly on overstory composition, while largely ignoring ground-flora composition and structure (Goebel et al. 2003b, McCarthy 2003). Studies that address ground-flora tend to focus on one survey, usually in mid-summer when spring ephemerals have disappeared. As flora is variable throughout a growing season, it is important to have a complete floristic record of a reference ecosystem (McCarthy 2003).

The objectives of this study were (1) to compare and contrast composition and structure of early and late spring ground-flora in riparian areas of an old-growth forest and (2) characterize a spring ground-flora reference condition for riparian restoration in north central Ohio.

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Study Area

This study was conducted at Johnson Woods State Nature Preserve (JWSNP), an 83 hectare old-growth forest in Wayne County, Ohio. The study area has a humid continental climate with a mean summer temperature of 20.6°C, and a mean winter temperature of -2.8°C (Bureau et al. 1984). Mean annual total precipitation is 90 cm, with sixty percent occurring between April and September, and mean snowfall amount is 91.4 cm (Bureau et al. 1984).

Johnson Woods is located on the western glaciated Allegheny Plateau in an area characterized by gently rolling ground moraine with broad flat-bottomed valleys (White 1967). Headwater streams tend to have small floodplains that rise shallowly into surrounding uplands, often lacking defined terrace or slope landforms. Soils were formed in silty lacustrine deposits resulting from Wisconsinian glacial lakebeds (Bureau et al. 1984). A Fitchville silt loam (fine-silty, mixed, mesic Aeric Ochraqualf) composes the small headwater stream valley located at JWSNP, while the uplands are composed of Sebring silt loam (fine-silty, mixed, mesic Typic Ochraqualf) and Glenford silt loam (fine-silty, mixed, mesic Aquic Hapludalf) (Bureau et al. 1984). These soils are classified as moderate to poorly drained, and both the Sebring and Fitchville soils have a seasonally high water table that allows surface ponding (Bureau et al. 1984).

The Western Glaciated Allegheny Plateau Ecoregion is characterized by beech-maple, mixed-mesophytic, and northern hardwood forests (McNab and Avers 1994). E. Lucy Braun (1950) first characterized the forest communities of Johnson Woods (formerly called Graber Woods). Wet depressions were composed of red maple-American elm communities, while flats with an increased humus layer were dominated by red maple and white oak mixed with red oak, ash, and beech. Low swells had white oak in the canopy with a beech and sugar maple understory (Braun 1950). Currently, well-drained areas are comprised of mixed-oaks and hickory in the overstory, with an increasingly shade-tolerant beech-maple understory. Low-lying areas typically have vernal pools and are comprised of mesic and wet-mesic species including red maple, buttonbush, and green ash.

Methods

Data Collection

In the spring of 2003, the spring ground-flora was sampled along the first-order headwater stream running through Johnson Woods. Two surveys were conducted, the first in early April, and the second in late May. Each time, seven transects bisecting the entire stream valley were sampled. The location of the first transect was determined randomly and each subsequent transect was spaced randomly at least 20 but no more than 40 m apart. Along each transect, the first 1 m² quadrat was located in the center of the stream channel with additional quadrats spaced 5 m apart moving perpendicular from the channel into the upland. Each half of a transect ranged in length from 25 to 40 m depending on valley width. A total of 116 quadrats were sampled during each survey.

Soil cores were taken from each quadrat location with a Giddings® probe to a depth of 30 cm. Before the core was disturbed, thickness of the A horizon (cm) was measured in the field. Soil samples were then transported to the laboratory for the remaining analyses. Soil particle fractions for sand (particle size > 0.1 mm), silt (particle size 0.1 – 0.001 mm), and clay (particle size < 0.001) were determined after air drying using the hydrometer method. Percent soil organic matter (% OM) was determined by loss on ignition and pH was measured using an electrode (Sparks et al. 1996).

At each quadrat, the landform was determined to be either floodplain or upland. Floodplain landforms were defined in this study to be a low-lying area adjacent the stream channel that floods regularly. Standard surveying methods and a clinometer were used to determine the distance of the quadrat from the center of the stream channel and its elevation above bankfull channel. All vascular species <1m tall rooted inside the quadrat were identified, and percent cover of the species was visually estimated and designated one of the following cover class codes: <1 percent (1); 1-5 percent (2); 6-10 percent (3); 11-20 percent (4); 21-40 percent (5); 41-70 percent (6); 71-100 percent (7). Species were sorted into

Table 1.—Mean values of environmental variables (± 1 SE) for landforms along first order stream at Johnson Woods State Nature Preserve, OH (*P-value<0.001).

Variable	Floodplain	Upland
Distance from Stream (DFS) (m)	8.65 (1.12)	*26.44 (1.28)
Elevation above Stream (ELEV) (m)	0.35 (0.06)	*2.66 (0.21)
pH	*5.11 (0.07)	4.59 (0.04)
Organic Matter (OM) (%)	*4.96 (0.30)	3.38 (0.10)
A Horizon Thickness (ATHICK) (cm)	10.25 (0.59)	9.22 (0.42)
Sand (%)	*26.70 (0.70)	22.93 (0.48)
Silt (%)	39.86 (1.29)	*51.35 (0.53)
Clay (%)	*33.45 (1.24)	25.73 (0.50)

functional lifeform guilds (forb, graminoid, pteridophyte, woody seedling, woody shrub, or woody vine). Nomenclature follows National PLANTS Database (USDA 2002).

Data Analysis

Mean values of the environmental data (soils, distance from stream, and elevation above stream) were determined for each landform. Mann-Whitney tests were conducted to determine differences in each factor between landforms. Principle components analysis (PCA) was implemented to analyze patterns in the environmental data distribution using CANOCO software (ter Braak and Smilauer 1997). Prior to analysis of environmental data by PCA, percent data (organic matter, sand, silt, and clay) were square root arcsine transformed to normalize variances. Pearson correlation coefficients were calculated between PCA axis scores and environmental data to correlate the PCA distribution with specific environmental variables.

Prior to analysis, each cover class code was replaced with the midpoint value of that cover class. Mean percent cover of all species and functional lifeform guilds were determined for each landform (floodplain or upland) for each survey (early or late). Richness (S =number of species), diversity (Shannon Wiener index= H'), and evenness ($E=H'/\ln(S)$) were calculated for each quadrat. Indicator Analysis (Dufrene and Legendre, 1997) was used to 1) determine if individual species cover was significantly different between floodplain or upland landforms for each survey, and 2) to indicate species typically found on each landform for each survey. Indicator Analysis is a technique that utilizes Monte-Carlo permutations and combines information on the relative abundance of a species in a particular group (i.e. landform class) and the likelihood of the species to be present in that group. Mann-Whitney tests were conducted to determine differences in mean percent cover of functional lifeform guilds between landforms for each survey, and to determine differences in mean diversity measures between landforms for each survey.

Results

Environmental characteristics

Of the 116 total quadrats sampled during each survey, 37 quadrats were classified as floodplain and 73 quadrats were classified as upland. Significant differences ($P<0.05$) between floodplain and upland landforms were found in all measured environmental variables except thickness of the A horizon (table 1). In comparison, floodplains had higher pH than uplands (5.11 ± 0.07 and 4.59 ± 0.04 , respectively) and higher percent organic matter (4.96 ± 0.30 and 3.38 ± 0.10 , respectively). While floodplains had higher percentages of sand and clay particles, uplands exhibited higher percent silt (table 1).

Environmental distribution

PCA demonstrated clear differences in environmental characteristics between floodplain and upland landforms. The first PCA axis explained 51.8 percent of the variation among quadrats and the second PCA axis explained an additional 14.9 percent (figure 1). Floodplain quadrats were positively associated with higher clay, sand, and organic matter content, as well as higher pH and thicker A

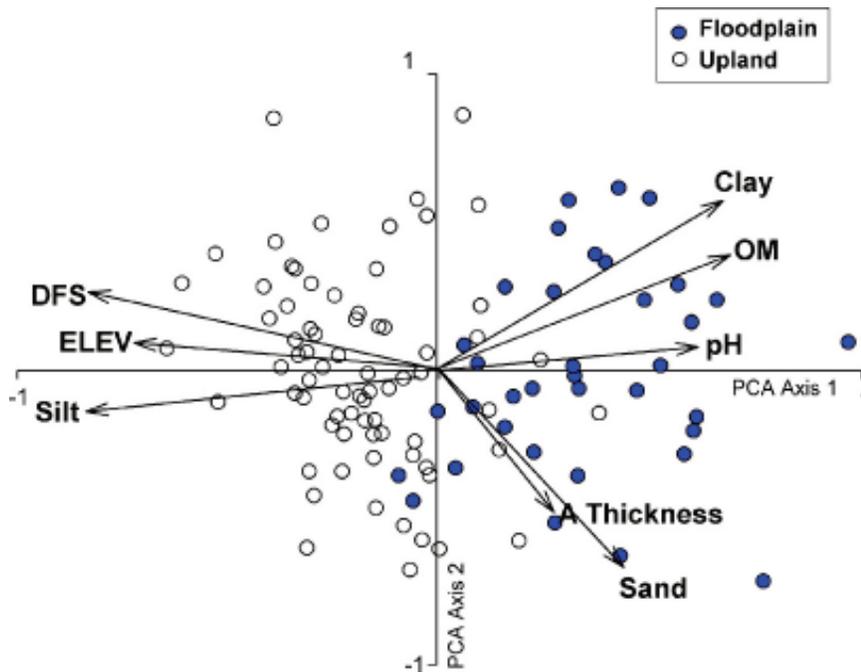


Figure 1.—Principal components analysis (PCA) showing biplot scaling of environmental data and spring 2003 ground flora sample quadrats from Johnson Woods State Nature Preserve, OH. See Table 1 for environmental variable acronyms.

Table 2.—Pearson correlation coefficients between PCA ordination axes and environmental variables (** p-value<0.001; * p-value<0.01) for Johnson Woods State Nature Preserve, OH.

	PCA1	PCA2
Eigenvalue	0.50	0.15
Cumulative variance explained (%)	50.10	65.00
Correlation Coefficient		
% Organic Matter	**0.735	*0.318
% Sand	**0.468	**−0.688
% Silt	**−0.918	−0.066
% Clay	**0.775	**0.546
Distance From Stream	**−0.800	0.155
Elevation Above Stream	**−0.769	0.081
pH	**0.655	−0.012
A Horizon Thickness	**0.378	**−0.534

horizon. Pearson correlation coefficients (table 2) indicated significant correlations ($P < 0.001$) for all variables along the first axis. The second axis was positively correlated with clay content and negatively correlated with thickness of the A horizon, sand, and organic matter content.

Ground flora composition

Indicator Analysis revealed significant differences ($P < 0.10$) in species composition between the floodplain and upland ground-flora communities for each survey (table 3). Early spring floodplains were characterized by *Floerkea proserpinacoides*, *Glechoma hederacea*, *Leersia virginica*, and *Ranunculus septentrionalis*, while the late spring floodplains were characterized by *Floerkea proserpinacoides*, *Geum vernum*, *Glechoma hederacea*, *Laportea canadensis*, *Leersia virginica*, *Lilium canadense*, *Poa alsodes*, and *Viburnum acerifolium*. Early spring uplands were characterized by *Anemone quinquefolia*, *Cardamine concatenate*, and *Podophyllum peltatum*, while late spring uplands were characterized by *Fagus grandifolia*, *Fraxinus pennsylvanica*, *Parthenocissus quinquefolia*, *Podophyllum peltatum*, *Polygonatum pubescens*, and *Prunus serotina*.

Table 3.—Mean percent cover of ground flora species on landforms for each survey at Johnson Woods State Nature Preserve (*significant (P<0.10) for landform during survey).

Species	Early Spring		Late Spring	
	Floodplain	Upland	Floodplain	Upland
<i>Acer rubrum</i>	0.00	0.04	0.08	0.06
<i>Acer saccharum</i>	0.00	0.30	0.42	1.70
<i>Aesculus sp.</i>	0.00	0.00	0.00	0.10
<i>Amphicarpa bracteata</i>	0.00	0.00	0.00	0.04
<i>Anemone quinquefolia</i>	1.16	*3.08	1.64	1.42
<i>Arisaema triphyllum</i>	0.08	0.00	2.03	1.21
<i>Aster sp.</i>	0.08	0.08	0.08	0.10
<i>Botrychium dissectum</i>	0.00	0.14	0.00	0.14
<i>Cardamine concatenata</i>	1.00	*2.30	0.00	0.19
<i>Carex albursina</i>	0.00	0.00	0.00	0.04
<i>Carex rosea</i>	0.00	0.00	0.00	0.04
<i>Carya ovata</i>	0.00	0.04	0.00	0.24
<i>Cinna arundinacea</i>	1.46	0.18	1.05	0.41
<i>Circaea quadrisulcata</i>	0.00	0.00	1.45	2.04
<i>Claytonia virginica</i>	5.64	7.78	0.18	0.83
<i>Dryopteris carthusiana</i>	0.00	0.23	0.00	0.50
<i>Dryopteris sp.</i>	0.00	0.10	0.00	0.00
<i>Erythronium americanum</i>	3.69	5.38	0.09	0.15
<i>Euonymus obovata</i>	0.42	0.10	0.42	0.04
<i>Fagus grandifolia</i>	0.00	0.53	0.00	*1.75
<i>Floerkea proserpinacoides</i>	*5.89	0.30	*4.77	0.39
<i>Fraxinus pennsylvanica</i>	0.08	0.04	0.60	*1.60
<i>Galium aparine</i>	1.01	1.37	1.82	1.66
<i>Geum canadense</i>	0.00	0.00	0.11	0.10
<i>Geum vernum</i>	1.15	0.33	*1.96	0.33
<i>Glechoma hederacea</i>	*0.55	0.20	*0.60	0.04
<i>Impatiens capensis</i>	1.23	1.27	8.11	6.92
<i>Laportea canadensis</i>	0.00	0.00	*5.42	1.36
<i>Leersia virginica</i>	*1.54	0.04	*4.03	0.14
<i>Lilium canadense</i>	0.08	0.04	*1.12	0.15
<i>Lindera benzoin</i>	0.00	0.00	0.65	0.68
<i>Lobelia cardinalis</i>	0.42	0.00	0.22	0.00
<i>Lycopodium flabelliforme</i>	0.00	0.00	0.00	0.04
<i>Mimulus ringens</i>	0.00	0.00	0.00	0.08
<i>Mitchella repens</i>	0.00	0.04	0.00	0.00
<i>Osmorhiza claytonii</i>	0.00	0.08	0.01	0.23
<i>Osmorhiza sp.</i>	0.00	0.04	0.00	0.00
<i>Ostrya virginiana</i>	0.00	0.08	0.00	0.00
<i>Parthenocissus quinquefolia</i>	0.00	0.14	1.01	*4.87
<i>Poa alsodes</i>	1.87	1.39	*3.42	1.61
<i>Podophyllum peltatum</i>	0.00	*0.25	0.00	*8.91
<i>Polygonatum pubescens</i>	0.00	0.11	0.32	*2.08
<i>Polygonum caespitosum</i>	0.00	0.00	0.85	0.41
<i>Polygonum persicaria</i>	0.08	0.04	0.38	1.20
<i>Polygonum sp.</i>	0.00	0.00	0.00	0.08
<i>Populus grandidentata</i>	0.00	0.00	0.00	0.04
<i>Prenanthes sp.</i>	0.00	0.01	0.22	0.51
<i>Prunus serotina</i>	0.00	0.36	0.16	*1.20
<i>Quercus alba</i>	0.00	0.00	0.00	0.11
<i>Quercus rubra</i>	0.00	0.00	0.00	0.18
<i>Ranunculus abortivus</i>	0.16	0.04	0.08	0.14
<i>Ranunculus septentrionalis</i>	*0.38	0.00	0.64	0.10
<i>Ribes cynosbati</i>	0.00	0.23	0.00	0.10
<i>Rosa multiflora</i>	0.42	0.23	0.42	0.42

Continued.

Table 3.—Continued.

Species	Early Spring		Late Spring	
	Floodplain	Upland	Floodplain	Upland
<i>Rubus idaeus</i>	0.00	0.00	0.00	0.10
<i>Rubus occidentalis</i>	0.00	0.04	0.00	0.10
<i>Sambucus canadensis</i>	0.16	0.22	0.30	0.70
<i>Senecio aureus</i>	0.16	0.10	0.43	0.10
<i>Smilacina recemosa</i>	0.00	0.00	0.00	0.04
<i>Smilax rotundifolia</i>	0.00	0.00	0.00	0.04
<i>Solidago sp.</i>	0.00	0.00	0.08	0.00
<i>Toxicodendron radicans</i>	0.00	0.10	3.01	3.22
<i>Trillium grandiflorum</i>	0.00	0.04	0.22	0.20
<i>Trillium sp.</i>	0.08	0.08	0.08	0.35
<i>Ulmus americana</i>	0.00	0.00	0.00	0.43
<i>Urtica dioica</i>	0.00	0.00	*0.50	0.22
<i>Viburnum acerifolium</i>	0.00	0.00	0.00	0.20
<i>Viola blanda</i>	0.00	0.04	0.00	0.00
<i>Viola pubescens</i>	0.46	0.29	0.00	0.14
<i>Viola sp.</i>	0.34	0.65	1.99	1.92

Table 4.—Diversity measures of ground flora for each survey by landform at Johnson Woods State Nature Preserve, OH. S=number of species, H'=Shannon-Wiener index, E=H'/ln(S).

	Early Spring		Late Spring	
	Floodplain	Upland	Floodplain	Upland
Richness (S)	4.0270	4.1520	5.7840	6.4810
Evenness (E)	0.8022	0.8227	0.8074	0.8672
Diversity (H')	0.9970	1.0856	1.2350	1.4642

Ground flora structure

Richness, evenness, and diversity did not differ ($P>0.10$) among landforms within the early spring or late spring survey (table 4). The late spring survey had higher richness and diversity values for both floodplains and uplands than the early spring survey (table 4). Mean percent cover of forbs differed between floodplain and upland landforms ($P<0.10$) for both the early and late spring surveys, while other functional lifeform guilds did not differ significantly ($P>0.10$) in mean percent cover between landforms for either survey (figure 2).

Discussion

Spring ground-flora communities at Johnson Woods differed both spatially and temporally. While the structure of ground-flora communities did not differ significantly between landforms, except by mean forb cover, indicator analysis showed that composition differed between landforms. Temporally, species richness was higher during the late spring survey. While some species present during the early spring survey were still present during the late spring survey, additional species emerged, changing the overall composition and structure of the ground-flora communities of each landform during the spring. Higher species richness in the late spring survey corresponded to others studies where the highest species richness during one growing season occurred in early June (Tremblay and Larocque 2001, Small and McCarthy 2001).

Based on the PCA, environmental variables appeared to explain a large portion of the variability observed in ground flora communities. One environmental variable not sampled directly, but noteworthy, is the degree of flooding on floodplain landforms during the sample periods. Many floodplain quadrats were inundated during each survey, therefore not supporting ground flora species during the

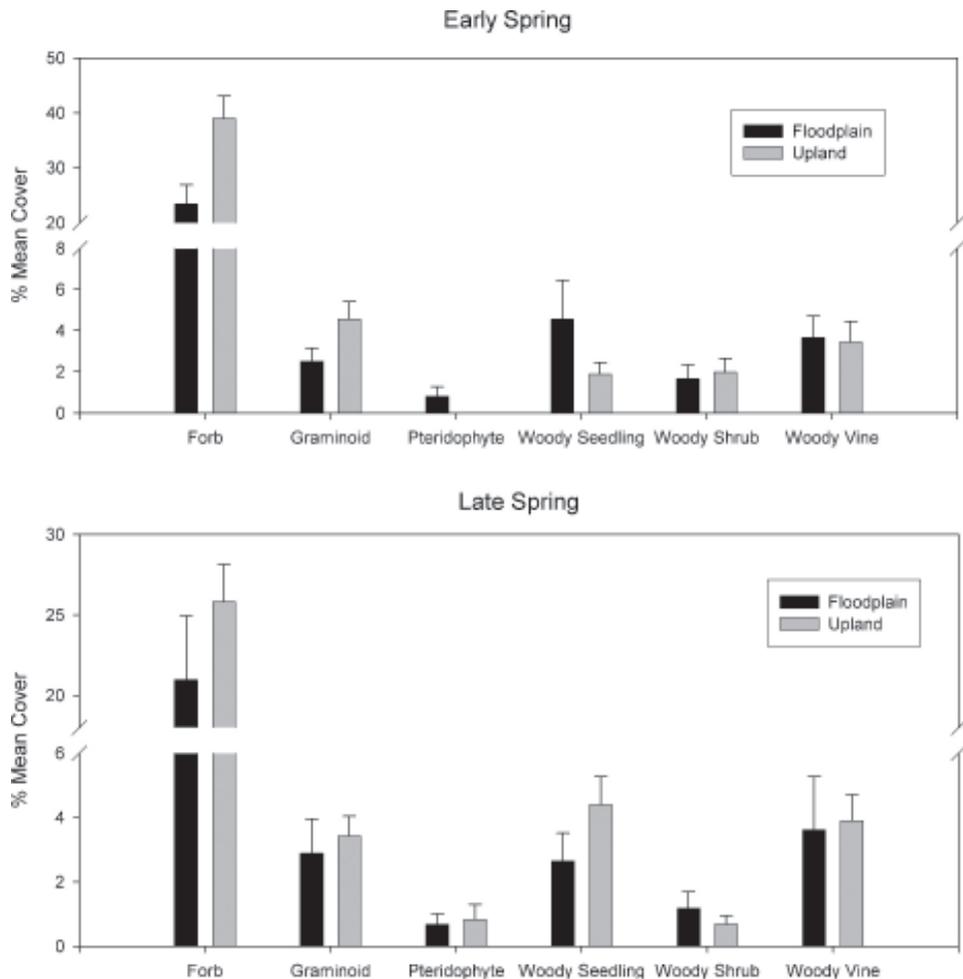


Figure 2.—Mean (± 1 SE) percent cover of plant functional guilds for each survey by landform at Johnson Woods State Nature Preserve, OH.

early and late spring. This may explain the lack of research comparing spring ground-flora communities on floodplains. Spring flooding may also explain the higher mean cover of forbs on upland quadrats than floodplains, contradicting summer structure at Johnson Woods where mean forb cover is higher on floodplains than uplands (Goebel et al. 2003a).

The need for multiple ground-flora surveys throughout a growing season has been expressed by McCarthy (2003) in order to capture a complete floristic record for an area. While McCarthy (2003) suggested a spring, mid summer, and late summer survey to achieve a complete record, this study showed the variability and temporal nature of ground flora communities within just a few weeks of a growing season. Multiple surveys in spring and summer may be required in order to gain a complete account of communities.

By quantifying the variability and underlying dynamics operating in these communities, better suites of reference conditions can be developed and used to guide riparian restoration efforts. Specifically, the variability as expressed in the ground-flora communities at Johnson Woods, as well as other relatively undisturbed forests, should be emulated in disturbed headwater riparian areas of north-central Ohio. From this study, it is apparent that additional research is needed to better understand the vegetation dynamics that characterize our reference ecosystems before actual restoration efforts can be judged successful.

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USING BEFORE-AFTER-CONTROL-IMPACT-PAIRS TO MONITOR THE EFFECT OF LONGWALL MINING ON MATURE FOREST STANDS IN SOUTHEASTERN OHIO

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ABSTRACT.—Longwall mining involves complete removal of coal from underground seams which results in varying degrees of surface subsidence. In Belmont County, OH, The Ohio Valley Coal Co. is planning to mine the Pittsburgh No. 8 coal seam that lies 300 - 600 feet below the surface of an old growth forest, Dysart Woods. The mining, and associated subsidence, is opposed by several groups on the grounds that it may harm the old growth forest. This study was designed to investigate the potential impacts of longwall mining and other environmental stresses by intensively monitoring two forest ecosystems before, during and after mining. One stand (test stand) is in an area that was undermined at a depth of 400 - 500 feet below the surface during the winter of 2001-2002. The other is in an area that was not undermined. Otherwise the two stands are relatively similar. Several standards were used to indicate the health and vigor of the trees and stands, including tree growth rates, vigor ratings, and mortality rates.

The two sites represent maturing stands of 5 acres or larger that possessed many of the attributes of old-growth forests. They both contained a mixture of species that are best described as “mesophytic hardwoods” including oaks, yellow-poplar, white ash, black cherry and a component of shade tolerant species such as American beech and sugar maple. The general approach was to collect pre-mining data for both stands from 1999 through most of 2001. The test site was mined under in late 2001, and then data were collected during all of 2002 to provide post-mining data. The ecological performance (tree growth, vigor, and mortality) was measured to determine if any divergence occurred between the two sites.

The trees responded to the drought in 1999 with a generally reduced radial growth at both sites. In fact the two sites behaved in a similar manner over the course of the study except for the unusually high rate of growth for the test stand in 2001. But the tree response was not associated with mining, since the growth, vigor and mortality of trees at the test site did not change appreciably relative to the control site in 2002, the post-mining year.

Introduction

Coal removal by the longwall method is a type of deep mining process that extracts all the coal in a particular geologic layer (seam) resulting in subsidence at the surface and fracturing of rock layers above the mine. Concerns have been expressed by several groups that longwall mining could result in damage to forests, because of the hydrologic alterations caused by subsidence. Of particular concern is Dysart Woods, an old growth remnant of mesophytic hardwoods in Belmont County, OH.

This study is designed to investigate the potential impacts of longwall mining on forests by intensively monitoring two forest ecosystems before, during and after mining. The methodology we used can be described as before-after-control-impact-pairs (BACIP). This approach is one of several similar methods termed “quasi-experiments” described by Zenner (2003) as being appropriate for elucidating ecological patterns and processes. In preparation for this study, two maturing hardwood stands with an oak component and other site and stand characteristics that were similar to old growth remnant forests in southeastern Ohio were located. One stand (test stand) is in an area that was mined under at a depth of 400- 500 feet below the surface during the winter of 2001-2002. The other is in an area that was not mined under. Although the slope aspects of the two stands were somewhat different, the stands

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were relatively similar in most other ways. The purpose of the control stand was to help quantify the effects of variables unrelated to longwall mining (weather, etc.) and to provide a baseline for comparison with the test stand. Several standards were used to indicate the health and vigor of the trees and stands. These include: tree mortality, density and type of regeneration, tree growth rate, vigor ratings of trees, etc.

Dysart Woods is a remnant old-growth stand that probably originated as part of an extensive forest that once covered vast areas of North America prior to European colonization (McCarthy 1995). Dysart Woods' two stands total approximately 30 acres. Both stands contain an old-growth cohort of oaks (mostly white oak, *Quercus alba*) and yellow-poplar (*Liriodendron tulipifera*). These old-growth trees are reportedly over 300 years of age, approaching the limit of their life expectancy as indicated by the rate of tree mortality, determined to be approximately 3.0% per year in 2003. Shade tolerant hardwoods such as sugar maple and American beech (*Fagus grandifolia*) are gradually replacing the old-growth oaks. A thorough ecological inventory of Dysart Woods was conducted in the fall of 1997 and the results are provided by Hicks and Holt (1999).

Our ecological inventory revealed that the forests displayed reverse J-shaped diameter distributions, typical of all-age, old-growth forests. However, the diameter distribution of the old-growth oak cohort was bell-shaped, typical of even-age stands. The successional replacement of the overstory is well under way at Dysart Woods where a mortality rate of approximately 3.0% per year was observed for large old-growth trees. Nearly all trees in mid-story and understory canopy positions are shade tolerant species such as sugar maple and American beech. Downs and Abrams (1991) concluded that such replacement of old-growth stands with shade tolerant species is inevitable. In fact, they observed a stage of this transition in maturing second-growth oak stands as well. Franklin et.al. (1987) emphasized that tree death and replacement is part of the normal ecological process of stand development.

The old growth stands at Dysart Woods may have originated after a major disturbance such as forest fire. Many authors agree that indigenous people probably used fire extensively to clear land (MacCleery 1992) and the dominance of oaks in many areas of the central hardwood region implies the involvement of fire as a disturbance (Abrams 1992). It is also generally documented that, in the absence of recurrent fire, remnant old-growth oak-hickory stands are becoming rare, regardless of efforts to preserve them, due to the successional replacement of oaks by shade tolerant species (Fralish et al. 1991).

In a comparison of old-growth and second-growth oak forests in Missouri, Shifley et.al. (1995) found that apart from the presence of more large trees (>17 in. dbh), the two types of forest shared many common attributes. Likewise, after comparing the composition and structure of a second-growth and an old-growth forest in Pennsylvania, Downs and Abrams (1991) concluded that they represent "early vs. late stages of oak replacement."

Since old-growth stands are relatively rare, small in extent and subject to changes, it is important to examine the factors that might accelerate these changes. Manion (1981) proposes a so-called "mortality spiral" that represents what occurs when trees are placed under external stress. Stress initiates a chain reaction that may involve attack by secondary organisms (insects and diseases), and ultimately results in death of the tree. Waring (1987) noted that stress results in re-allocation of photosynthates. Repeated drought stress results in trees with relatively larger root systems, sacrificing aboveground growth to favor root growth. Waring also observed that stem growth often shows the impact of stress more quickly than other indicators, stating: "Stem growth only occurs once the resource demands for foliage and root growth have been accommodated." The extension that can be drawn from Waring's conclusion is: if longwall mining initiates stress in trees, this stress will be detectable as a reduced rate of annual radial growth of the affected trees.

The USDA Forest Service, Forest Health Protection program periodically reports forest health conditions in the United States. In a report covering the Northeastern Area (USDA 1993), they list

general forest stressors as “fire, weather and air pollution.” Decline and mortality of trees could potentially be an effect of longwall mining. The USDA Forest service has published a series of Forest Health Monitoring Fact Sheets and they list the following as “indicators” of forest health:

1. Tree Crown Condition Indicator – diameter, live crown ratio, density, dieback, foliage transparency, crown vigor.
2. Tree Growth Indicator – radial growth measured at dbh (diameter breast height).
3. Tree Mortality Indicator – unusually high rates of mortality, often with a spatial pattern.
4. Tree Damage Indicator – presence of fungal fruiting bodies, wounds, cracks, broken or dead roots, etc.
5. Tree Regeneration Indicator – failure of, or abnormal, regeneration compared to expected levels.

Methods

Because of the difficulty and expense of monitoring ecosystem response using replicated experimental designs, we selected the BACIP method, using trees as the experimental unit instead of sites. Osenberg et al (1994) indicate that this method is applicable for comparing impacts of an environmental change, but care must be given to the site selection and the parameters chosen for comparison. Underwood (1994) recommends multiple control locations to improve the power of such experiments, but Stewart-Oaten and Murdoch (1986) indicate that the proper choice of a control site and replication through time will enable BACIP designs to provide reliable results. The primary criteria used for site selection in this study were: Sites should have the proper status regarding mining activity and possess the appropriate species and forest conditions so as to be analogous to old-growth forests. The test site was selected from an area in Belmont County to the west of the village of Centerville, OH and the control site was approximately 15,000 feet away, in an area not planned for mining. Data were collected at both sites beginning in May of 1999. The mining passed by the test site in May, 2001 and went under the site between November and December of 2001, thus data were collected for three growing seasons prior to and one season post mining. The two sites were deemed appropriate since they had the proper species mix, soil and site characteristics, as well as characteristics of old growth. The difference in aspect of the two sites was not manifested in substantial differences in species composition and site quality.

Approach

The general approach of the BACIP design is to establish a baseline by measuring the control and test stands during the pre-mining interval to insure that tree in the two stands responded in similar ways. The ecological performance of the two stands is observed during and after mining to determine if any divergence occurs between the test and control sites after mining.

Initial Ecological Inventory. The stands were sampled using point sampling to obtain an estimate of basal area. Fixed area plots of 0.1-acre were used to obtain most data on trees (2 in. dbh and up), and 6-ft.-radius plots (2 per overstory plot) were measured to obtain regeneration information. A 100% sample of all larger trees (>16 in. dbh) was conducted over the entire stands. Sixteen 0.1-acre overstory plots at the test site and 15 plots at the control site were installed, which is the approximate number to estimate the average basal area to an accuracy of 10% of the true mean. Sample points were located systematically through the area using a sampling grid, beginning from a random starting point. At the center of each sample location, a permanent stake was placed with a label to identify its location. The basal area at this point was estimated using a 10-BAF (basal area factor) prism. The Society of American Forester's cover type (Eyre 1980) was determined based on the stand composition in the area around the sampling point. The slope position aspect and slope inclination were determined at the sampling points. All trees (>2 in. dbh) within a 0.1-acre circular plot around the center point were measured and numbered starting from a north compass bearing and proceeding clockwise until a complete circle has been turned. Each tree's number was painted at dbh. The species, dbh to the nearest 0.1 in., total height to the nearest foot, crown class, vigor class and expected longevity (a subjective judgement) was recorded for each tree. A paint mark was placed at the position where dbh

was measured in order to assure that the same location is used in each re-measurement. Trees in the plots were re-measured at the end of the growing season each year.

Two 6-ft.-radius sample plots, one at plot center and another 18 ft. from center on a random azimuth were established. All woody vegetation was counted by species, and tallied by height classes (<6", 6"-4', >4'). An estimate of total ground cover in percent was also recorded.

In addition to the trees sampled in the 0.1-acre plots, all the large trees (including standing dead) in the stand (>16 in. dbh) were measured and located on a map. These trees were numbered and labeled using a tag attached to the base of the tree. Data recorded for these trees was the same as collected for the trees in the fixed-area plots, including an estimate of the years since death for standing dead trees. These trees in the "large tree" sample were measured for dbh, total height and vigor class each year at the end of the growing season.

From the population of larger trees, a sub-sample of approximately 10 trees was selected from each of 5 species (two per species) in each stand (test and control) for intensive study ($N_{tot} = 20$). The 5 species were: white oak, northern red oak (*Quercus rubra*), yellow-poplar, sugar maple, and white ash (*Fraxinus americana*). Criteria for selecting the intensively sampled trees were size and age (generally larger and older trees were given preference), vigor and appearance (only healthy appearing trees were selected), and proximal competition. Only trees in the dominant and codominant crown class were used, excluding trees at the edge of large canopy gaps or in open-grown situations. In the spring of 1999, two increment cores were extracted from each of the intensively studied trees—the first at a random azimuth and the second 90 degrees clockwise from the first. One core extended to the center of the tree for age determination and both were averaged to construct a skeleton plot of average radial growth over the past 20 years. These trees were equipped with band dendrometers that permitted continuous measurement of growth in circumference. In this way, it was possible to accurately track the radial growth of the trees over time. This precise measurement provided a quality check on the annual measurements of the large trees, which was done using a steel diameter tape. The two banded trees of each species at each site have been monitored biweekly throughout the growing season and monthly during the dormant season for the duration of the study.

Subsequent Inventories. The permanent plot centers have been revisited each year at the end of the growing season for both test and control stands, including three growing seasons in the pre-mining period and one in the post-mining period. All trees in the sample plots were measured for the same characteristics as in the initial sampling, including any mortality that occurred. Regeneration samples were taken in the beginning of the study, but not at the end. It was felt that impacts unrelated to longwall mining (white-tailed deer, droughts, periodic seed crops, shading, etc.) were much more influential on seedling densities than other attributes we were studying, therefore these data were unlikely to yield meaningful results. Regarding the large tree samples for each stand, all trees that were larger than 16 inches dbh in 1999 were measured at the end of each growing season for dbh, total height, vigor rating and expected longevity.

Using the approach indicated above, it was possible to track the radial growth, height growth, vigor and mortality of the population of trees at both sites. The permanent plots allowed for the documentation of the performance of trees in all canopy strata while the large tree sample keyed on the largest and oldest cohort of trees in the stand. This is especially important when relating the present results to old growth forests. The BACIP approach with a test/control pairing and pre-and post-mining data provided an opportunity to calibrate the performance of trees at the two sites before mining and to observe their responses to environmental variations that are unrelated to mining, e.g. the 1999 summer drought. The focus was, therefore, to examine the data for a departure in the growth/vigor/mortality that may occur in the post-mining period for trees at the test site, compared to those at the control site.

Preliminary Analysis

Data from the initial inventory were entered from field data sheets to a spread sheet program where averages for dbh, height, crown class, vigor rating, number of trees per acre, etc. were calculated by stand and by species. Other variables computed by stand and species are: regeneration densities, importance values and measures of species diversity. The utility of these computations was to establish whether or not the test and control sites were relatively similar. Using average basal areas per acre, average dbh, number of trees per acre and site index, the relative stocking of the stands was determined. Differences in stocking that may exist could help explain differences in diameter growth rates between the two stands. Diameter distributions of the two stands were plotted to see if they conform to the reverse J-shaped form expected for old-growth stands. Data on tree densities by species and number of different species were used to compute the Shannon-Weiner H' , a measure of species diversity.

Annual measurements of basal area increment (BAI) and vigor change for both the 0.1 acre fixed-area plots and the large tree sample ($\text{dbh} \geq 16$ in) were tabulated and compared. For both sets of sample data, we used analysis of variance (PROC GLM, SAS Institute 2001) to determine if there were significant differences between the means of test and control stands. The following effects were evaluated: site, year, species, and site * year. For six of the more abundant species (white oak, northern red oak, sugar maple, yellow-poplar, American beech, and white ash), we also evaluated the site * year * species interaction for annual BAI and vigor change.

Diameter growth changes (at breast height) for the 10 intensively measured trees (two each for white oak, northern red oak, yellow-poplar, sugar maple, and white ash) were analyzed using linear regression. To compare growth between species, sites, and years, we fit a least squares line ($y = mx + b$, where y = circumference at day_x, m = slope of the line, and b = intercept) to a portion of the growth curve for each sample tree for the period May 1-September 1 of each year. We then evaluated the significance of the following effects: species, site, year, as well as interactions (site*year, species*site*year) on the slope, m , using PROC GLM (SAS Institute 2001). Finally, we used Duncan's Multiple Range Test to identify significantly different mean values.

Growth ring analysis provided a recent growth history of trees in both stands. If trees in the two stands were initially growing at similar rates, but differed after mining, it would suggest that mining could be responsible for the difference. Thus, the comparisons include both pre- and post-mining comparisons on the same trees as well as treatment-control comparisons between trees in the test and control stands.

Results

Comparison of Initial Site Conditions

The ecological inventory that was conducted in the spring of 1999 revealed that the two stands were relatively similar. Based on data from the 0.1-acre plots, the average dbh for all trees down to 2 inches was 7.2 for the control and 8.4 for the test stand. The control and test stands were both approximately 7 acres in size. There were 175 trees exceeding 16 inches in dbh in the control stand and 222 in the test stand. Thus the density of large trees (> 16 inches dbh) was 24.2 trees per acre in the control stand and 32.2 large trees per acre in the test stand. Table 1 summarizes much of the point/plot data collected at the two sites. As can be seen, the predominant cover type in the control stand was type 60 (beech-sugar maple) while 97% of the plots in the test stand fell in type 27 (sugar maple). The basal area stocking in the two stands was virtually identical with 84 square feet/acre in the control stand and 85 in the test stand. Stem density, seedling density and species diversity was similar for the two stands (table 1). The slope inclination was similar between the two stands, although more plots occurred on lower slope positions at the control site compared to the test site. Both stands were situated along the side slope of a ridge but in the case of the control site, the aspect generally faced southwest whereas the aspect at the test site generally was inclined toward the northeast. The soils, although not identical, were similar, being dominated by Lowell-Westmoreland silt loams. Site quality, expressed as site index (50-year base) of northern red oak trees was estimated from age and height data of 6 trees at the control site and 5 from the test site. The estimated site index at the control site was 76 ft. and it was 71 ft. at the test site, which did not constitute a statistically significant difference.

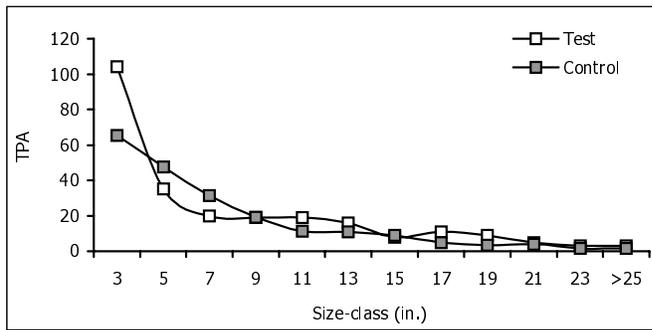


Figure 1.—Size-class distribution for trees ≥ 2 in dbh from 0.1 ac permanent plots, Control and Test sites.

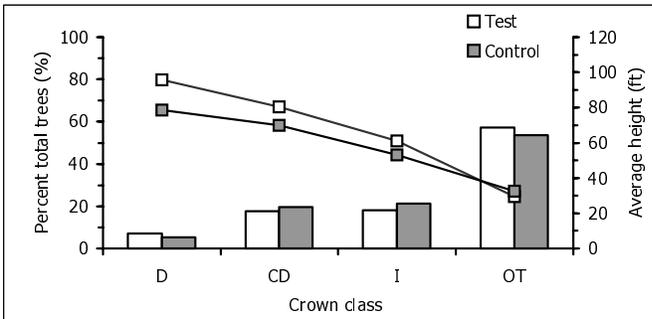


Figure 2.—Percent trees and average height (ft) by crown class from 0.1 ac. permanent plots, Control and Test sites.

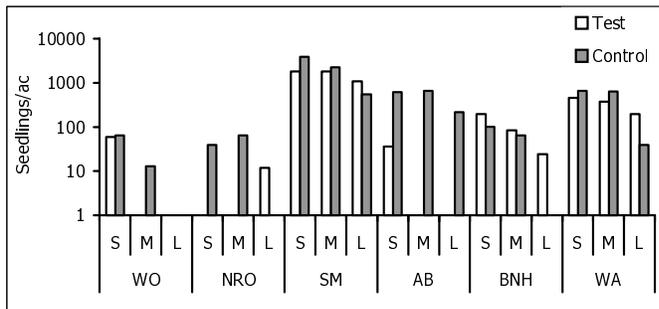


Figure 3.—Tree seedling/sprout size-class distribution for more abundant species from 0.1 ac permanent plots, Test and Control sites. Small = <6" tall, Medium = 6"-4' tall, Large = >4' tall. Species codes: WO = white oak, NRP = northern red oak, SM = sugar maple, AB = American beech, BNH = bitternut hickory, WA = white ash. Note that the y-axis is \log_{10} scale.

The diameter distribution of trees at both the control and test sites conformed to a reverse J-shaped curve that is typical of, but not unique to, mature second-growth and old growth forests (fig 1). The canopy structure of the two stands was very similar as well. In Figure 2 it can be seen that trees in the dominant crown class were fewer in numbers in the control stand as compared to those in the test stand, but had a greater average height (averaging 79 ft. in the control and 96 ft. in the test stand).

The initial regeneration density was fairly abundant at both sites (12,000 seedlings/acre in the control stand and 7,500 in the test stand). Most of the regeneration was shade tolerant species such as sugar maple and American beech. Oak regeneration was found at both sites, but in relatively small numbers, and only smaller-sized seedlings (fig. 3).

Tree species importance values represent a useful means for comparing forest ecosystems. As indicated in Table 2, sugar maple was the dominant species in both stands (being very dominant in the test stand). At the control site, American beech was well represented, followed by yellow-poplar and red

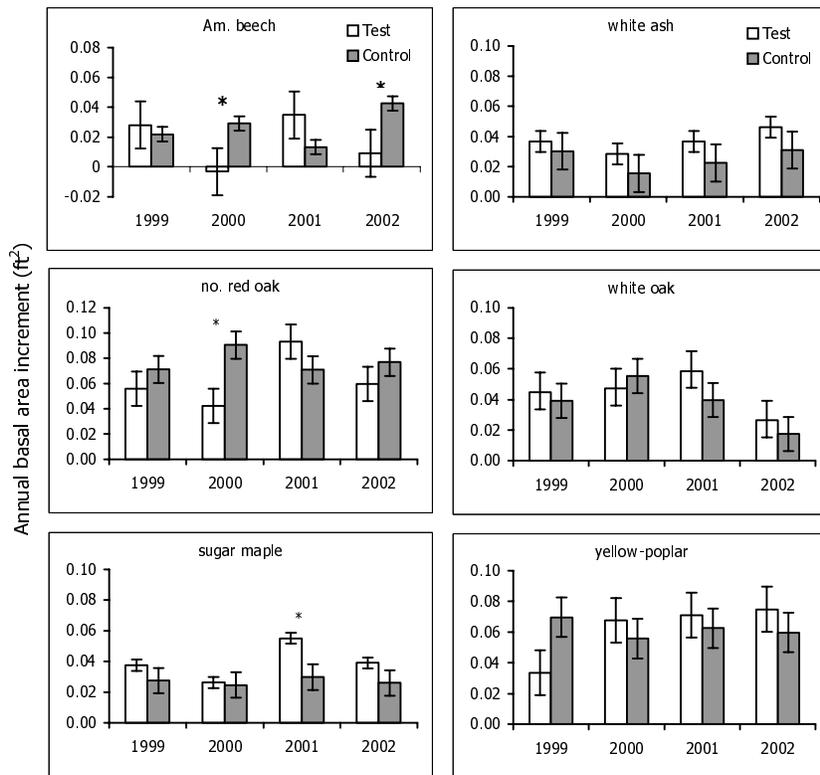


Figure 4.—Mean basal area increment (ft²), 1999-2002, for six tree species (dbh ≥16 in. dbh) for Test and Control sites. Means within years that are significantly different ($\alpha = 0.05$) are indicated by an asterisk.

and white oaks, while at the test site, the importance values fell off dramatically after sugar maple, with red oak ranked in the top 5 species, as well as shagbark hickory, slippery elm and white ash.

The test and control sites were comparable to the other old growth stands in Belmont County (Hicks and Holt 1999; Lafer and Wistendahl 1970). For example, the same species were represented in the overstory of Dysart Woods as in the test and control stands, with the exception of black cherry at the test site and bigtooth aspen and chestnut oak at the control site. In the test and control stands, as well as in other old growth stands, sugar maple had the highest importance value of any species. Dysart Woods had a larger average diameter, somewhat higher basal area and a greater proportion of white oak in the canopy than the test and control stands. But in spite of these differences, the test and control stands were comparable to each other and to nearby old growth remnants.

Ecosystem Changes (1999-2002)

As stated previously, radial growth is considered a good indicator of tree response to environmental stress. Since the underground mining activity occurred beneath the test site in the late winter of 2001, the growing season of 2002 was the first opportunity to look for mining-induced stress. We used the basal area increment (BAI) of the population of large trees (>16 inches dbh) to see if an effect of mining could be detected. We separated them by species. Six species contained sufficient trees to make statistically viable comparisons. Summary results of ANOVA for BAI and vigor class change are given in Tables 3 and 4.

In Table 5 the least-squares comparisons that were made between the main effects site (test and control) and years show that 4 comparisons were significant at the 5 percent level. These include the test vs. control in 2001, the test 1999 vs. the test 2001, the test 2000 vs. the test 2001 and the test 2001 vs. the test 2002. The test vs. control in 2002 (the post-mining year) was not significant. On

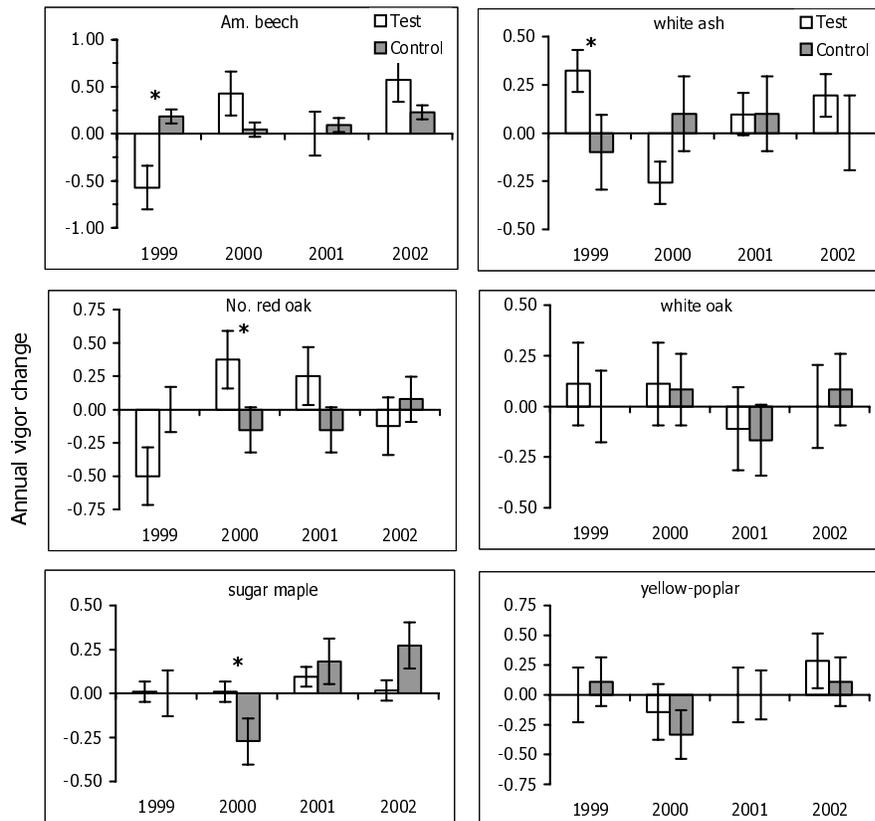


Figure 5.—Mean vigor change, 1999-2002, for six tree species (dbh \geq 16 in.) for Test and Control sites. Vigor rating 1 = healthy crown with few dead branches, 7 = tree dead. A positive vigor change indicates a decline in vigor. Means within years that are significantly different ($\alpha = 0.05$) are indicated by an asterisk.

closer inspection, the reason for all these significant differences appears to be due to the fact that the BAI of the trees at the test site was unusually high in 2001 (the year immediately preceding the undermining of that site). In Table 5, it can be seen that the average BAI for large trees at the test and control sites was approximately 0.04 ft² per year, except in 2001 for the test site, when it was almost 0.06 ft². Since 2001 was a pre-mining year, this would not constitute an effect of mining. The significant interaction that was detected between site, species and year is somewhat more complex in its interpretation. In Figure 4 it can be seen that American beech, the most abundant species in the large tree population at the control site, had somewhat erratic growth, being very high in 2000 and 2002, and low in 2001. Sugar maple, the most abundant species at the test site showed a spike in growth in 2001 at that site. The rapid growth of this species at the test site in 2001 also partially explains the significantly higher overall increment of large trees at the test site in 2001.

Regarding average vigor ratings for trees in the large-tree sample, no significant differences occurred after mining, either with regard to site or year (fig. 5, table 6). There were a few instances in years prior to mining that the difference in average vigor rating of trees of some species was found to be significantly different at one site compared to the other. This was the case for red oak and sugar maple in 2000 (pre-mining). In both cases, the test site trees showed a significant positive difference (reduction in vigor) compared to those at the control site. This was also the case for white ash in 1999 (pre-mining), but the reverse was true for American beech in 1999 (pre-mining), where the trees at the test site actually were rated lower (increase in vigor), compared to those at the control site. The pre-mining vigor changes at the two sites probably represent random fluctuations, or the effect of observers, since vigor rating is a subjective value. Since there were no significant changes in vigor at either site in

Table 1.- Summary data for 0.1 acre sample plots, Control and Test Sites.

Parameter	Control	Test
SAF Cover Type (% total)		
Type 27	20	93
Type 28	7	0
Type 52	13	7
Type 53	13	0
Type 59	13	0
Type 60	33	0
Basal area (0.1 ac. plots, ft ² /ac.)	84.00 ± 7.5	84.95 ± 9.4
Basal area (BAF - 10)	115	95
Stem density (TPA)	210	255
Seedling density (stems/ac.)	12,119	7486
Species diversity (<i>H'</i> , Shannon-Weiner)	2.38	1.86
Species evenness (<i>E</i>)	0.781	0.701
Species richness (trees, ≥ 2 in. DBH)	21	14
Slope percent (% , average)	36.6	26.1
Slope position (% total)		
ridge	0	18.8
upper slope	0	18.8
lower slope	86.7	62.5
bottom/hollow	13.3	0
Aspect class (% total)		
North	0	43.8
Northeast	0	25
East	0	18.8
Southeast	6.7	0
South	6.7	12.5
Southwest	33.3	0
West	40	0
Northwest	13.3	0

Table 2.- Species importance values (IV) for all trees ≥ 2 in. dbh. $IV = (RBA/ac+RF+RA)/3$.

Species	Test	Control
white oak	5.4	13.4
no.red oak	6.0	8.3
black oak	2.8	0.5
scarlet oak	--	1.2
chestnut oak	--	1.1
sugar maple	50.6	22.6
red maple	--	5.1
Am. beech	3.3	19.8
shagbark hickory	7.6	1.0
black cherry	2.2	3.8
yellow poplar	1.0	7.2
white ash	7.5	0.5
bigtooth aspen	--	4.3
black walnut	0.6	0.6
black gum	1.1	3.4
dogwood	--	2
e. hophornbeam	--	2.8
slippery elm	7.5	1.0
hackberry	1.9	--
bird cherry	1.9	--
pignut hickory	--	0.5
musclewood	--	0.6
mockernut hickory	--	0.5
Am. elm	1.2	
Totals	100.0	100.0

the post-mining period, this can be interpreted to indicate that there was no detectable effect of mining on tree vigor rating to date. Another way of looking at vigor ratings is to observe what proportion of the species increased, decreased or remained unchanged in vigor. Regarding the six species that were compared in prior analyses as to their vigor ratings, at the test site in 2002 one species (17%) remained unchanged in 2002 compared to 2001, one species (17%) improved in vigor and the remaining four species (66%) declined in vigor. Species at the control site were similar, with one species (17%) showing no change and five species (83%) showing a decline in vigor rating for 2002. These declines in vigor could be related to natural phenomena, such as the drought of 1999.

Tree mortality is another measure of health (Elliott and Swank 1994). The annual mortality rate estimated for old growth forests often ranges from 1% to 2% per year. The mortality rates observed in our large tree sample were below 1% at both sites (table 7). The average annual mortality rate for the test site was 0.77%, whereas the rate for the control site averaged 0.41% per year. Interestingly, about 70% of the mortality at the test site occurred in 1999 after the severe summer drought of that year and only one tree died at the test site in 2002, following mining.

Data from the 0.1-acre fixed-area plots provided useful information, especially regarding trees in the lower canopy positions (understory). Results from our fixed-area plots for radial and basal area growth of understory trees is summarized in Table 8. Sugar maple, a very shade tolerant species, dominated the understory of both sites, and because of low numbers of other species, sugar maple appears to be the only species that provides a legitimate comparison between the understory response of the two sites. Basal area growth of understory sugar maples at both sites was extremely slow, owing to the limited resources in the understory and the small size of the trees. At the control site, the average annual growth per understory sugar maple was 0.0015 square feet over the four years whereas understory sugar maples at the test site averaged 0.004 square feet per tree per year. In 2002, the post-mining year,

Table 3.- Analysis of annual basal area increment for trees ≥ 16 in. dbh, 1999-2002, by site and by species.

Factor	df	SS	MS	F	p
Site	1	0.0002	0.0002	0.12	0.726
Year	3	0.0067	0.0023	1.50	0.213
Species	5	0.1528	0.0306	20.40	<0.001
Site x Year	3	0.0162	0.0054	3.61	0.013
Site x Year x Species	35	0.0791	0.0023	1.51	0.030

Table 4.- Analysis of annual vigor change for trees ≥ 16 in. dbh, 1999-2002, by site and by species.

Factor	df	SS	MS	F	p
Site	1	0.1167	0.11666	0.31	0.577
Year	3	2.6072	0.86905	2.32	0.074
Species	5	1.2250	0.24499	0.65	0.659
Site x Year	3	1.7860	0.59532	1.59	0.190
Site x Year x Species	35	20.1076	0.57450	1.53	0.025

Table 5.- Least squares means of annual basal area increment (ft^2) for site x year interaction for trees ≥ 16 in. dbh, and p-values for least square means comparison for site x year interaction. The first letter indicates C=Control or T=Test, and the final four digits indicates year. Values less than 0.05 indicate that basal area increment values are significantly different ($\alpha=0.05$)

Site	BAI (ft^2)	C-1999	C-2000	C-2001	C-2002	T-1999	T-2000	T-2001
		0.043	0.045	0.040	0.042	0.039	0.035	0.058
C-2000	0.045	0.754						
C-2001	0.040	0.554	0.365					
C-2002	0.042	0.863	0.627	0.675				
T-1999	0.039	0.548	0.374	0.956	0.655			
T-2000	0.035	0.195	0.113	0.452	0.255	0.516		
T-2001	0.058	0.022	0.045	0.005	0.014	0.007	0.001	
T-2002	0.043	0.914	0.691	0.662	0.959	0.646	0.268	0.025

Table 6.- Least squares means of annual vigor change for site x year interaction for trees ≥ 16 in. dbh.

Year	Control	Test
1999	0.03	-0.10
2000	-0.09	0.09
2001	0.01	0.06
2002	0.13	0.16

understory sugar maples at the test site grew at a rate of 0.005 square feet per tree, while understory sugar maples at the control site also exceeded their four-year average and grew at a rate of 0.004 square feet.

With regard to vigor rating for the understory trees, the results are also displayed in Table 8. Here it can be seen that in 2002, the post-mining year, 67% of the species at the control site showed declining vigor (a higher number for vigor rating) while only 43% of the species at the test site showed declining vigor. Sugar maple, the most abundant species, did show a greater increase in vigor rating (a larger decline in vigor) at the test site than at the control after mining. However, understory trees are not

Table 7.- Annual mortality for all trees, by sample unit, site, and year.

Sampling unit	1999	2000	2001	2002	Average %
<i>0.1 acre plots</i>					
Test (N ₁₉₉₉ = 272)	2	1	2	2	0.64
Control (N ₁₉₉₉ = 327)	0	15*	3	8**	1.99
<i>Large tree sample (dbh ≥ 16 in)</i>					
Test (N ₁₉₉₉ = 226)	5	1	0	1	0.77
Control (N ₁₉₉₉ = 183)	0	2	1	0	0.41

*12 of these were overtopped trees (primarily maples, dogwood, and hophornbeam).

**6 of these were overtopped trees (primarily maples, dogwood, and hophornbeam).

Table 8.- Mean basal area increment (ft²), and mean vigor change for more abundant understory (overtopped) tree species in 0.1 ac. fixed area plots, Test and Control sites.

	Species	N	Basal area increment (ft ²)				Vigor change			
			1999	2000	2001	2002	1999	2000	2001	2002
Control	white oak	11	-0.004	0.005	-0.001	0.000	-0.1	0.1	0.0	0.0
	northern red oak	6	0.000	0.003	0.001	0.002	0.0	0.1	0.0	0.1
	scarlet oak	4	-0.001	0.003	0.001	0.000	0.0	0.1	0.0	0.0
	sugar maple	69	-0.001	0.004	-0.001	0.004	0.0	0.1	0.0	0.1
	red maple	11	0.008	0.003	0.000	0.005	0.5	0.0	0.0	0.1
	Am. Beech	35	0.000	0.004	0.001	0.001	0.0	0.1	0.1	0.0
	black cherry	4	0.004	-0.003	-0.003	0.000	0.1	-0.1	-0.1	0.0
	yellow-poplar	5	-0.006	-0.001	0.001	0.009	-0.2	0.0	0.0	0.2
	blackgum	6	0.000	0.000	-0.005	0.001	0.0	0.0	-0.1	0.0
	dogwood	6	-0.004	0.001	0.010	0.002	-0.1	0.0	1.0	0.1
	hophornbeam	9	0.000	0.004	0.001	0.009	0.0	0.1	0.0	0.4
	Test	sugar maple	144	0.002	0.006	0.003	0.005	-0.3	0.3	0.0
shag. Hickory		7	-0.005	0.009	0.002	0.002	0.2	-0.7	0.3	1.0
hackberry		4	-0.002	0.005	-0.003	-0.001	-0.3	1.0	0.0	0.0

good subjects for assessing mining-related impacts since they are exposed to competition from overtopping trees, which probably overwhelms other potential impacts.

Mortality of trees in the 0.1-acre plots (including understory trees) is presented in Table 7. Note that trees at the control site showed a higher rate of mortality than those at the test site (1.99% compared to 0.64%). In 2002 the mortality rate for trees in the 0.1-acre plots at the control site was above the 4-year average at 2.4%, while the mortality rate at the test site was just above its 4-year average at 0.73%. These results seem to corroborate the results from the large tree sample, that being that no detectable effect of mining on tree health was apparent.

Regarding seasonal diameter growth for the intensively monitored trees, a plotting of cumulative growth for each sample tree over the term of the study was prepared. These are shown in Figure 6. When slope values for the four growing seasons were compared, the effects of species and year were highly significant ($p < 0.0001$), as was the interaction of species-site-year ($p = 0.033$). The significance of year as an effect can be attributed to annual variations in climatic conditions (fig.7). For example, using Ohio weather data, precipitation during the 1999 growing season (May-September) totaled 13.5 cm; totals for 2000 and 2001 were 20.0 and 20.1 cm, respectively. The Palmer Drought Severity Index (PDSI) during the 1999 growing season averaged -2.5, a value that is slightly above the threshold for severe drought (-3.0). In contrast, PDSI for the 2000 and 2001 growing seasons averaged 0.76, slightly above normal.

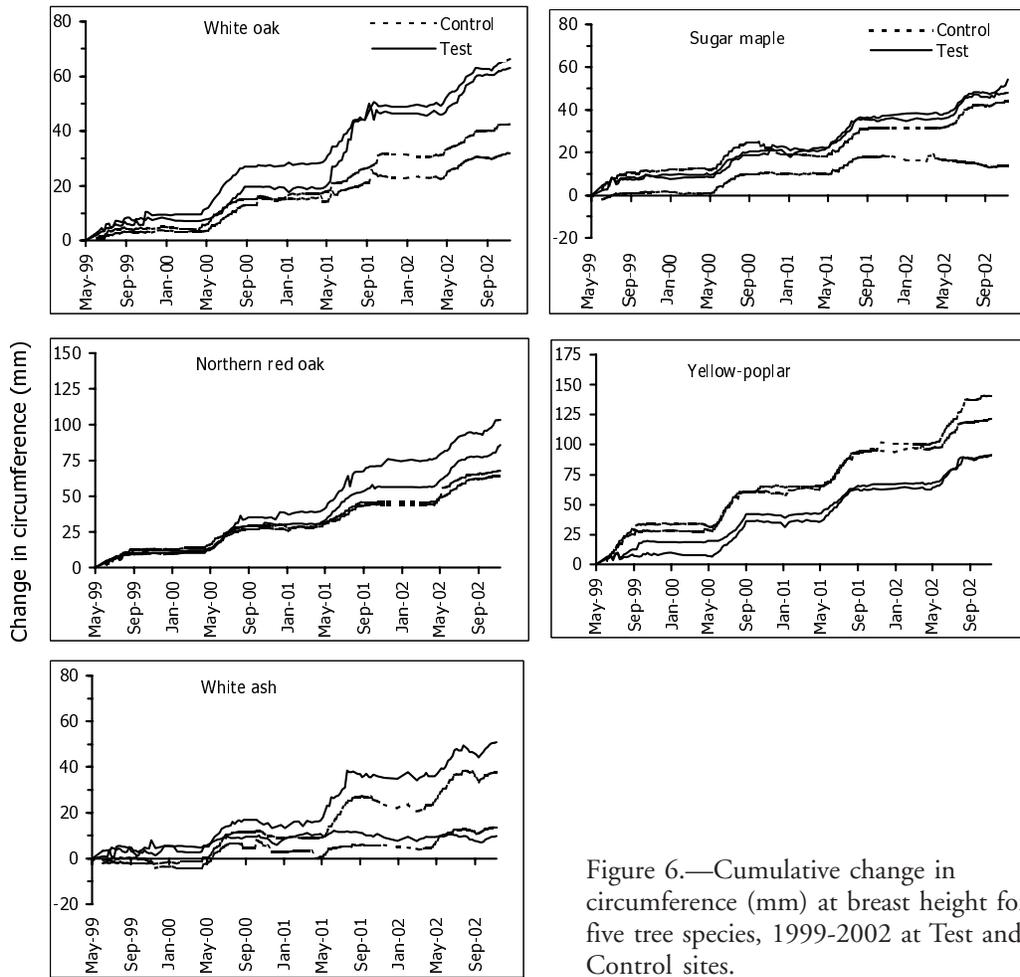


Figure 6.—Cumulative change in circumference (mm) at breast height for five tree species, 1999-2002 at Test and Control sites.

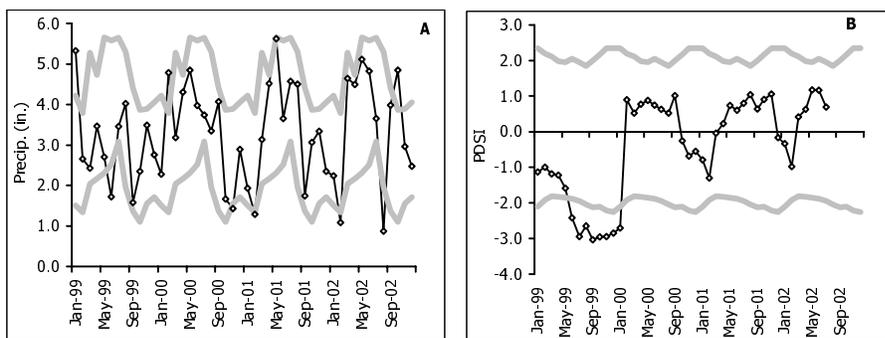


Figure 7.—Monthly precipitation (A) and Palmer Drought Severity Index (B), 1999-2002, Climate District 12, Ohio. Gray lines indicate + 1 Stdev. From 100 year mean.

The significant effect of species on rate of radial growth was also expected. The five species examined in this study represent a spectrum of adaptation ranging from species typically associated with mesic habitats (sugar maple and yellow-poplar), a drought tolerant species generally associated with more xeric sites (white oak), as well as two species with wider amplitude (northern red oak and white ash) (Burns and Honkala 1990). Life history strategies of the species present another gradient, ranging from fast growing and opportunistic species such as yellow-poplar and northern red oak to more conservative species such as white oak and sugar maple (Bormann and Likens 1979, Burns and Honkala 1990). The combination of habitat preference and life history strategies resulted in distinct growth responses for the five species over the four sample years. These growth differences are reflected in slopes of mean growing season growth trend, following the order, from greatest to smallest: yellow-poplar > northern red oak > white oak, sugar maple, and white ash.

Over the four-year period, the effects of site and the interaction of site and year yielded no statistically significant differences ($p > 0.05$). The absence of a site effect, in part, corroborates the initial selection of study site locations. The fact that the interaction of site and year was non-significant suggests that mining under the test site during the winter of 2001 did not result in detectable significant changes in overall growth trends when compared to the previous years. These findings are similar to those obtained by Runkle (1993) from a single-point-in-time study.

Conclusions

This study used a BACIP design to investigate the impact of longwall mining on a mature forest ecosystem. Because tree growth is a function of the environment they are exposed to, an effort was made to study the growth processes that could be affected by mining.

Two sites were selected that supported mature forests with a species composition similar to nearby old-growth forests. The radial growth of trees in the dominant canopy was used as a primary measure of the trees' response to changes brought about by mining. The study was conducted over a period of four growing seasons, three prior to mining under the test site and one after mining.

The trees responded to the drought in 1999 by showing a generally reduced radial growth at both sites. Tree responses did not appear to be associated with mining, since the growth, vigor and mortality of trees at the test site did not change appreciably relative to the control site in 2002, the post-mining year. Based on the results of this well documented and controlled study, longwall mining under the conditions tested had no discernable effect on the health of the forest or trees above the mine after one year. The BACIP design is a practical approach for studies of this type and continuation of the monitoring for several post-mining years will provide a more definitive answer regarding the impact of longwall mining on maturing forests.

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CONTROLLED CROSS-POLLINATIONS WITH AMERICAN BEECH TREES THAT ARE RESISTANT TO BEECH BARK DISEASE

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ABSTRACT.—American beech tree pollen with viability ranging from 30 to 50 percent was used to perform controlled crosses between resistant parents, a resistant and susceptible parent, and a resistant and intermediate parent. The germinative capacity of the seeds collected from the controlled crosses varied from 12 to 84 percent, possibly indicating mating incompatibilities. The percentage of barren seed ranged from 61 to 89 percent. Open-pollinated seeds were collected from both resistant and susceptible trees, including some of the parental trees used in the cross-pollinations. The germinative capacity and percentage of barren seed were comparable to that found in the cross-pollinated seeds, indicating these numbers were not a result of the pollination bagging process interfering with seed development. Self-pollinated controls indicated a high level of self-sterility. The full- and half-sib families generated will be challenged with scale insect eggs to determine their resistance phenotype and gain insight into the mechanism(s) of inheritance of resistance.

Beech bark disease is initiated when the bark of beech trees is colonized by the beech scale insect, *Cryptococcus fagisuga*, predisposing the tree to subsequent invasion by fungi of the genus *Nectria*. An estimated 1-2 % of beech trees remain disease free in stands long-affected by beech bark disease and challenge trials have shown that they are resistant to the scale insect. Increasing the number of resistant beech trees while reducing the proportion of susceptible trees is currently thought to be the best management approach to minimize the overall impact of beech bark disease (Mielke et al., 1986). However, very little is known about the mechanism(s) of resistance and how resistance is inherited. Studies in European beech (*Fagus sylvatica*) concluded that some beech trees have a genetic predisposition to infestation by the beech scale (Gora et al., 1994; Krabel & Petercord 2000) and a correlation was established between scale infestation and the genotype of the host tree based on isozyme analysis (Krabel & Petercord 2000). No such correlation was found in American beech (*Fagus grandifolia* Ehrh. Houston & Houston, 1994; Houston & Houston, 2000). The inability to correlate resistance with a specific isozyme pattern indicates that a more sensitive marker system such as one that is DNA-based may be required to detect linkage. Furthermore, to gain meaningful insight into the genetic mechanism(s) of resistance, the inheritance of the trait must be studied within a family. To address this, we have initiated a research program to determine the requirements for performing controlled-cross pollinations and to generate both full- and half-sib families in an effort to identify how resistance may be inherited. Such knowledge is a prerequisite to establishing a breeding program and developing seed orchards in order to supply a source of resistant American beech for restorative use and for use in pre-emptive plantings as a way to minimize the impacts of beech bark disease.

Study Area

All of the parent trees used in the cross-pollinations were located in Ludington State Park, Ludington, MI. Ludington State Park provided the ideal setting for this work for several reasons. Of critical importance was the fact that it is the location of a “killing front” of the disease—both heavy scale infestation and *Nectria* are present. Because of the intense disease-pressure at this site, mature beech trees that remain scale-free have a high probability of being truly resistant and not merely “escapes”. Even so, trees chosen as putatively resistant parents are currently being tested through insect challenge experiments to confirm their resistance. The estimated minimum age for seed production in American beech is 40 years, and a beech tree of that age can reach heights between 70 and 120 feet (Rudolf &

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Leak, 1989). Based on our observations, flowers are most prevalent in areas of the canopy which are in direct contact with sunlight, generally toward the top of the tree. The trees chosen for use as parents in this study had to not only be free of scale and canker in the case of resistant parents, but they also had to be near a road. The paved roadways leading into and out of the campground areas in Ludington State Park provided a way to maneuver a bucket truck close enough to the parent trees to allow access to the flowers. Other areas included in this study were a 150 acre stand in Sebois County, Maine and Delaware State Park, Ohio. The stand in Maine is considered an aftermath forest, and has been heavily affected by beech bark disease. This stand was of particular interest for the collection of seed for half-sib families because all of the susceptible trees have been removed (D. Struble, personal communication) and there is subsequently a high probability of also having a resistant pollen donor. The trees in Delaware State Park were used in self-fertilization experiments and were chosen simply based on proximity to our research facility.

Methods

Pollen collection

Dormant branches, harvested on May 13, 2002 were approximately 3-4 ft. in length with a diameter of ½". Once a branch was excised from the tree it was placed in a bottle of tap water and parafilm was wrapped around the opening to prevent loss of water during transport. Upon returning to the laboratory the bottom ½ inch of each branch was removed, and the branches were placed in four-liter flasks containing 0.25 X MS media (Murashige & Skoog, 1962). To keep the harvested dormant branches alive long enough in the greenhouse for flower emergence to occur, it was critical to minimize the amount of bacterial and fungal growth in the media. Such contamination results in blockage of the phloem, preventing water and nutrients from being taken up by the cut branch. For this reason, 25 mg/L of rifampicin and 10 mg/L of nystatin were added to the liquid media. Media was changed every 2-3 days, and at each change the bottom portion (¼" to ½") of the branch was cut. The flasks were kept on benches in the greenhouse under ambient light with daytime temperatures of 18° C and nighttime temperatures dropping to 13° C. Once bud break occurred, a 0.75 % solution of sucrose was added to the media. After five days pistillate flowers began to emerge and by nine days staminate flowers were clearly visible. Mature, dehiscent anthers were clipped and ground over a 53 µm nylon sieve to separate the pollen from the flower debris. Sieved pollen was transferred to glass vials sealed with a rubber septa and placed in 50 ml polypropylene tubes along with a packet of silica gel and stored at 4° C (P. Sisco, personal communication).

Pollen Viability

Once pollen was harvested, viability was confirmed by looking at pollen tube formation on artificial media. The pollen germination media contained 1X aspen culture media (Ahua, 1984) and 15 % sucrose, pH 5.6. The media was passed through a 0.45 µm filter prior to adding 20 g/L gelatin. The gelatin was dissolved by incubating the solution in a 55° C water bath. The media was cooled to 40° C and 40 mg/L nystatin and 25 mg/L rifampicin were added. The media was poured into 65 mm x 10 mm petri dishes and stored at 4° C. To test the pollen, a paintbrush was used to pick up pollen grains and lightly tapped to dust pollen onto the gelatinous germination plates. The plates were kept at room temperature in the dark. Pollen grains that had developed germination tubes were counted after a minimum 48 hr. incubation period using a dissecting microscope. To calculate the percent viability, the number of pollen grains that had a visible germination tube out of a minimum of 100 pollen grains observed was determined for each sample.

Controlled Cross-Pollinations

Pollinations bags were made out of Tyvek® home wrap by folding and sealing the edges with Goop® household waterproof adhesive, and reinforcing the folded, sealed edges with staples. Pollination bags were placed over dormant branches on April 22, 2002. Heavy duty outdoor carpet tape and staples were used to seal the edge of the bag that came in contact with the base of the branch. On May 23, 2002 the bags were inspected for defects, and then removed so that pollen could be applied directly to the stigmas of individual flowers using a paintbrush. Following application of the pollen, the bags were immediately put back into place to prevent any other contaminating pollen from entering. Once all

anthers in the immediate vicinity had finished dehiscing, the bags were removed (June 13, 2002) to allow for normal seed development. The bags were again inspected for any damage and any holes or tears were noted.

Seed Collection, Storage and Germination

Two types of seed collection bags were put into place on July 30, 2002, to prevent predators from eating the seeds prior to their harvest and to collect any burrs that may have abscised prematurely. One type of bag was a commercially available mesh cotton laundry bag with a drawstring. A second type of bag was constructed from nylon mesh window screening. The seams of these bags were sealed using strips of Tyvek and Goop® adhesive. The seed collection bags were removed Sept. 5, 2002 at which point seeds were harvested. Burrs resulting from controlled-cross pollinations were removed directly from the branch, placed in plastic baggies and stored on ice for transport. Open-pollinated seed was collected directly from unbaggged branches or by removing entire branches and transporting them back to the laboratory in water. At the time of harvest, all burrs were still tightly closed and slightly green in color. The burrs were laid out on flat trays and allowed to air dry overnight, which resulted in the triangular-shaped nuts being released. The seeds were then placed in plastic bags filled with damp peat moss and stored at 4° C for 130-150 days. After the period of cold treatment, seeds were dissected from their seed coats and sowed in flats of Metro Mix 510 amended with micronutrients (90 g per 3 cubic yard bag of Metro Mix) and osmocote (Scott's Corp., Dublin, OH; 300 g per bag). The flats were put in the greenhouse under a 16 h photoperiod with daytime temperatures set at 22° C, and nighttime at 20° C. Drip irrigation was used daily to keep the media moist.

Results

Pollen Collection and Viability

To our knowledge, there are no published reports of successful controlled cross-pollinations in *Fagus grandifolia* and very little is known about flower development, stigma receptivity and the timing of pollen release. In *F. sylvatica* and *F. grandifolia*, both female and male flowers can be found in the same bud (Nielson & De Muckadeli, 1954; Garrison, 1957). For this reason, the branches could not be emasculated at the time the pollination bags were initially put into place by removing male buds. *F. sylvatica* is metandric, meaning that the female flowers are receptive before male flowers on the same tree begin to release pollen. Our observations confirmed this was the case with *F. grandifolia* as well. Female flowers emerged up to five days before male flowers were observed. The onset of female receptivity, which was identified by curvature of the stamen as was reported in *F. sylvatica* (Nielson & De Muckadeli, 1954), occurred prior to pollen release. Although little is known about the frequency of self-fertilization in *F. grandifolia*, reports in *F. sylvatica* indicate that self-fertilization is an infrequent occurrence (Nielson & De Muckadeli, 1954). For this reason and because of the time constraints involved, instead of removing male flowers we chose to minimize self-fertilization by performing the pollinations prior to the occurrence of pollen release. This made it necessary to force pollen production so a viable source of pollen was available during the brief window of time when the female flowers were receptive but the male flowers had not begun to dehisce.

The dormant branches harvested on May 13, 2002 were very close to bud break. Some trees had pistillate flowers already emerging, while others didn't emerge for another day or two after placement in the greenhouse. Staminate flowers emerged between 2 to 5 days after the appearance of the female flowers. Pollen was collected between May 15th and May 19th and was immediately dusted onto germination media prior to being stored at 4° C in a sealed vial along with silica gel packs as a desiccant. Results of the pollen viability assays performed on May 19, 2002 show viability levels between 30 and 50 percent (table 1). The viable pollen was used for pollinations on May 23, 2002 and at this time the female flowers appeared to be in stage 2, meaning they were receptive according to the criteria reported by Nielson & De Muckadeli (1954). None of the parental trees were actively releasing pollen at this time. Pollen viability assays were performed again after the pollinations were complete, on June 24, 2002 (Table 1). At this time, the pollen had been transported on ice, removed from the ice for several hours while it was brushed on individual flowers, and returned to storage at 4° C for one month. A drop in viability was observed, and this reduction in viability varied between

Table 1.—Pollen viability in trees of American beech.

Pollen Donor	Maternal Parent	Viability 5/19/02	Viability 6/24/02	No. of flowers brushed	No. nuts per flower
1501 (I)	1504 (R)	50 %	4 %	69	0.51
1506 (S)	1504 (R)	30 %	16 %	249	0.32
1504 (R)	1506 (S)	50 %	41 %	98	0.97
1504 (R)	1505 (R)	40 %	28 %	115	0.53
1506 (S) @ 4° C	none		51 %	50 %	-
1506 (S) @-20° C	none		54 %	50 %	-

Table 2.—Open-pollinated seedlings derived from seeds of American beech trees.

Parent Tree	No. Seeds	% Full	Germinative Capacity	No. of Plants
1506 (S)	802	35	60 %	168
1504 (R)	2081	28	8.5 %	49
1511 (S)	478	24	2.6 %	3
ME (R)	283	75	53 %	149

individual samples (Table 1). It was encouraging to note that the pollen retained viability after being in the uncontrolled field environment for such a long period of time. It is also interesting to note that the pollen collected from tree 1504, which exhibited the smallest drop in viability also resulted in the highest number of filled seed being produced per flower, 0.97. In contrast, 1501 pollen had a drop in viability from 50 percent to 4 percent, yet still produced 0.51 nuts per flower. A portion of the pollen collected from 1506 was not used in any of the pollinations and was instead aliquoted and stored at either 4° C or -20° C to study how well viability was maintained at these two temperatures over a period of time. The starting viability was determined on May 19th and viability was assayed again on June 24th (table 1). There were no differences in viability between 1506 pollen that had been stored at 4° C and 1506 pollen that had been stored at -20° C. In addition, viability of pollen stored at both of these temperatures did not decrease from the initial test date to the final test date, over a month later. Some samples of pollen were freeze-dried overnight, and this resulted in a complete loss of viability (data not shown). All of the pollen samples shown in Table 1 were again assayed for viability after one year in storage at 4° C and all had dropped to 1 percent or below. Although we did not have enough pollen to do extensive studies on optimal long-term storage conditions, a batch of 1503 pollen (not used for pollinations) was aliquoted and half stored at -20° C for one year and the other half stored at -80° C. After one year, the viability of the 1503 pollen stored at -20° C had dropped from 50 percent to 1.5 percent, but the pollen stored at -80° C retained a level of 47 percent viability (data not shown). In the future, studies will be dedicated to working out optimal conditions for long-term storage and will look at not only storage temperature but moisture content as well.

Open-Pollinated Seed

Open-pollinated seed (table 2) was obtained from two of the parents used in the cross-pollinations study, the resistant tree 1506 and the susceptible tree 1504. Open-pollinated seed was also collected from the susceptible tree 1511 and from a tree located in Sebois County, Maine-ME. Between 24-35 percent of the seeds collected from trees at Ludington State Park (1506, 1504, and 1511) were full (table 2). This figure is only slightly higher than the reported 13 - 29 percent of sound nuts collected from 20 trees in East Lansing, MI (Gysel, 1971). A greater degree of variability in the percent sound seed was reported by Sain and Blume (1981) for American beech seed collected from trees in the Great Smokey Mountains National Park (GSMNP). In this report, the number of full seed ranged from 0.8 to 95 percent. Comparisons of trees in high elevations with trees in low elevations found no correlation between elevation and the production of sound nuts. The percentage of seeds collected from the ME tree that were sound was much higher (75 percent) than those collected from Ludington State Park.

Table 3.—Controlled cross-pollinated seed from American beech trees.

Cross (female x male)	Full	Seeds			Total	% Full	Germinative Capacity	Total No. Plants
		Germinated	Rotten	Empty				
1506 (S) x 1504 (R)	11	84	0	146	241	39	81 %	77
1504 (R) x 1506 (S)	49	31	10	585	675	13	12 %	11
1504 (R) x 1501 (I)	35	0	0	98	133	26	37 %	13
1505 (R) x 1504 (R)	28	33	0	170	231	26	84 %	51

This value was comparable to the values reported by Leak & Graber (1993) for seed collected from beech in the White Mountain National Forest. Over a 6-year period of time seeds from this area were consistently between 75 and 88 percent sound. Studies in *F. sylvatica* also reported variability in the number of seed produced that were full, ranging from 3 to 92 percent (Nielsen & De Muckadeli, 1954). Nielsen & De Muckadeli (1954) also reported an experiment in which branches were emasculated prior to pollen release and then bagged to prevent pollen from entering. These branches still produced large amounts of nuts, but all of them were barren thus proving that *F. sylvatica* has the ability to produce fruit lacking a seed in the absence of fertilization, a phenomenon known as parthenocarpy. Observations by H.J. Garrison (1957) on developing *F. grandifolia* fruit led her to conclude that in the absence of pollen a seedless fruit develops and it is similar in size to fruit containing a seed. This would suggest that the variability in the quantities of empty seed produced may be dependent on pollen production and staminate flower development as well as weather conditions such as high winds and heavy rainfall which may interfere with pollinations.

Cross-Pollinated Seed

The results of the controlled cross-pollinations are listed in Table 3. Overall, the germinative capacity (the percent of sound seed that germinated) was variable, ranging from 12 to 84 percent. However, compared to open-pollinated seeds from this study, the average germinative capacity of cross-pollinated seeds was greater. This was also the case in *F. sylvatica* (Nielsen & De Muckadeli, 1954). The percent of barren seed was similar in both the cross-pollinations and the open-pollinations, with the exception of the 1504 female x 1506 male cross which produced a slightly lower number of 13 percent sound seeds. This similarity between cross-pollinated and open-pollinated seed production provides an indication that the pollination bagging process did not negatively affect seed development. The production of a lower percentage of sound seed in the 1504 female x 1506 male cross could possibly be attributed to an incompatibility between the parents or perhaps 1504 does not produce vigorous pistillate flowers. Open-pollinated 1504 flowers produced seed with a low germinative capacity and the two controlled crosses that used 1504 as the maternal parent produced seed with lower germinative capacities compared to seeds from crosses that used 1505 or 1506 as the maternal parent.

American beech are monoecious, having both staminate and pistillate flowers on the same plant, making it possible to perform reciprocal crosses. However, due to other limiting circumstances such as the quantity of pollen harvested and the number of branches that could be reached for bagging, only one set of reciprocal crosses was performed in this study. The susceptible tree 1506 was used as both a pollen donor and a maternal parent along with the resistant tree 1504. Interestingly, when 1504 was used as a pollen donor with both 1506 and the resistant tree 1505, seeds with a high germinative capacity (81 and 84 percent, respectively) were produced. But when 1504 was used as a maternal parent with either 1506 pollen or 1501 pollen, the seeds produced had germinative capacities of 12 and 37 percent, respectively. For tree 1506, the opposite pattern was observed; this tree was more successful as a maternal parent than a pollen donor. Differences in compatibility between pairs of parents is not uncommon and identifying compatible combinations is an important part in developing seed orchards for tree improvement (Lambeth, 1993).

Self-Fertilization

Virtually nothing is known about the extent of self-fertilization in the American beech. Self-pollination experiments that were done in European beech demonstrated that *F. sylvatica*, for the most part has a

Table 4. Self-fertilization in trees of American beech.

Tree	Self-Pollinated Controls			Open-Pollinated		
	Total Seed	Full Seed	% Barren	Total Seed	Full Seed	% Barren
1504 (R)	32	0	100	2081	583	72
1501 (I)	44	10	75	NA	NA	NA
DSP-1973	111	0	100	89	12	86
DSP-RDC	62	0	100	60	4	93
DSP-FAY	7	0	100	134	24	82

high degree of self-sterility (Nielsen & De Muckadeli, 1954). However, there was a significant amount of variability of self-fertilization between individual trees and from year to year in the same individual. Rates as high as 40 percent were reported, although the average was 13.6 percent and the majority of trees included in the study had between 0 and 5 percent self-fertilized nuts. It is possible that the higher rates of self-fertilization reported were actually due to a contaminating pollen source. During the time this study was performed, the molecular techniques that would be required to rule out a pollen contaminant had not yet been developed. To address the question of self-fertilization in *F. grandifolia*, self-pollination control experiments were set up by placing the pollination bag over the branch and instead of removing it to apply pollen, the bag was kept in place until after pollen release was complete. The only pollen that would be available to fertilize the flowers would be from the staminate flowers on the same branch. Only two of the parents used in the cross-pollinations, 1504 and 1501, had enough accessible branches to set up additional self-fertilization experiments. Therefore, to try to get an idea of the self-fertility in *F. grandifolia* in general, three trees located in Delaware State Park were included in these studies, DSP-1973, DSP-RDC, and DSP-FAY. No occurrence of beech scale has ever been reported at Delaware State Park, so the resistant or susceptible phenotype of these trees is unknown, although they are most likely susceptible. Of the five trees tested for the ability to self-fertilize, only one tree, 1501, produced any full seed (Table 4). Furthermore, of the 10 full seeds obtained from 1501, only 2 germinated (data not shown). The remaining 4 trees produced 100 % barren seed. In each case, open-pollinated seed from the same tree yielded at least a portion of sound seed. In the case of 1504, the open-pollinated seed was 28 percent full. The open-pollinated seed from the trees in Delaware State Park produced between 7 and 18 percent sound seed (Table 4). Our observations of the self-fertilization experiments revealed an abundance of pollen was available for fertilization within the pollination bags. The extremely low occurrence of sound seed in the self-fertilization experiments, especially when compared to open-pollinated seed, is therefore likely due to a high degree of self-sterility in the American beech.

Discussion

Both half-sib and full-sib families were successfully generated for use in genetic studies. The seedlings are currently being tested for their resistant/susceptible phenotype through the use of insect challenge experiments (Houston, 1982). Based on estimates of between 1-2 % of beech trees displaying resistance to the scale insect, the half-sib family from the ME parent might yield one or two resistant trees. It is possible that the percent of resistant trees may be higher in controlled crosses between resistant parents. For thorough genetic mapping studies, a larger full-sib population will be required. The data presented here have provided the fundamental information needed to increase the numbers of controlled cross-progeny. Our data also indicate that there is a high degree of self-sterility in American beech. However, the degree of self-sterility can vary between individuals so it is possible that higher levels of self-fertilization can occur. Because we did not emasculate the branches used in our controlled cross-pollinations, some of the seed that was obtained may be from self-fertilization. It is also possible that the two plants obtained from the self-fertilization of 1501 were the result of contaminating pollen entering the bag. In order to confirm the parentage of each of our cross-progeny, and to more definitively address the issue of self-sterility, we are currently developing DNA-based markers. These markers will also allow a more definitive approach to determining the percent of self-fertilization that may have occurred in our half-sib and self-fertilized progenies.

Acknowledgment

The authors thank the Michigan Department of Natural Resources and the Maine Department of Forestry for their cooperation. This work was funded in part through the USDA Forest Service FHM Evaluation Monitoring Program.

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THE ROLE OF XYLEM SAP ABSCISIC ACID IN LEAF ABSCISSION OF DROUGHTED SEEDLINGS OF ISOHYDRIC AND ANISOHYDRIC TEMPERATE DECIDUOUS ANGIOSPERMS

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ABSTRACT.—Post-recovery patterns of leaf abscission were studied in relation to peak-drought xylem sap ABA concentration [ABA] for seedlings of five woody species [drought-sensitive, isohydric black walnut, black willow and eastern cottonwood, and drought-tolerant, anisohydric white oak and black oak]. The potential role of xylem sap [ABA] as a signal of the need for leaf area adjustment to reduce water loss was the object of study. Potted seedlings were allowed to deplete soil moisture and then were rewatered. At peak drought and during recovery, predawn and midday leaf water potential, and leaf abscission were measured. Xylem sap [ABA] was measured in well-watered and droughted seedlings before rewatering of the latter. Water potentials and xylem sap [ABA] recovered rapidly after rewatering. Long-term (six week) leaf abscission increased with water stress and xylem sap [ABA] measured at peak drought in isohydric species. However, anisohydric, drought tolerant oak species lost leaves only under severe water stress, although xylem sap [ABA] was elevated at peak drought. Xylem sap [ABA] may serve as a signal for leaf area adjustment in isohydric, but not anisohydric, species.

Introduction

While the recovery phase of leaf-level response to water stress is physiologically and ecologically significant, it has received far less attention than the period of developing water stress. The degree and rate of return of processes to pre-stress levels varies among species and can have a substantial impact on plant fitness (Quick and others 1992, Roe and others 1995). An important response to water stress that is often manifested after relief from water deficits is leaf senescence and abscission. Drought deciduousness is often linked with extreme environments, viz. deserts and Mediterranean regions (e.g., Addicott 1991, Kozlowski and others 1991), but also is obvious in certain species in temperate forests. Studies of black walnut (*Juglans nigra* L.), white oak (*Quercus alba* L.), post oak (*Q. stellata* Wangenh.), sugar maple (*Acer saccharum* Marsh.) and poplar (*Populus* spp.) (Parker and Pallardy 1985, Pallardy and Rhoads 1993, 1997) showed that post-drought leaf abscission is related to the peak level of plant water stress experienced. Additionally, field observations anecdotally suggest that drought-accelerated leaf abscission also occurs in other riparian species (e.g., willow (*Salix*), other poplars). While such abscission may be considered an effective avoidance mechanism (protecting critical meristem tissues in buds and cambium), it represents an obvious disadvantage for carbon gain and growth. These same taxa are characterized as light-demanding species and loss of leaf surface area would place them at a distinct competitive disadvantage in height growth capacity.

The concentration of xylem sap abscisic acid ([ABA]) in water-stressed woody plants often increases to very high levels (e.g., Loewenstein and Pallardy 1998a, b), suggesting that ABA from the roots might play a role in drought-induced senescence and abscission (when severe drought develops and very high ABA levels accumulate in xylem sap). While control of senescence and abscission is complex, involving actions and interactions of at least three plant hormones (ABA, ethylene and IAA) as well as plant developmental stage, there is a substantial body of literature showing that ABA plays an important role in these responses (reviewed by Nooden and Leopold 1988). Abscisic acid levels in leaves increase during senescence (Plummer and others 1991), and exogenous ABA promotes leaf senescence and

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Table 1.—Morphological characteristics (mean \pm SE) of seedlings prior to initiating the experiments.

	height (cm)	diameter (mm)	number of leaves/leaflets
black willow	65.44 \pm 2.16	3.31 \pm 0.09	73.62 \pm 4.65
cottonwood	40.47 \pm 5.67	5.67 \pm 0.13	28.62 \pm 0.77
black walnut	22.75 \pm 4.81	4.81 \pm 0.10	41.07 \pm 1.09
white oak	10.21 \pm 0.64	2.75 \pm 0.11	6.65 \pm 0.56
black oak	37.27 \pm 2.03	6.70 \pm 0.18	22.34 \pm 1.25

abscission (Plummer and others 1991, Suttle and Abrams 1993, Lin and others 1999). Although there have been a few reports of leaf abscission in plants in which xylem sap [ABA] has been elevated by water stress or introduction of exogenous ABA into the xylem stream (e.g., Chen and others 1997), there has been no study of the relationship between drought-induced elevation in xylem sap [ABA] levels and subsequent senescence/abscission responses of woody plants. Here we report on species variation in xylem sap ABA levels under drought as they relate to subsequent plant senescence/abscission responses, including both isohydric and anisohydric species (*sensu* Tardieu 1996). Isohydric species must maintain relatively high plant water potential (ψ) to avoid injury during drought while anisohydric plants exhibit substantial depression of ψ during drought without lasting impairment.

Methods and Materials

Plant Materials

Black walnut, black oak and white oak seeds from open-grown trees were obtained from the Missouri State Tree Nursery (Licking, MO, U. S. A.). Hardwood cuttings were collected from black willow (*Salix nigra* Marsh.) and eastern cottonwood (*Populus deltoides* Bartr. ex Marsh) trees near Columbia, MO, U. S. A. (38° N lat., 92° W long). Black willow, eastern cottonwood and black walnut are typically found in riparian habitats, are drought-sensitive, and exhibit isohydric patterns of water relations; white and black oaks are abundant in more xeric habitats, are more drought tolerant, and are anisohydric (Bahari and others 1985, Burns and Honkala 1990, Loewenstein and Pallardy 1998a,b). Seeds and cuttings were planted in 2.5 L (10.5-cm diameter) plastic pots containing a 2:1:1 mixture of sand, peat moss, and silt-loam soil. A top-dressing of slow-release fertilizer (14-14-14 N, P, K, Osmocote, Sierra Chemical Co., San Milpitas, CA, U.S.A.) was applied soon after plants were established and modified half-strength Hoagland's solution was applied approximately every two weeks. Plants were grown in an evaporation-cooled greenhouse under 50% shade. Photoperiod was extended to 14 h with sodium vapor lamps during short-day periods of the year. Experiments were conducted separately for each species and were initiated when plants were three to six months old.

Experimental Procedure

Prior to beginning treatments the height, diameter and number of leaves of all plants was determined (Table 1). Eighty-one plants per species were randomly assigned to either a drought treatment (D, n=24), a drought with recovery treatment (R, n=42) or a well-watered control (W, n=15) and then plants were randomly arranged on a greenhouse bench. Drought was imposed on plants in treatments D and R by withholding water. Plants were randomly selected from all three treatments throughout the dry-down period for assessment of predawn and midday leaf water potentials (ψ_{pd} and ψ_{md} , respectively) and midday stomatal conductance (g_s). Water potential of one leaf per plant was estimated before dawn and at midday with a pressure chamber (Pallardy and others 1991) and g_s was measured with a steady-state porometer (LI-1600, LI-COR, Inc., Lincoln, NE, U.S.A.) Upon completion of these measurements, plants selected for that sample day from treatment R were reirrigated and leaf abscission was monitored for six weeks. Xylem sap was collected for determination of xylem sap [ABA] from the D and W plants. (As collection of xylem sap involves excising the shoot, [ABA] data could not be gathered for the plants which were monitored for leaf abscission.) A total of 42 plants per species were monitored for leaf abscission over the six-week recovery period. Total length of soil drying ranged from 5 days (eastern cottonwood), 1 week (black willow), 2 weeks (black oak), 4 weeks (black walnut)

to 6 weeks (white oak). The experiment was conducted over a two year period (1997-1998) at times of the year that varied from March to August. Leaf area was measured with a LI-3000 Leaf Area Meter fitted with a belt conveyor (LI-COR, Inc. Lincoln, NE, U. S. A.). A spot of water-based paint was placed on each leaf (with different colors for neighboring plants) to facilitate monitoring of leaf abscission. The paint had no obvious adverse effect on the leaves.

Analysis of ABA

Concentration of ABA in the xylem sap was determined by enzyme-linked immunoassay (ELISA) (Sigma, St. Louis, U.S.A.) (Loewenstein and Pallardy 1998a). Dilution/spike recovery tests (Jones 1987) indicated the presence of nonspecific interference in the sap of all five species. Lyophilization sufficiently reduced this interference that quantification of ABA in samples was possible (Loewenstein and Pallardy 1998a). Lyophilized samples were reconstituted with 25 mM Tris buffered saline adjusted to pH 7.5 with reagent-grade HCl. All samples were run in duplicate and ABA standards were included, in triplicate, in each assay for the construction of a standard curve.

Statistical Analysis

Species variation in daily water potential and g_s data obtained during recovery from water stress were subjected to analysis of variance. Relationships between leaf abscission and peak drought Ψ_{pd} and xylem sap [ABA] were subjected to analysis using Pearson correlation coefficients calculated separately for each species.

Results

Water Potentials and Stomatal Conductance

In general, Ψ_{pd} and Ψ_{md} recovered rapidly after water-stressed plants were provided moist soil (data not shown, see Loewenstein and Pallardy 2002). Predawn Ψ_l of water-stressed plants recovered to at least control levels within one day of rewatering for all five species. Further, significant over-recovery of Ψ_{pd} was observed for rewatered black willow and eastern cottonwood plants on Day 1. Mean Ψ_{pd} of rewatered black oak plants on Day 1 was somewhat lower than that of control plants (-0.32 MPa vs. -0.18 MPa, respectively), but the difference was not statistically significant in this species. For all species, recovery of g_s was somewhat more delayed after rewatering than was that of Ψ , but there were no significant differences in g_s between previously droughted and well-watered plants by the second day after rewatering had begun.

Leaf Responses to Water Stress and Its Relief

While water stress generally induced some leaf abscission in all species, in black willow and eastern cottonwood severe water stress also resulted in retention of necrotic leaves. Water stress, if severe enough, may overtake and prohibit the carefully coordinated program of senescence leading to abscission (Dangl and others 2000). As these leaves are not functional, they are collectively referred to hereafter as "abscised." True abscission occurred only after plants had been rewatered. In the isohydric riparian species leaf abscission measured after 6 weeks at high soil moisture generally increased monotonically as the minimum Ψ_{pd} to which the plant had been exposed declined (Figs. 1-3). Abscission commenced at relatively high Ψ_{pd} values (> -0.5 MPa), especially in eastern cottonwood and black willow. Isohydric black walnut plants reached minimum Ψ_{pd} values that were somewhat higher and lost proportionally less leaf area after 6 weeks compared to other species (cf Fig. 3 with Figs. 1 and 2). However, the relationship between leaf area loss and Ψ_{pd} was still statistically significant ($r=0.62$, $p < 0.01$) in black walnut. In contrast with isohydric species, leaf abscission of black oak seedlings occurred only after Ψ_{pd} had declined to -5 MPa or less (Fig. 4). There was no detectable pattern of leaf abscission in response to Ψ_{pd} in white oak seedlings over the range of water stress induced in the experiment (minimum Ψ_{pd} -3.6 MPa), and some seedlings exhibited abscission that appeared unrelated to water stress (Fig. 5). For example, a plant which exhibited Ψ_{pd} of -0.65 MPa lost 75 percent of its leaves while one subjected to Ψ_{pd} less than -3.4 MPa lost only one of 16 leaves (data not shown). Shoot die-back occurred in severely stressed black willow and, to a lesser degree, in eastern cottonwood plants.

The relationship between Ψ_{pd} and leaf loss in black willow and eastern cottonwood depended on the time at which observations were made after rewatering. In these species, at Ψ_{pd} values indicative of mild- to moderate water stress (>-1 MPa), leaf loss measured two weeks after rewatering was

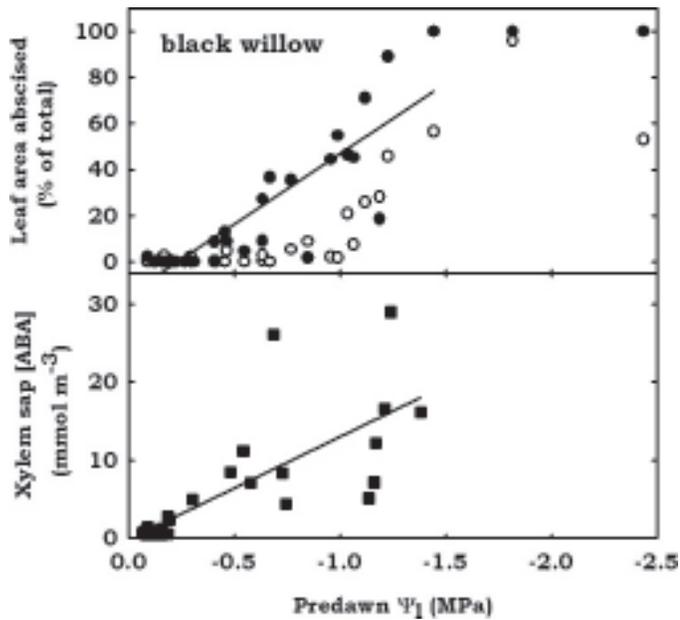


Figure 1.—Relationship between percent leaf abscission (top panel) and xylem sap ABA concentration (bottom panel) and minimum predawn leaf water potential developed during drought in black willow seedlings. Abscission and xylem [ABA] relationships were constructed from different sets of seedlings. (○)—Leaf abscission after two weeks at high soil moisture; (●)—Leaf abscission after six weeks at high soil moisture. Linear regression line relationships between 6-week abscission and xylem sap [ABA] and Ψ_{pd} are statistically significant ($p < 0.05$). Mean total leaf area (abscised and retained leaves) after six weeks at high soil moisture was 545.7 ± 22.1 (SE) cm^2 .

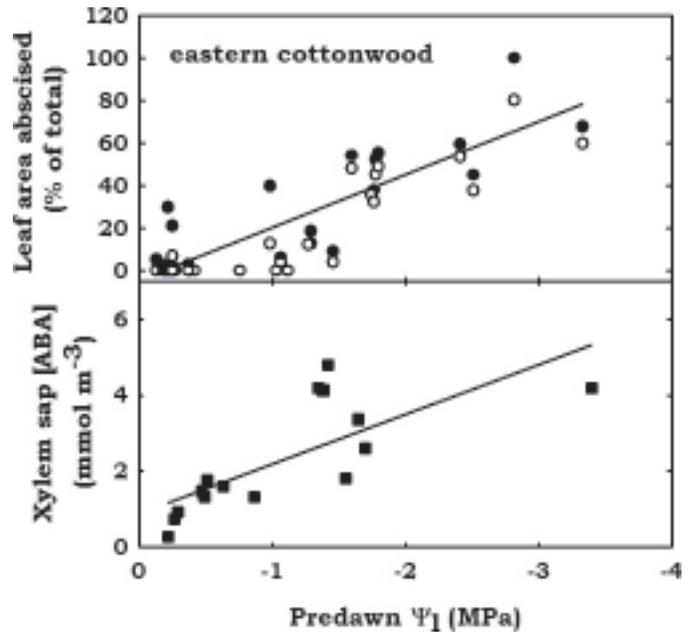


Figure 2.—Relationship between percent leaf abscission (top panel) and xylem sap ABA concentration (bottom panel) and minimum predawn leaf water potential developed during drought in eastern cottonwood seedlings. Abscission and xylem [ABA] relationships were constructed from different sets of seedlings. (○)—Leaf abscission after two weeks at high soil moisture; (●)—Leaf abscission after six weeks at high soil moisture. Linear regression line relationships between 6-week abscission and xylem sap [ABA] and Ψ_{pd} are statistically significant ($p < 0.05$). Mean total leaf area (abscised and retained leaves) after six weeks at high soil moisture was 1106.2 ± 29.4 (SE) cm^2 . Inset illustrates leaf abscission at high Ψ_{pd} .

substantially less than that after six weeks (Figs. 1 and 2). Under more severe water stress ($\Psi_{pd} < -1$ MPa), leaf loss in both species (including dead leaves that remained attached) was more rapid.

Xylem Sap ABA and Leaf Abscission

Because sampling for xylem sap [ABA] required sacrifice of the shoot, the response to water stress of this variable and leaf abscission was measured in different groups of plants. The relationships between xylem sap [ABA] and leaf abscission were thus examined by plotting both of them against measured Ψ_{pd} (Figs. 1-5).

In the isohydric riparian species, which were more prone to drought-induced leaf abscission, xylem sap [ABA] and leaf loss after six weeks at high soil moisture both increased with decreasing Ψ_{pd} . In black willow, black walnut and eastern cottonwood, leaf loss measured at six weeks and [ABA] were significantly correlated with predawn Ψ ($r = 0.60-0.84$; $p < 0.01$). However, short-term (2-week) observations indicated that early leaf abscission in these species was not as well correlated with Ψ_{pd} (or xylem sap [ABA]) (Figs. 1-3). The situation in the oaks was quite different. In both oak species, xylem sap [ABA] increased progressively with decreasing Ψ_{pd} , but there was no leaf abscission even as xylem sap [ABA] increased to very high levels (Fig. 4, 5). The inability to express sufficient sap for analysis at the very low levels of Ψ associated with abscission in black oak precluded inspection of xylem sap [ABA]-leaf abscission relationships in this region.

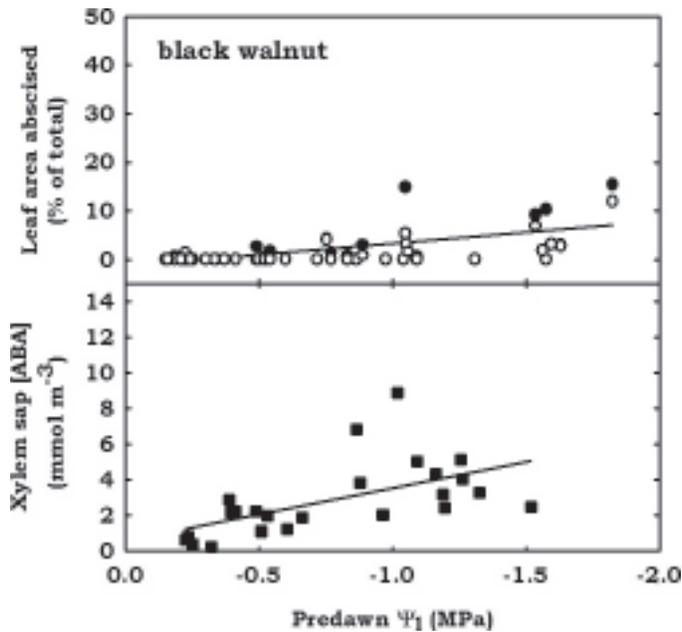


Figure 3.—Relationship between percent leaf abscission (top panel) and xylem sap ABA concentration (bottom panel) and minimum predawn leaf water potential developed during drought in black walnut seedlings. Abscission and xylem [ABA] relationships were constructed from different sets of seedlings. (○)—Leaf abscission after two weeks at high soil moisture; (●)—Leaf abscission after six weeks at high soil moisture. Linear regression line relationships between 6-week abscission and xylem sap [ABA] and Ψ_{pd} are statistically significant ($p < 0.05$). Mean total leaf area (abscised and retained leaves) after six weeks at high soil moisture was 1069.0 ± 37.1 (SE) cm².

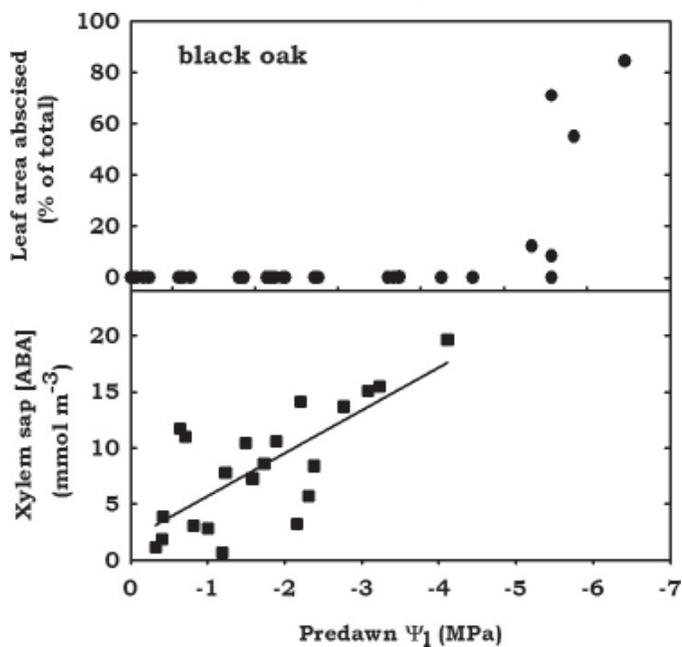


Figure 4.—Relationship between percent leaf abscission (top panel) and xylem sap ABA concentration (bottom panel) and minimum predawn leaf water potential developed during drought in black oak seedlings. Abscission and xylem [ABA] relationships were constructed from different sets of seedlings. Linear regression line relationship between xylem sap [ABA] and Ψ_{pd} is statistically significant ($p < 0.05$). Mean total leaf area (abscised and retained leaves) after six weeks at high soil moisture was 1704.2 ± 69.8 (SE) cm².

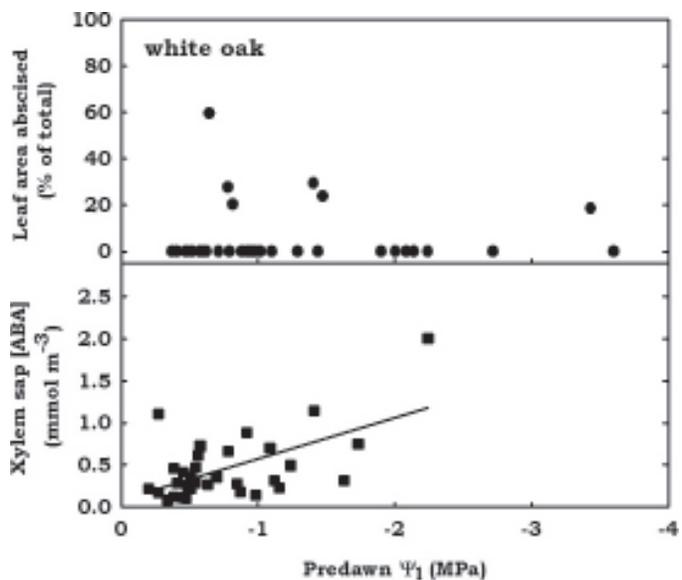


Figure 5.—Relationship between percent leaf abscission (top panel) and xylem sap ABA concentration (bottom panel) and minimum predawn leaf water potential developed during drought in white oak seedlings. Abscission and xylem [ABA] relationships were constructed from different sets of seedlings. Linear regression line relationship between xylem sap [ABA] and Ψ_{pd} is statistically significant ($p < 0.05$). Mean total leaf area (abscised and retained leaves) after six weeks at high soil moisture was 177.7 ± 9.0 (SE) cm².

Discussion

Two consistent, contrasting patterns of leaf abscission response to water stress were exhibited. In isohydric riparian species, a progressive increase in abscission was observed as water deficits exceeded mild levels ($\psi_{pd} < -0.5$ MPa). In contrast, there was very little leaf abscission in anisohydric oak seedlings unless water deficits were quite severe. The current and future carbon costs of leaf loss in the isohydric, riparian species may be one reason why they are restricted to habitats where abscission-inducing levels of water deficits are less frequent. On the other hand, the likelihood that most terrestrial habitats, even riparian ones, will experience severe soil water deficits at some point during the relatively long life of a forest tree suggests that the leaf abscission (and its consequent preservation of meristem hydration) protects tissues that are vital to long-term survival. Meteorological droughts are both frequently punctuated by sporadic rainfall and self-reinforcing (Dirmeyer 1994, Dole 2000); hence, abscission responses of temporarily rehydrated plants to a developing drought prepare the plant for the higher likelihood of prolonged soil moisture deficits.

The relationship between abscission and xylem sap [ABA] depended on species and time after rewatering had been resumed. In riparian species, the patterns of increase in abscission and xylem [ABA] with decreasing ψ_{pd} were similar if long-term (six week) leaf loss was considered. However, when abscission was assessed two weeks after rewatering, it was poorly correlated with both ψ_{pd} and xylem [ABA] unless water stress was severe. The delay in abscission in moderately-stressed plants is not altogether surprising because, once initiated, the orderly cellular disassembly of senescence which terminates in leaf abscission can be an extended process (Becker and Apel 1993). Under severe stress, leaves were prone to die before completion of senescence (Dangl and others 2000) but were here classified as functionally “abscised” even though they were still attached to the plant. In anisohydric oak species leaf abscission was lacking at all but the most severe levels of water stress, although xylem [ABA] increased linearly with declining ψ_{pd} .

The responses shown in the present study suggest that in isohydric species xylem sap [ABA] could serve as a signal of water stress severity and the degree of leaf area adjustment required. Xylem feeding or spray applications of ABA accelerate leaf and fruit abscission in numerous species, although the exact mechanism has not been determined. Abscisic acid induces senescence-related genes in daylily (*Hemerocallis* hybrid cv. ‘Stella D’Oro’) petals and detached leaves of *Arabidopsis thaliana* (Park and others 1998) and has a weak, but detectable, influence on a protease regulatory subunit that is upregulated during leaf senescence in *Arabidopsis* (Nakashima and others 1997, Kitsaki and others 1999). Exogenous ABA also reduced levels of free-radical scavenging enzymes in detached, senescing rice leaves (Lin and others 1999). In anisohydric species such as upland oaks, leaf abscission is apparently unresponsive to xylem [ABA] except, perhaps, at a threshold value indicative of severe water stress.

Acknowledgements

Research supported by the U.S.D.A. National Research Initiative under Grant USDA-CSREES 96-35100-3215.

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ELECTROLYTE LEAKAGE FROM STEM TISSUE AS AN INDICATOR OF HARDWOOD SEEDLING PHYSIOLOGICAL STATUS AND HARDINESS

Barrett C. Wilson and Douglass F. Jacobs[†]

ABSTRACT.—It is important to identify rapid and accurate methods for assessing hardwood seedling quality and physiological status. Evaluation of electrolyte leakage (EL) from plant tissues is promising for this purpose. It has successfully predicted the physiological status of conifer seedlings and has been used experimentally on European hardwood species. Three species of hardwoods, northern red oak (*Quercus rubra* L.), black walnut (*Juglans nigra* L.), and black cherry (*Prunus serotina* Ehrh.), were evaluated for cold hardiness after being subjected to three storage methods (freezer, cooler, ambient) of varying duration. Higher EL values at longer durations represent a loss of dormancy and increase in cell damage over time. For all species, an increase in EL over time corresponded to a decrease in the number of days required for budbreak under greenhouse conditions. While trends were similar for all species and storage methods, DBB and EL levels did not appear to be related to greenhouse height growth. Further study is needed to assess the viability of EL as a predictor of seedling hardiness and quality for commonly produced hardwoods.

The assessment of seedling quality is an important component of reforestation and afforestation programs. Use of low quality seedlings may result in low growth rates and plantation establishment failure (Sampson et al. 1996). It is important to identify high quality stock that has the potential for vigorous root and stem growth. New roots are more efficient in supplying the newly planted seedling with water, which is essential for withstanding transplant stress (Stone 1955, Nambiar et al. 1979, Rietveld 1989, Larcher 1995). Vigorous seedlings grow at increasingly faster rates compared to seedlings of low vigor (Vyse 1982; Burdett 1990) and growth differences evident shortly after transplanting may be maintained over the life of a planting (Paterson and Fayle 1984). Plantings established with high quality seedlings may ultimately require less maintenance (Paterson and Fayle 1984, Burdett 1990) and are more resistant to insect and disease stress (Schoeneweiss 1981, Cordell et al. 1989, Landis et al. 1989; Sutherland et al. 1989). Lifting procedures, post-lifting cold storage, and choice of planting site are important considerations in maintaining a high level of seedling quality. Transporting seedlings from the nursery to storage and then to the planting site subjects them to numerous physical and environmental stresses that lead to a reduction in vigor needed for establishment success (McKay 1996, Maki and Colombo 2001). Seedling vigor is associated with physiological activity of healthy tissue (Sampson et al. 1996). Therefore, the need for practical and efficient methods to monitor changes in seedling physiological status is apparent.

The primary focus of most quality assessment research has been on conifer species, as these are produced in the greatest quantities (Moulton and Hernandez 2000). However, hardwood seedling demand is increasing, primarily as a result of ecological restoration and conservation practices (King and Keeland 1999, Stanturf et al. 2000). Consequently, research seeking to improve quality assessment methods for hardwood seedlings is increasing in significance.

Cold hardiness assessment is one method for evaluating quality. It provides a measure of dormancy status (Ritchie 1984); predicts the ability of seedlings to withstand stresses associated with lifting, storing, and planting (O'Reilly et al. 1999); and provides an indication of field performance potential

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(Pardos et al. 2003). Physiological methods of testing cold hardiness are also rapid (McKay 1992), allowing for timely management decisions in nursery operations.

Many methods have been employed for evaluating cold hardiness of plants. Visual methods such as whole plant freeze tests and shoot tissue browning tests have been effective (Timmis 1976; Liu et al. 1998), but are more time consuming. Water relations (Ameglio et al. 2001), bud mitotic activity (Calme et al. 1994), abscisic acid concentration (Li et al. 2003), soluble sugar concentration (Tinus et al. 2000), and chlorophyll fluorescence (Rose and Haase 2002) are among the many physiological indicators used for conifers. Measurement of these indicators, while more rapid, has seen limited application to hardwood forestry. The electrolyte leakage (EL) method was chosen for this study. It has been used extensively in conifer research, where it was shown to be a reliable approach to determining hardiness (Colombo et al. 1995, Bigras 1997, Tinus and Burr 1997). It is expected that similar results can be achieved with hardwood seedlings.

Electrolytes are contained within the membranes of plant cells. These membranes are sensitive to environmental stresses such as chilling and freezing conditions. Cold temperatures reduce enzymatic activity, alter metabolism, and decrease the photosynthetic capacity of plant tissues (Dubey 1997). In plant membranes, these changes are often associated with increases in permeability and loss of integrity (Campos et al. 2003). Unstressed, undamaged plant cells maintain electrolytes within the membrane. As the cells are subjected to stress, electrolytes leak into surrounding tissues. An estimation of cell damage and hardiness can be made by comparing the conductivity of the leaked contents from injured and uninjured tissues in water (Mattsson 1996, McNabb and Takahashi 2000).

For conifers, needles (Burr 1990) are the most commonly sampled tissues for EL. Sampling of leaves is particularly applicable to evergreens, which can be collected and tested throughout the dormant period. Roots (McKay 1992) and stem tissues (Colombo et al. 1995) have also been used. The efficacy of EL in predicting hardiness has resulted in its use in operational practice at some nurseries, particularly for determining lifting windows and storability (Tinus 1996). For hardwoods, relatively little information is available. Most hardwood EL research in the past was performed using roots of European species (Edwards 1998; McKay et al. 1999; O'Reilly et al. 2001). Important hardwood species of eastern North America have not been extensively studied. In addition, the impacts of different storage regimes on hardwood seedling EL and physiological status have not been widely published. Therefore, the objectives of this study were to 1) evaluate EL from stem tissues as a method for estimating hardiness and physiological status in eastern hardwoods and 2) observe changes in physiological status that occur in response to different methods and duration of post-lifting storage.

Methods

1-year-old bare-root northern red oak (*Quercus rubra* L.), black walnut (*Juglans nigra* L.), and black cherry (*Prunus serotina* Ehrh.) seedlings of bulk seed origin were hand-lifted from Vallonia Nursery (Vallonia, IN, Indiana DNR Division of Forestry) on 14 November 2002 and divided among two storage treatments: cold storage (2°C) and freezer storage (-2°C). A third treatment consisted of non-lifted seedlings of each species remaining in one of three respective nursery beds to receive ambient environmental conditions during the storage period. Seedlings were not specifically selected for uniform size, with initial heights ranging from 23-90 cm for northern red oak, 13-75 cm for black cherry, and 24-105 cm for black walnut. All seedlings were grown under standard nursery practices and selected from a small section of one of three beds (one bed per species). For each species, seedlings were bundled with moistened peat moss into two kraft paper rolls, with one bundle assigned to the cold storage treatment and one to the freezer storage treatment. The bundles were then immediately transported to Purdue University, West Lafayette, IN for storage. Cold storage was in a thermostatically controlled walk-in cooler and freezer storage was in a Conviron PGR-15 (Controlled Environments Ltd., Winnipeg, MB) growth chamber. Beginning January 2003 and ending April 2003, seedlings of each species were removed from each of the storage regimes at monthly intervals (28 January, 25 February, 27 March, 26 April). Removal dates corresponded to four storage durations. Weather data for the Vallonia Nursery area during the storage period are shown in Table 1.

Table 1.—Weather conditions for Seymour, IN (24 km from Vallonia State Nursery) during the period from 15 November 2002 to 26 April 2003.

Month	Temperature (°C) ¹	Precipitation (cm)	Snow cover (days)
November ²	2.86	1.04	0
December	0.49	6.53	6
January	-4.75	4.09	6
February	-2.93	13.28	20
March	6.59	3.35	3
April ³	11.77	7.90	0

¹Daily average

²15 November–30 November

³1 April–26 April

For each species on each removal date, five trees per storage method were potted into 4-gallon Treepots™ containers (Stuewe and Sons, Inc., Corvallis, OR) using Scotts Metro-Mix 366-P (Scotts Co., Marysville, OH) and placed in a greenhouse at Purdue University's Horticulture and Landscape Architecture Plant Growth Facility. The exception was ambient trees in January, which could not be lifted because of frozen soil. Greenhouse environmental conditions were maintained at 23.9°C day and 17.8°C night, with a photoperiod determined by natural day-length. Water containing a complete fertilizer solution was applied as needed. This solution contained (in mg/liter) 200 N, 29 P, 167 K, 67 Ca, 30 Mg, and micronutrients. Nutrients were supplied from a 1000 mg/liter 15-5-15 commercial fertilizer (Miracle Gro® Excel® Cal-Mag; Scotts Co., Marysville, OH). The pH was adjusted to range from 5.7 - 6.0, with alkalinity reduction achieved via 93% sulfuric acid (Ulrich Chemical, Indianapolis, IN) at 0.08 ml/liter. The number of days to first budbreak (DBB) was recorded for each seedling as an indicator of dormancy status and physiological activity (Englert et al 1993). Measurements of height were recorded at potting and at 30, 60, and 90 days after potting (DAP).

For every storage method-duration combination, a 1 cm long section of stem, cut at both ends, was removed from the top one-third of sixteen seedlings of each species. The sixteen stem samples were individually placed into 20 ml copolymer polypropylene vials (RPI Corp., Mt. Prospect, IL) containing 15 ml of deionized water. Four sample vials were randomly assigned to one of four freeze test treatments: 3° C (control), -10° C, -20° C, and -40° C. Sample vials were capped and the control treatment placed into a 3 C refrigerator where it was not exposed to freezing temperatures. The remaining treatments were placed into a Cryomed 1010 programmable freezing unit (Thermo Forma, Marietta, OH) cooled by liquid nitrogen. Before placement of samples into the freezing chamber, the unit was first cooled to a temperature of 0°C. Once the samples were in the chamber, the rate of temperature decrease was set at -0.25°C/minute. Upon reaching each test treatment, the temperature was held for 20 minutes before decreasing again. After the 20 minute hold time, the respective vials were removed and placed in a refrigerator to thaw overnight. Sample vials of all treatments were then removed from refrigeration to complete thawing at room temperature. After thawing, stem EL (total dissolved solids, ppm) was measured with a HI 9813 portable conductivity meter (Hanna Instruments, Inc., Woonsocket, RI). A measure of maximum conductivity was obtained by placing the vials in a Getinge/Castle autoclave (Getinge USA, Inc., Rochester, NY) for steam sterilization at 110 C for 20 minutes. EL values were expressed as a percentage of maximum conductivity.

Experimental Design and Data Analysis

Descriptions here are for each species. For brevity, -20°C was selected being representative of trends observed during the freeze tests, and only its results are presented and discussed. With the -20°C test, the general experimental design is a split-plot with storage method as the whole-plot factor and duration as the sub-plot factor. Since there was only one replication of each storage method, a storage method main effect or storage method × duration interaction cannot be tested. However, within storage method there is a completely randomized design of 4 durations with 4 seedling replicates per duration for the -20° C test and 5 seedling replicates per duration for the greenhouse growth portion of

Table 2.—Mean (\pm SE) percent electrolyte leakage of stem samples frozen to -20° C (EL) and number of days to first budbreak (DBB) of selected hardwood species. Seedlings were stored by one of three methods and removed after durations ranging from 2 to 5 months.

	Species	Duration	Response/ Ambient ¹	Storage method	
				Cooler	Freezer
Northern red oak	EL	2	27.81 (1.28) ab	20.79 (1.14) b	24.59 (0.48) b
	EL	3	19.87 (1.88) b	24.53 (2.61) ab	24.85 (1.97) b
	EL	4	31.25 (5.86) ab	42.03 (2.11) a	
	EL	5	40.48 (0.77) a	39.15 (5.97) a	40.83 (1.82) a
DBB	DBB	2	42.20 (2.35) a	45.60 (2.56) a	
	DBB	3	32.20 (1.32) a	28.40 (1.12) b	34.20 (3.69) b
	DBB	4	26.60 (4.41) a	29.00 (5.62) b	26.40 (2.38) bc
	DBB	5	0 (0.00) b	12.80 (1.83) c	19.25 (0.75) c
Black cherry	EL	2	30.54 (3.70) a	27.29 (2.59) b	25.18 (1.35) c
	EL	3	40.18 (4.40) a	42.94 (4.26) ab	46.01 (5.23) b
	EL	4	43.25 (3.19) a	62.51 (10.30) a	82.78 (0.96) b
	EL	5	49.02 (3.52) a	47.89 (0.79) ab	56.12 (5.96) a
DBB	DBB	2	16.40 (0.87) a	18.60 (6.60) a	
	DBB	3	8.00 (1.90) a	17.25 (7.42) a	
	DBB	4	0 (0.00) b	9.60 (1.21) a	
	DBB	5	0 (0.00) b	9.00 (1.68) a	
Black walnut	EL	2	28.98 (2.55) b	19.81 (1.52) b	32.94 (1.64) b
	EL	3	20.44 (2.24) b	24.75 (3.19) b	29.51 (1.12) b
	EL	4	43.07 (8.88) ab	24.90 (0.42) b	44.90 (2.27) ab
	EL	5	62.71 (9.86) a	52.79 (8.18) a	50.11 (6.90) a
	DBB	2	—	31.20 (1.56) a	29.20 (5.66) a
	DBB	3	16.00 (0.63) a	17.00 (1.97) b	22.40 (1.47) a
	DBB	4	5.60 (0.51) b	12.80 (1.43) b	25.00 (5.93) a
	DBB	5	0 (0.00) c	5.00 (1.05) c	17.40 (0.75) a

¹For measured responses within each storage method, means followed by the same letter in a column did not differ significantly ($P \leq 0.05$, Tukey method).

the study. Therefore, EL and DBB differences within each storage method that result from changes in duration can be tested. Analysis of variance was performed using SAS statistical software version 8.2 (SAS Institute, Cary, NC). Results are summarized in table format (Table 2). Regression analyses were also performed to assess the relationship of EL to DBB for each species (Table 3).

Results And Discussion

Electrolyte Leakage

All species showed a similar trend when exposed to progressively longer storage durations. This was evident for all three storage methods (Table 2). For northern red oak, stem EL of ambient seedlings was 27.81% after 2 months storage and rose to 40.48% after 5 months. During the same period, stem EL of cooler-stored and freezer-stored seedlings rose from 20.79% to 39.15% and 24.59% to 40.83%, respectively. Regardless of storage method, EL levels increased in response to extended storage duration. Stem EL from ambient black cherry seedlings increased from 30.54% after 2 months storage to 49.02% after 5 months. Cooler-stored black cherry exhibited EL levels ranging from 27.29% (2 months) to 47.89% (5 months). Freezer-stored black cherry showed the greatest increase in stem EL from 2 to 5 months, rising from 25.18% to 56.12%. Again, duration impacted EL in all storage methods. Stem EL of black walnut seedlings left in the nursery beds began at 28.98% after 2 months

Table 3.—Simple linear regression of electrolyte leakage (EL) values at -20°C on number of days to first budbreak (DBB) for northern red oak, black cherry, and black walnut seedlings across all storage methods. Paired values from all durations were used.

Species	R ²	P-value	Regression equation
Northern red oak	0.60	0.0049	DBB = 63.17 – 1.17(EL)
Black cherry	0.14	0.2457	<i>no observed relationship</i>
Black Walnut	0.40	0.0359	DBB = 32.78 – 0.44(EL)

and ended at 62.71% after 5 months. Stem EL of cooler-stored seedlings rose from 19.81% to 52.79%, while that of freezer-stored seedlings rose from 32.94% to 50.11%. The change of EL over the course of storage was smallest for freezer-stored seedlings.

Increasing levels of stem EL with extended storage indicates the seedlings' chilling requirements had been met, resulting in a loss of dormancy and cold hardiness over time. Individual species responded differently to the different storage methods. The response of northern red oak differed little among storage methods. This might be attributed to a greater chilling requirement or higher level of dormancy compared to the other species. Stem EL of freezer-stored black cherry seedlings appeared to be somewhat higher than the other storage methods. This may be a result of a low chilling requirement and an increased sensitivity and susceptibility to freezing temperatures in a storage environment. EL of ambient-stored black walnut seedlings at the end of the 5-month period was 10-12 percentage points greater than that of cooler and freezer-stored seedlings. It may be that some form of cold storage is better able to maintain a level of hardiness for prolonged durations compared to the other two species.

Budbreak

The number of days to budbreak (DBB) was used as an indicator of seedling dormancy status. DBB followed a trend that was opposite that of stem EL (Table 1). DBB decreased substantially with longer storage durations, while stem EL generally increased. For northern red oak, ambient-stored seedlings required the fewest DBB, decreasing from 32.20 to 0 during the storage period. DBB of cooler-stored seedlings was 42.20 after 2 months and 12.80 after 5 months, while that of freezer-stored seedlings ranged from 42.60 to 19.25 during that period.

Black cherry seedlings required few DBB in all storage methods. Seedlings left in the nursery needed only 8 DBB after 3 months and had already leafed out after 4 months. DBB of cooler-stored seedlings decreased from 16.40 after 2 months to 1.20 after 5 months. Freezer-stored seedlings needed 18.60 DBB after 2 months and 9 DBB after 5 months. Although it could not be tested, it appeared that method of storage influenced DBB to some extent. Seedlings stored in the freezer for 4 and 5 months required more DBB than those left in the nursery during the same period. From this data, it would appear that overwinter storage is essential for late planting of black cherry.

DBB of ambient-stored black walnut seedlings decreased from 16 after 3 months to 0 after 5 months. Cooler-stored seedlings needed 31.20 DBB after 2 months and 5 DBB after 5 months, while DBB of freezer-stored seedlings only decreased from 29.20 to 17.40. There were clear differences due to storage duration, with seedlings stored for 2-3 months generally requiring more DBB than those stored for 4-5 months. However, when looking at this effect within each storage method, only ambient and cooler-stored seedlings followed this trend. Freezer-stored seedlings tended to require more DBB than the other storage methods and did not show a great decrease in DBB with increased storage duration.

Overall, DBB decreased with longer storage durations, while stem EL levels decreased. This is consistent with the results of Calme et al. (1994). Fewer DBB and high EL levels both point to an increase in seedling physiological activity and loss of hardiness. A significant negative relationship ($p \leq 0.05$) between EL and DBB was shown for northern red oak and black walnut, but not black cherry

(Table 3). The lack of a relationship in black cherry might be attributed to its behavior in freezer storage. High levels of shoot desiccation, although not quantified, were observed in the freezer. Packaging materials did not completely enclose the seedlings, thereby exposing much of the stem tissue to freezing conditions. Consequently, there was a higher occurrence of terminal bud mortality compared to cold-stored and ambient seedlings. This mortality may have altered seedling budbreak patterns and also resulted in the high EL levels. This could explain seedlings having high EL and high DBB at the same time compared to the other storage methods.

Height Growth

Height growth of all species did not seem to be extensively affected by the different storage regimes (data not shown). Height growth of northern red oak in each storage method appeared to increase over time to a point and then decline. Black cherry seedlings lifted from the nursery at the end of 5 months were already growing and did not survive transplanting to greenhouse conditions. Growth of ambient and cooler-stored black walnut seedlings was similar, while freezer-stored seedlings generally had the smallest height increase. It is difficult to discern relationships between height growth and EL or DBB. It is possible that a seedling's growth response is more closely associated with its inherent genetic capability than either EL or DBB.

Future Considerations

Future hardwood EL studies should be expanded to compare different plant tissues such as roots and buds. Investigating lateral vs. terminal buds or fine vs. coarse roots would also be useful to increasing our understanding of EL in hardwoods. Seedling packaging methods should also be considered, particularly when dealing with sub-freezing conditions. In addition, because the seedlings for this project were germinated from seeds collected over a large geographic range in Indiana, it is likely that genetic variability is responsible for some of the observed differences or lack thereof. Therefore, it may be helpful to reduce this variation by using plant material of known parentage. Rapid methods for measuring cold hardiness and dormancy status will benefit foresters by allowing prompt assessment of a seedling's ability to withstand transportation to field planting sites without excessive desiccation or loss of stored reserves. This knowledge is useful in planning when certain batches of seedlings should be shipped and matching a seedling's physiological status with appropriate planting sites. Information gathered during the course of this preliminary study will be valuable in planning and implementing related projects in the future.

Acknowledgments

The authors would like to thank Jim Wichman, Bob Hawkins, and the staff of Vallonia Nursery for their assistance at the nursery; Charles Michler and Ron Overton of the Hardwood Tree Improvement and Regeneration Center and Taj Mohammed of Purdue University's Department of Forestry and Natural Resources for their help with the EL procedure; Rob Eddy and the staff of Purdue's Horticulture Plant Growth Facility for their assistance in the greenhouse; and Judy Santini of Purdue's School of Agriculture for assistance in statistical analysis.

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RE-EVALUATING THE SIGNIFICANCE OF THE FIRST-ORDER LATERAL ROOT GRADING CRITERION FOR HARDWOOD SEEDLINGS

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ABSTRACT.—Numerous authors have reported on the importance of the number of first-order lateral roots (FOLR) when evaluating the morphological quality of nursery hardwood seedlings. Studies have shown that seedlings with a greater quantity of FOLR outperform seedlings with a lesser quantity of FOLR in the field. However, the FOLR measure may be limited in its ability to quantify root system morphology. Additionally, the number of FOLR is correlated with other morphological characteristics, and few studies have directly evaluated the importance of FOLR relative to other variables for predicting outplanting success. A trial in southern Indiana compared the ability of the initial number of FOLR to predict field performance for three hardwood species (black cherry, white oak, and northern red oak) relative to four other initial morphological variables (shoot height, stem diameter, root volume, and whole plant fresh weight). Regression analyses indicated that regardless of species, FOLR tended to be among the least effective predictors of total height and diameter after one field season. These results indicate that FOLR may provide a less accurate indicator of hardwood seedling morphological quality than other easily-measured variables.

The success of reforestation and afforestation operations may be improved if nurseries produce seedlings with target morphological and physiological characteristics that have been quantitatively linked with outplanting success (Rose et al. 1990). Morphological characteristics may be easily assessed by both nursery workers and field foresters. As such, these characteristics are often used to help evaluate seedling quality. Commonly-measured morphological characteristics include shoot height, stem diameter, and root system size (Rose et al. 1990).

Because shoot growth following planting is most limited by water availability (Burdett et al. 1984), roots must rapidly extend through the soil profile to re-establish root-soil contact and absorb water to minimize transplant shock (Sands 1984). The capacity of roots to do this is likely related to both seedling physiological quality (Ritchie and Dunlap 1980) and root system morphology. Various measurements of nursery seedling root system morphology have been proposed, including root mass, root volume, number of first-order lateral roots (FOLR), root length, and root area index.

Many measurements of root system morphology are either destructive, tedious and time consuming, or both. This generally limits their application to only selective research projects. Two root system measurements that are relatively rapid and non-destructive, and thus may have application in operation, are root volume and number of FOLR. Root volume is measured using the water displacement method (Burdett 1979). Nursery seedling root volume has been directly correlated with reforestation success for Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) in the western United States (Rose et al. 1991a; Rose et al. 1991b; Rose et al. 1997). Carlson (1986) reported that root volume helps determine the potential for water uptake prior to new root growth in loblolly pine (*Pinus taeda* L.). Little research, however, has examined the importance of initial root volume in eastern deciduous tree species. Instead, the majority of research involving root morphology of these species has focused on FOLR (i.e., the number of first-order lateral roots exceeding 1 mm in diameter at junction with the tap root). Numerous studies have reported that hardwood seedlings with more FOLR perform better in the field than those with less FOLR (e.g.,

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Table 1.—Division of seedlings from each of the three species into four categories based on root volume (in cm³, with means in parentheses).

Species	Root volume (cm ³)			
	R1	R2	R3	R4
Black cherry	4-17 (11.6)	18-31 (24.7)	32-56 (41.7)	57-265 (93.6)
Northern red oak	5-19 (15.0)	20-28 (23.9)	29-40 (34.0)	41-144 (59.3)
White oak	6-21 (16.2)	22-31 (26.4)	32-44 (38.3)	42-255 (61.2)

(Kormanik 1986; Teclaw and Isebrands 1993; Thompson and Schultz 1995; Schultz and Thompson 1996; Dey and Parker 1997; Clark et al. 2000; Ponder 2000)).

Though clearly linked to improved plantation establishment, measurement of number of FOLR likely does not provide the most accurate characterization of true root system size. Using the FOLR approach, no distinction is typically made between small and large FOLR. Additionally, lateral root length and the quantity of higher-order lateral roots (i.e., root fibrosity) are not accounted for. As such, root volume may provide a better assessment of seedling root system quality and subsequent outplanting performance for some hardwood species than the commonly used FOLR grading criterion. Additionally, although the significance of FOLR as an indicator of outplanting success for hardwood seedlings has been established, few studies have directly compared the effectiveness of FOLR for predicting seedling field performance to other easily-measured morphological variables. Thus, the objective of this report is to present preliminary first-year data from a study comparing the ability of five morphological variables (height, stem diameter, fresh weight, number of FOLR, and root volume) to predict outplanting success of three hardwood species in the Central Region.

Materials And Methods

In February 2002, nursery-grown (1+0) seedlings of three hardwood species commonly planted in the Central Region, northern red oak (*Quercus rubra* L.), white oak (*Quercus alba* L.), and black cherry (*Prunus serotina* Ehrh.), were obtained from the Indiana DNR State Nursery in Vallonia, IN. Approximately 1,000 seedlings from each species were washed free of soil, tagged, and measured in a lab at Purdue University for height, stem diameter, fresh weight, number of FOLR, and root volume. Seedlings in each species were then grouped into four root volume categories, representative of 25 percentiles within the root volume distribution for the population of seedlings from each species (table 1). Seedlings were stored in bags in a cooler at 2°C while not being sampled and prior to planting.

Seedlings from the resulting 12 treatments (three species × four root volume categories) were then outplanted into a replicated experimental design on a field planting site in southern Indiana at Purdue University's Southeast Purdue Agricultural Center (SEPAC) (39°01'N, 85°35'W) in April 2002. Twenty seedlings from each treatment were planted into each of ten blocks for a total of 2400 seedlings in the experiment. An electronic deer fence was installed immediately following planting and maintained throughout the experiment. Weed control using herbicide application was conducted prior to planting and as needed during the first growing season. Initial field height and stem diameter measurements were assessed on each seedling immediately following planting and were measured again in November 2002, following the first growing season.

Data were first analyzed using Analysis of Variance (ANOVA) to determine if seedling field performance differed among root volume categories. If the effect of root volume was significant ($p < 0.05$) in the ANOVA, Fisher's Least Significant Differences Procedure was used to identify significant differences ($\alpha = 0.05$) among root volume treatments. Regression analyses were used to determine the relative importance of the five morphological variables for predicting plantation establishment success, and coefficient of determination values (R^2), were assessed for each regression. All data was analyzed using SAS Software (SAS Institute Inc, Cary, NC).

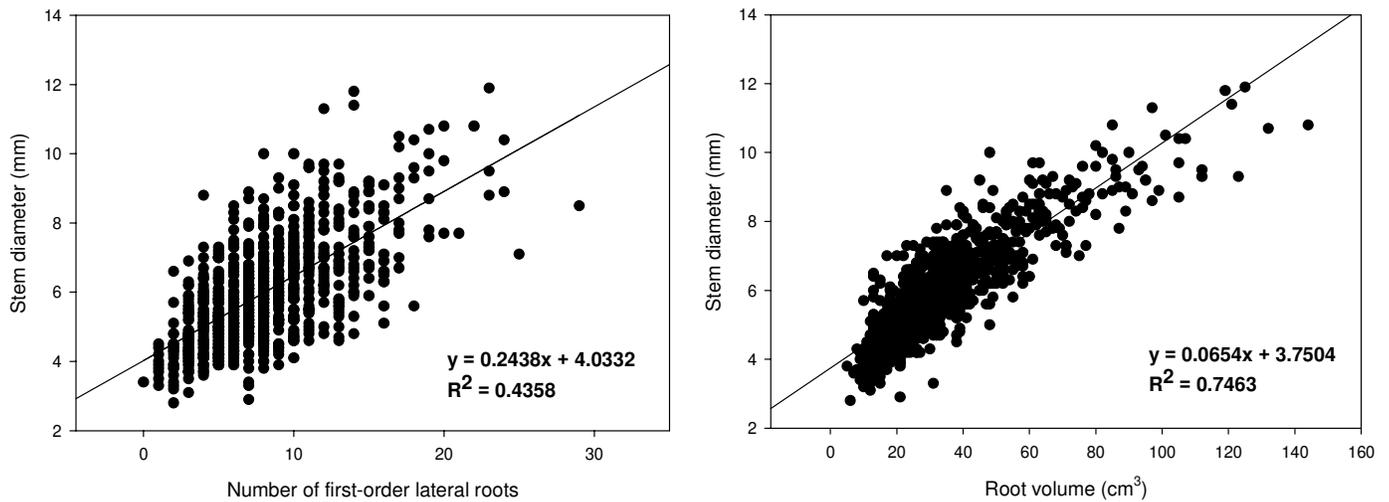


Figure 1.—Regression relationships using FOLR to predict stem diameter (top) and root volume to predict stem diameter (bottom) for northern red oak seedlings. Similar trends for this comparison existed for black cherry ($R^2 = 0.80$ vs. 0.53) and white oak ($R^2 = 0.68$ vs. 0.45).

Results

Analysis of the initial lab morphological data showed that regardless of species, root volume was a better predictor (i.e., higher R^2 values) of all other morphological variables than number of FOLR (data not shown). Figure 1 shows the relationships between both root volume vs. stem diameter and FOLR vs. stem diameter for northern red oak. The same trends were similar for the other two species. We found that seedlings with the same number of FOLR were often very different morphologically (fig. 2).

For each species, seedlings in larger root volume categories had a significantly greater initial field height and diameter than those in smaller root volume categories (table 2). Field height and stem diameter following the first growing season were again significantly larger for those seedlings in larger root volume categories (table 2). First-year height and diameter growth did not differ by root volume category for black cherry (table 2). For white oak, however, height and diameter growth were significantly different among root volume categories, with means increasing with increasing root volume (table 2). For northern red oak, height growth did not differ among root volume categories, but diameter growth was significantly lower for seedlings in R3 and R4 vs. those in R1 and R2 (table 2). Survival was > 88% for all root volume \times species combinations and did not differ among root volume treatments for any species (table 2).

Table 3 shows regression analyses for morphological variables measured in the lab and field performance during the first growing season. In general, R^2 values for lab measurements and total height or diameter were relatively high, while those for lab measurements and height or diameter growth and survival were relatively low. With the exception of height:diameter, FOLR generally had the lowest R^2 value of any lab measurement for predicting total height or total diameter following the first growing season. For black cherry and northern red oak, R^2 values for height or diameter growth and survival were always less than 0.12. Height:diameter was the only significant variable predicting height growth for these two species. No variables were significant for diameter growth of black cherry, and FOLR was the only significant variable for diameter growth of northern red oak. Height:diameter had the highest R^2 value for survival of black cherry, and height was the only significant variable for survival of northern red oak. For white oak, height and root volume had the highest R^2 values for height growth, and height:diameter had the highest R^2 value for diameter growth. Diameter and FOLR were the best predictors of survival for white oak.

Discussion

Seedlings with more FOLR will generally perform better in the field than those with less FOLR. However, it is difficult to conclude that this is truly a function of the number of FOLR, or the correlation of FOLR with other variables. Many studies (e.g., (Kormanik 1986; Schultz and Thompson 1996; Teclaw and Isebrands 1993)) that have examined the ability of FOLR to predict outplanting success have not directly compared the significance of this attribute to other morphological variables. Those that have often report that, as a single variable, FOLR is not as important as others for predicting outplanting performance. Dey and Parker (1997) examined the ability of initial height, stem diameter, and FOLR to predict outplanting performance of northern red oak seedlings underplanted in a central Ontario shelterwood. The number of initial FOLR was not as important as initial height or stem diameter for predicting either second-year height or diameter of these seedlings. Additionally, initial stem diameter was more closely correlated with most measurements of root system morphology for seedlings excavated after two years than was initial height or FOLR. Interestingly, the correlation between initial and second-year FOLR was very weak and non-significant. Kaczmarek and Pope (1993a, b) also reported that initial height and in particular stem diameter, were more strongly related to seedling development in the field than was FOLR.

Though the current results are preliminary, FOLR was generally the weakest initial morphological variable for predicting seedling height and diameter following one growing season regardless of species. This adds further support to the concern that FOLR in itself should not be the sole morphological grading criterion for hardwood seedling. Rather, a combination of morphological characteristics may be necessary to adequately characterize seedling morphological quality (Kaczmarek and Pope 1993a).

Although FOLR is correlated with root system size, root volume likely provides a more accurate measure. Root volume better captures the discrepancy between small vs. large diameter roots, short vs. long lateral roots, and few vs. many second- and third-order lateral roots. We also found that particularly for seedlings with 15 or more FOLR, measurement of root volume was



Figure 2.—Top photo: example black cherry seedlings, each with 12 FOLR and differences in height (91 vs. 76 cm), diameter (9.4 vs. 6.5 mm), fresh weight (98 vs. 50 g), and root volume (59 vs. 31 cm³). Middle photo: example white oak seedlings, each with 16 FOLR and differences in height (27 vs. 30 cm), diameter (11.3 vs. 6.6 mm), fresh weight (93 vs. 47 g), and root volume (62 vs. 33 cm³). Bottom photo: example northern red oak seedlings, each with 11 FOLR and differences in height (48 vs. 37 cm), diameter (7.4 vs. 5.5 mm), fresh weight (81 vs. 40 g), and root volume (53 vs. 28 cm³).

Table 2.—Mean field parameters for different root volume categories in each species. For each field parameter and each species, means with the same letter within a row did not differ significantly at $\alpha = 0.05$.

	Initial height (cm)	Final height (cm)	Height growth (cm)	Initial diameter (mm)	Final diameter (mm)	Diameter growth (mm)	Survival (%)
<i>Black cherry</i>							
R1	56.1 a	83.7 a	27.0 a	4.5 a	9.2 a	4.7 a	88 a
R2	72.5 b	100.8 b	28.4 a	5.7 b	11.1 b	5.3 a	93 a
R3	87.4 c	118.5 c	30.5 a	7.2 c	12.7 c	5.5 a	92 a
R4	107.1 d	131.7 d	24.4 a	9.9 d	14.7 d	4.7 a	95 a
<i>White oak</i>							
R1	17.5 a	32.9 a	15.4 a	4.4 a	5.7 a	1.3 a	98 a
R2	21.2 b	39.8 b	18.6 b	5.3 b	7.0 b	1.7 ab	99 a
R3	25.8 c	48.0 c	22.4 c	6.4 c	8.1 c	1.6 ab	99 a
R4	35.3 d	57.7 d	22.4 c	7.8 d	9.9 d	2.1 b	99 a
<i>Northern red oak</i>							
R1	29.3 a	41.0 a	11.8 a	4.0 a	5.4 a	1.4 b	98 a
R2	34.7 b	48.5 b	13.7 a	4.8 b	6.2 b	1.4 b	99 a
R3	41.9 c	55.7 c	13.8 a	5.7 c	6.9 c	1.1 a	99 a
R4	56.4 d	70.8 d	14.4 a	7.3 d	8.3 d	1.0 a	100 a

Table 3.—Regression coefficients of determination (R^2) values and corresponding significance levels between lab measurements and field measurements (total following one growing season, and growth) for three hardwood species. Bold text identifies the highest significant ($p < 0.05$) R^2 value for each field variable. For significant regressions, the symbol in parentheses indicates whether the slope of the linear regression line was positive or negative.

Lab measurement	Total height	Total diameter	Height growth	Diameter growth	Survival percentage
<i>Black cherry</i>					
Height	0.71 *** (+)	0.37 *** (+)	0.05	0.02	0.04
Diameter	0.75 *** (+)	0.50 *** (+)	0.01	0.001	0.08
FOLR	0.66 *** (+)	0.49 *** (+)	0.001	0.008	0.10 * (+)
Fresh weight	0.67 *** (+)	0.43 *** (+)	0.02	0.008	0.06
Root volume	0.67 *** (+)	0.44 *** (+)	0.02	0.006	0.07
Height:diameter	0.45 *** (-)	0.52 *** (-)	0.10 * (-)	0.09	0.12 * (-)
<i>White oak</i>					
Height	0.87 *** (+)	0.80 *** (+)	0.36 *** (+)	0.17 ** (+)	0.09
Diameter	0.77 *** (+)	0.72 *** (+)	0.31 *** (+)	0.07	0.12 * (+)
FOLR	0.59 *** (+)	0.52 *** (+)	0.23 ** (+)	0.02	0.12 * (+)
Fresh weight	0.83 *** (+)	0.77 *** (+)	0.35 *** (+)	0.10 * (+)	0.10 * (+)
Root volume	0.84 *** (+)	0.77 *** (+)	0.36 *** (+)	0.10 * (+)	0.10 * (+)
Height:diameter	0.48 *** (-)	0.43 *** (+)	0.21 ** (+)	0.34 *** (+)	0.01
<i>Northern red oak</i>					
Height	0.90 *** (+)	0.77 *** (+)	0.02	0.04	0.11 * (+)
Diameter	0.91 *** (+)	0.82 *** (+)	0.08	0.03	0.08
FOLR	0.75 *** (+)	0.57 *** (+)	0.07	0.10 * (-)	0.10
Fresh weight	0.91 *** (+)	0.77 *** (+)	0.06	0.05	0.09
Root volume	0.90 *** (+)	0.76 *** (+)	0.06	0.06	0.09
Height:diameter	0.20 ** (-)	0.13 ** (-)	0.11 * (-)	0.03	0.06

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.0001$

less time consuming than that of number of FOLR. As such, root volume may provide a more effective quantitative measurement of root system morphology than FOLR. In the present study, root volume had higher R^2 values than FOLR for predicting total height and diameter of both white oak and northern red oak following one growing season. Continued research under a variety of environmental conditions will help determine the importance of various morphological variables for predicting hardwood seedling outplanting success.

Acknowledgments

Ron Overton and Bob Karrfalt assisted with lab measurements. Anthony Davis, Amy Ross-Davis, and Barrett Wilson assisted with field measurements. Indiana DNR Vallonia Nursery donated seedlings for this experiment. Financial support for this research was provided by the Mary S. Rice Farm Fund.

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PRELIMINARY FINDINGS AND METHODS OF MEASURING DIFFERENCES IN WOOD COLORATION AS A RESULT OF GIRDLING EASTERN BLACK WALNUT

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ABSTRACT.—Eastern black walnut (*Juglans nigra* L.) is often planted at spacings that require pre-commercial thinning. These thinnings are deemed pre-commercial due to the small diameter of the trees and the low ratio of dark wood to light wood (i.e. heartwood to sapwood). As a consequence of size and wood quality, these thinnings are often an expense rather than a source of revenue. In an effort to increase the value of these thinnings it would be beneficial to increase the ratio of dark wood to light wood. One way to increase the amount of dark wood is through costly processing using steam. However, several non-scientific studies have reported that dark wood can be increased by girdling small trees and allowing them to remain on the stump for a limited period of time. This study was designed to explore this idea. In a black walnut plantation scheduled for thinning, ten trees were randomly selected and double-girdled. At that time, increment cores were taken 30.5 cm above the top girdle. These trees were allowed to remain on the stump for 22 months before they were harvested. Following their harvest the trees were sawn in half to reveal both the dark and light wood over the length of the log. This study found that in seven of the nine logs sampled there was a slight increase in the proportion of dark wood to total wood. However, a t-test failed to identify any differences from samples taken prior to girdling. Further, it was visually evident that none of the seven logs with increases showed a consistent change throughout the length of the log, and therefore it is not likely that girdling will improve the marketability of the log or the market value of the tree. The conclusions of this study suggest that consistency in methods of sampling are required to make valid comparisons related to any movement of wood coloration within a log.

For many Missouri landowners, the idea of planting eastern black walnut (*Juglans nigra* L.) trees for the sawlog or veneer market may seem like a poor investment, since it may take 60 to 80 years before a return is realized. From a financial perspective, the uncertainty and risk involved with a 60 to 80 year investment makes it seem somewhat tenuous. However, even though harvesting black walnut nuts can generate an intermediate cash flow, the prospect of selling marketable sawlogs or veneer is often the deciding factor for choosing black walnut for planting.

In Missouri, markets for small diameter hardwood timber are being explored. By marketing smaller diameter trees, the investment period can be shortened considerably. Small diameter markets may work for native oaks (*Quercus* spp.); however, black walnut's appeal in the market is due to its dark colored heartwood. Small diameter black walnut logs have a large amount of light colored sapwood in relation to the darker colored heartwood (Panshin and DeZeeuw 1980). Because of this ratio of light wood to dark wood, black walnut may be an unlikely candidate for the small diameter market.

Although there are commercial methods, such as steaming (Chen and Workman 1980), that increase the amount of coloration in black walnut, these methods are often too expensive for small sawmills. The search is on for a low cost method of increasing the proportion, or amount, of dark colored wood in small diameter black walnut. The purpose of this study was to determine if girdling and leaving trees standing for a period of time would provide a cheap, effective way to increase the colored wood content in small diameter eastern black walnut trees. When the girdled tree is harvested, the light colored sapwood will theoretically have been darkened to match the color of the heartwood, thus

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Table 1. Location, reference/log number, DBH, and measurements of wood characteristics used in computing the percent change of light to dark wood for the ten girdled trees.

Location	Ref./Log Number	DBH (cm)	Increment Borer Readings			Log Readings ¹			
			C-H	C-B	% H-T	C-H	C-B	% H-T	Avg. %
Row 4 - #22	I	24.6	7.8	10.9	71.6%	8.8 9.6	10.2 11.1	86.3% 86.4%	86.4%
Row 4 - #18	II	24.9	7.1	10.0	71.4%	6.8 7.5	10.2 11.0	66.7% 67.5%	67.1%
Row 4 - #14	III	28.7	6.7	13.7	49.1%	6.6 5.8	13.8 11.3	47.9% 51.1%	49.5%
Row 2 - #16	IV	23.4	7.4	9.5	77.9%	6.7 7.2	8.7 9.3	77.5% 77.4%	77.5%
Row 2 - #18	V	27.2	8.1	11.2	72.3%	10.3 9.9	11.7 13.5	87.6% 73.7%	80.7%
Row 2 - #20	VI	25.9	7.0	12.5	56.0%	10.2 7.9	10.5 12.2	96.9% 64.7%	80.8%
Row 2 - #22	VII	27.4	No reading			10.4 8.9	13.9 12.6	74.7% 70.4%	72.5%
Row 2 - #25	VIII	22.6	7.3	10.2	71.6%	7.0 7.8	9.0 10.5	78.2% 74.7%	76.5%
Row 3 - #19	IX	26.4	6.3	10.0	62.5%	7.9 6.6	10.7 10.3	74.0% 64.2%	69.1%
Row 3 - #24	X	26.9	5.3	10.5	50.5%	7.3 7.1	10.8 11.1	67.1% 64.5%	65.8%

¹ Log measurements are taken for both radii of the colored wood on the open face of the cut log.

C-H : heartwood width in cm for both increment borer and log measurements

C-B : total width in cm for both increment borer and log measurements

% H-T: Percentage heartwood width to total width in cm for both increment borer and log measurements

increasing the quantity of useable wood (Chen, Stokke, and Van Sambeek 1997, 235). If successful, this may facilitate the marketing of trees grown in agroforestry configurations that require thinning prior to the trees reaching a commercial size.

This study was a preliminary test of methods and procedures to answer two main questions. First, does girdling a black walnut tree and allowing it to dry on the stump increase the amount of colored wood? Second, what effect does this method have on the market value of the small diameter eastern black walnut log?

Methods

This study was conducted at the Sho-Neff Plantation near Stockton, Missouri. Sho-neff consists of about 194.2 ha with 123.8 ha of eastern black walnut trees of various ages planted at various spacings. The plantation is divided into 25 areas for research and management.

In 2001, Hammon's Products Company, owner of the Sho-Neff, conducted a thinning to remove approximately one-third of all trees in the plantation. Ten trees (table 1) marked for thinning were chosen at random for this study from Area 16A of Sho-Neff. These trees were native black walnut seedlings, planted in 1976 at a spacing of 6 m x 12 m. Annual crops of soybeans, wheat, and milo had been grown between the rows of trees from 1977- 1988. A traveling gun system irrigated the area for several years while crops were being grown.

On April 4, 2001, these ten trees were double girdled with a chainsaw. Girdling started at about 25.4 cm from the ground. Approximately a 7.6 cm space was left between each girdle. An increment core representing one radii was taken at approximately 30.5 cm above the top girdle at the time of girdling. The trees were left standing until January 2003. On January 24, 2003, the trees were felled, cut to 2.4 m- 3 m lengths, hauled to the Horticulture and Agroforestry Research Center at New Franklin, Missouri and placed inside a storage shed.

On July 8, 2003, a Woodmizer sawmill was used to cut the logs lengthwise. When possible, the increment bore entry points were identified and efforts were made to orient the log such that parallel cuts would approximate the angle of increment sampling. However, due to imperfections in log straightness and loss of some bore entry points, there were times when this was not possible. Slabs of about 2.5 cm thickness were removed from the logs until the approximate center of the log was exposed. By cutting the logs lengthwise, any color change could be examined in terms of consistency along the length of the log. Photographs of the slabs were taken for reference purposes. The slabs were numbered the same as the logs.

To analyze the logs, measurements were taken from the center of the log to the edge of the dark colored wood (C-H) and then from the center of the log to the beginning of the bark (C-B). These measurements were taken using a digital caliper, and were taken at about 30.5 cm above the top giridle (the height of increment bore samples) on the half-log that was left after the slabs were cut off. Each log was measured twice, from the center out to the left and from the center out to the right. After measuring C-D and C-B on both sides of the log, an average estimate of the relationship between dark colored wood to total wood was calculated.

The increment cores were also measured using a caliper and measured from the center to the edge of the dark wood (C-H), and from the center out to the beginning of the bark (C-B). Dark colored wood was calculated as a percent of total wood. A comparison of the C-H/C-B ratio from the increment cores was compared to the average C-H/C-B ratio from the logs to determine if any color change had occurred.

A t-test of pre- and post-girdling mean measurements of colored wood was conducted using SAS (1999) to test whether the differences were significantly different from zero. Pre-girdling measurements used the dark wood measured from the increment borer. Post-girdling used a measure of colored wood that was calculated as an average of the colored wood measured across both radii inside a log. For example, the colored wood for log I measured 7.8 cm (Increment bore), while the inside log measurement used was 9.2 cm (the average of two radiuses across the width of colored wood from the logs interior).

Results

To analyze color movement, differences in the percentage of dark colored wood to total wood were compared from the increment borer samples and the logs. At the time of the girdling, dark colored wood and heartwood were synonymous. However, after waiting the 22 months from the time of girdling, dark colored wood could be a combination of heartwood and wood that may be dark in color due to stain or various other reasons (e.g., movement of the dark heartwood color into the lighter colored sapwood)(Bamber and Fukazawa 1985).

Table 1 shows the ratio of dark wood (C-H) to total wood (C-B) from the increment core readings taken at the time of girdling. Log VII had no reading because a portion of the increment borer sample was missing. This made it impossible to determine where the center of the log would have been. It is interesting to note the variance in the percentage of heartwood in trees that are of the same age. One of the trees had as much as 78% heartwood, while another had as little as 49%. Increment core measurements indicate that most of the trees had about 72% heartwood; however, the mean ratio was approximately 65% heartwood to total wood. The proportion of heartwood to total wood was inversely related to the DBH of the tree, which supports other studies on this topic (Nelson 1976). For example, the tree with 49% heartwood had the largest DBH.

Caliper measurements taken from the cut logs after the trees were allowed to stand for 22 months are also shown in Table 1. Those measurements show that the percent of dark colored wood ranged from 49% to 86%. The mean ratio of dark colored wood to total wood is approximately 73%. This average ratio based on cut logs is 8% higher than might have been expected based on increment core readings taken at the time of girdling.

Table 2. Comparison of the change in dark wood content as a percentage and an actual distance averaged over the width of the log face.

Location	Ref./Log Number	DBH (cm)	Ratios		Color Change	Dark Wood (cm)		Change (cm)	Avg Change (cm)
			Inc. Borer	Log		Inc. Borer	Log ¹		
Row 4 - #22	I	24.6	71.6%	86.4%	14.8%	7.8	8.8	1.0	1.4
							9.6	1.8	
Row 4 - #18	II	24.9	71.4%	67.1%	-4.2%	7.1	6.8	-0.3	0.0
							7.5	0.4	
Row 4 - #14	III	28.7	49.1%	49.5%	0.4%	6.7	6.6	-0.1	-0.5
							5.8	-1.0	
Row 2 - #16	IV	23.4	77.9%	77.5%	-0.4%	7.4	6.7	-0.7	-0.5
							7.2	-0.2	
Row 2 - #18	V	27.2	72.3%	80.7%	8.3%	8.1	10.3	2.2	2.0
							9.9	1.8	
Row 2 - #20	VI	25.9	56.0%	80.8%	24.8%	7.0	10.2	3.2	2.0
							7.9	0.9	
Row 2 - #22	VII	27.4	No reading	72.5%	---	No Reading	10.4		---
							8.9		
Row 2 - #25	VIII	22.6	71.6%	76.5%	4.9%	7.3	7.0	-0.3	0.1
							7.8	0.5	
Row 3 - #19	IX	26.4	62.5%	69.1%	6.6%	6.3	7.9	1.7	1.0
							6.6	0.4	
Row 3 - #24	X	26.9	50.5%	65.8%	15.3%	5.3	7.3	2.0	1.9
							7.1	1.8	

¹ Log measurements are taken for both radii of the colored wood on the open face of the cut log.

Looking at the difference in dark wood to total wood ratios (table 2) for both the increment borer samples taken at the time of girdling and the log readings taken 22 months later, it is evident that some of the logs experienced increases in the amount of dark wood. Logs VI, I, and X showed the most difference in dark wood to total wood ratio. Log VI had a difference in colored wood of nearly 25%, whereas logs I and X had a difference of approximately 15%. However, looking at log VI, there were 7 cm of heartwood at the time of girdling and 22 months later the log had 7.9 cm measuring from one side of the log center and 10.2 cm measuring from the other side. On log VI it is unclear from which side of the log the increment bore sample was taken: however, the side that had 10.2 cm of dark wood was the side that had significant bark loss. The other side had only gained 86 cm in dark wood. Dividing that growth rate by nearly 2 years makes that a 0.43 cm increase in dark wood per year. Table 2 shows the amount of heartwood measured on the increment borer samples compared to the amount of dark wood measured in both directions from the center of the logs 22 months after girdling. Logs VI, V, and X showed the greatest differences in dark wood content, averaging about a 2 cm increase in colored wood. The remaining logs showed very little difference in the amount of colored wood, with some logs even showing a loss in colored wood percentage.

A t-test of pre- and post-girdling mean measurements of colored wood did not identify any differences as significant at an alpha=0.05. For the same alpha, the power of the test was 0.325.

Discussion

In sampling wood to measure color changes, the results of this study would point to the need for a sampling procedure that allows for repeated measurements to be taken within close proximity to one another. This is desirable since tree growth does not occur equally on all sides of the tree. Additionally, due to the growth habits and resulting log character traits (crookedness), the procedure used in this study did not cut the logs exactly parallel to the original increment borer sample location, and thereby may have introduced differences in colored wood measurements due to a lack of concentric tree growth.

By either taking repeat samples from a point close to the original core, or by cutting parallel to the initial core sample, any differences in tree growth and colored wood development may have been better accounted for.

Although several of the logs showed an increase in colored wood, this same increase could possibly have been achieved by allowing the trees to grow for another 22 months. Prior to girdling, these trees grew in diameter at a rate of approximately 1 cm per year. Therefore, it stands to reason that following another two growing seasons, the amount of heartwood in each tree would likely have increased.

A factor that would have improved this study would have been to increase the size of the sample. Additionally, enhancement would have resulted from having controls that were not subjected to girdling. Following a final harvest and processing, these trees would have provided a comparison for the growth rate occurring over a given time and any changes in the amount of their heartwood as a percentage of total wood.

Seven of the nine logs that were compared showed at least a slight increase in the proportion of dark wood to total wood. However, out of the seven logs showing an increase in dark colored wood, it was visually evident that none of the logs had experienced a consistent color change throughout their length. In fact, the only visible color increase in log VI was present in an area approximately 33 cm long and only on one side of the log. This increase in colored wood was in the area of the log where bark had separated and fallen off at some time following girdling, yet prior to final harvest. This study was in no way designed to test the effect of bark removal on wood coloration, but may raise the question as to the response of wood coloration following bark removal.

The inconsistency in the color along the length of the log leads to the second question regarding the economic potential for this type of treatment. If the color is inconsistent along the length of the log, then the log is no more valuable than it was before the treatment was applied.

While value is most often attributed to the amount of heartwood present in black walnut, a trees defects and damage resulting from other agents may decrease its value. This should be an additional concern when deciding whether or not to leave a deadened tree on the stump to cure. From this study, a visible factor that is likely to affect marketability of the logs was the distinct presence of insect damage. Finally, stain, or spalt, was noticed in the wood of these logs even though they had been stored in a dry, protected building after they were felled. This too is typically associated with a reduction in log value.

In conclusion, it appears that increasing dark colored wood in eastern black walnut by girdling the trees and leaving them standing for up to 2-years is not likely to increase the marketability of the log or the market value of the tree. Any increases in dark wood formation would likely be nullified by the increased risk of insect infestation and stain formation. Steaming the logs to increase dark wood formation would be the recommended method of increasing marketable wood. However, research is needed to find cheaper methods of increasing dark colored wood than by steaming, and for improving the utilization of small diameter black walnut logs.

Acknowledgments

This work was funded through the University of Missouri Center for Agroforestry under cooperative agreement AG-02100251 with the USDA ARS Dale Bumpers, Small Farms Research Center, Boonville, AR. The results presented are the sole responsibility of the P.I. and/or MU and may not represent the policies or positions of the ARS.

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TOWARD GUIDELINES FOR HARVEST INTENSITIES AND REGENERATION TARGETS WITH MINIMAL IMPACT UPON RETAINED GENETIC DIVERSITY IN CENTRAL HARDWOOD TREE SPECIES

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ABSTRACT.—There is an urgent need for a coordinated and systematic approach to the *in situ* conservation of the genetic resources of commercially important forest tree species in the Central Hardwoods. Effective *in situ* management of genetic resources would benefit from clear guidelines for how many adult trees can be harvested with minimal impact on allelic diversity. We are constructing a computer model for this purpose, and present preliminary results based upon replicate harvests of a virtual forest stand consisting of 200 adult trees. Our model explores how much regeneration is needed so that there is no more than a 10 percent risk of retaining less than 90 percent of the original allelic diversity. In the absence of regeneration, up to 55 percent of the adult trees can be harvested without exceeding the 10 percent risk level. At higher harvest intensities, locally-derived regeneration is needed to replace the alleles removed from the adult population. When all 200 adult trees are harvested, the 10 percent risk level is not exceeded if there are at least 116 regenerants, provided that these are derived from pre-harvest random mating among the adults. In the presence of substantial pollen flow from a genetically differentiated outside pollen source (*e.g.*, 10-20 percent pollen flow), the minimum amount of regeneration needed is reduced. This indicates that outside pollen can be more efficient, relative to pollen from within the stand, at replacing alleles lost from the adult population.

Genetic diversity of commercially important timber species is a critical resource that, in addition to providing a basis for tree improvement programs, is essential to tree species' future adaptive potential (Ledig 1988, 1992, Young and others 2000). However, in the Central Hardwood forest region, beyond the selection and breeding of elite trees for commercial purposes, it seems that scant attention has been paid to gene conservation in economically important forest tree species such as black walnut (*Juglans nigra* L.), northern red oak (*Quercus rubra* L.), or black cherry (*Prunus serotina* Ehrh.). Such tree species in the Central Hardwood region face a triple threat in regard to their genetic diversity. First, since European settlement, much of the formerly continuous forest in this region has been cleared for agriculture resulting in today's highly fragmented landscape (Hicks 1998). Second, in many of the remaining stands, the most valuable species have been subjected to repeated high grading, the targeted logging of phenotypically superior individuals (McGuire and others 1999). And third, shade intolerant species, such as those mentioned above, are exhibiting widespread recruitment failure, largely as a result of current management practices under which creation of large gaps is often avoided and fires are suppressed (Lorimer 1993). In the face of these mounting pressures on the genetic resources of Central Hardwood tree species, it would seem that the time is ripe for a coordinated and systematic approach to gene conservation in this region.

Gene conservation in trees is generally accomplished by a combination of *ex situ* and *in situ* approaches (Ledig 1988). *Ex situ* programs utilize seed stores, seed orchards, clone banks, and/or progeny or provenance tests (Lipow *et al.* 2002). *In situ* programs attempt to conserve genetic resources in their natural surroundings through management of native stands (Ledig 1988). *In situ* gene conservation is

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not necessarily incompatible with economic exploitation; it should be possible to adapt management practices to limit the potential deleterious impact of timber harvest upon local gene pools (Ledig 1988, Millar and Libby 1991). This could be achieved by ensuring that the regeneration of the tree species of interest is derived from local seed sources, preferably from pre-harvest matings (Glaubitz and others 2003b). Toward this end, guidelines for the amount of regeneration needed to minimize genetic diversity loss, depending on the intensity of harvest, are sorely needed (Jennings and others 2001). We are developing a computer simulation model to help provide such guidelines for Central Hardwood tree species, beginning with black walnut as a model. The key role that computer simulation models will play in forest tree conservation genetics has recently been emphasized by Boshier and Young (2000).

In this paper, we present results from our preliminary model, exploring the amount of regeneration needed in a forest stand to minimize the loss of genetic diversity caused by harvest, under the assumption that the regeneration was produced by pre-harvest random mating among the original adults. We also explore the modulating effect of pollen flow from a genetically differentiated outside pollen source, on either the allelic richness (the average number of alleles per locus; A_R) of the stand after harvest, or on the proportion of the alleles present in the original adult population that are retained (proportional allelic retention; PAR). We demonstrate that, in the presence of pollen flow, PAR is superior to A_R as a gauge of the immediate impact of harvest upon local genetic diversity.

Methods

General Scenario

We have modeled a genetically diverse forest tree population, consisting of 200 adult individuals of a single species, in which numerous replicate, partial harvests were performed. We will refer to this population as the *focus stand* or *focus population*. We have examined the potential genetic effects of harvesting on this focus stand, depending on the amount of surviving regeneration. In our current model, the regeneration population is produced prior to harvest via random mating among the adults, with varying amounts of pollen flow from an outside pollen source. In this scenario the regeneration could derive, in theory, from any or all of the following:

1. Advance regeneration that has not reached reproductive maturity,
2. A seed-bank of dispersed seed derived from pre-harvest matings, or
3. Seedlings grown from a representative seedlot collected from the focus stand prior to harvest.

Our simulation model treats the adult and regeneration populations in the focus stand as two discrete, non-overlapping generations.

Simulation Model Details

To explore the potential impacts of various harvest intensities on genetic diversity, we simulated genotypes at ten unlinked, Mendelian loci for two populations, the focus population and the outside pollen source. We wanted the focus population to have exactly 150 alleles in total, and the outside pollen source and focus population to have a Nei's genetic identity (Nei 1972) of 0.7 to 0.8. To obtain a pair of populations with these characteristics, we first generated a 'reference population' consisting of ten thousand individuals. This reference population started out completely monomorphic at all ten of the simulated loci, and then was allowed to evolve for one hundred thousand generations at constant size, under a 'k-alleles model' of mutation (KAM; Crow and Kimura 1970), with a mutation rate of 0.001 and a maximum number of alleles (k) of 60 at each locus. Each subsequent, non-overlapping generation was formed by random mating. From this large and highly genetically diverse reference population, a smaller population of 400 randomly-chosen individuals was founded and allowed to evolve via drift and mutation, again under random mating with KAM mutation, but with a tenfold lower mutation rate of 0.0001. This mutation rate was selected because it falls within the typical range observed at microsatellite loci (Whittaker and others 2003). After 45 generations, this population was split into two, namely, the focus population and the outside pollen source. These two populations were

each held at a constant size of 200 randomly mating individuals and allowed to drift independently (with mutation at the same rate of 0.0001) until the allelic richness in the focus population fell to exactly 15.0 (this required 14 generations of independent evolution).

In this manner, both the focus population and the outside pollen source had levels of genetic diversity and allele frequency distributions similar to what we have observed with microsatellite DNA markers¹ in a sample of 40 black walnut trees from a protected stand in the Hoosier National Forest, IN (data not shown). The degree of genetic similarity between the focus population and the outside pollen source was determined by the ratio of the number of generations that they co-evolved as one population versus the number of generations that they evolved independently: the numbers that we employed (45 and 14) resulted in a Nei's genetic identity (Nei 1972) between the two populations of 0.75.

With a focus population and outside pollen source in hand, the next step in our simulation was to perform numerous replicate harvests under a range of values of the following three parameters:

1. The proportion of the adults harvested from the focus stand,
2. The number of surviving regenerants, and
3. The proportional contribution of pollen from the outside pollen source.

The regeneration population was formed by random mating among the adult population in the focus stand prior to harvest, with pollen gametes contributed via Mendelian inheritance from randomly selected individuals from the outside pollen source according to the desired rate of gene flow via pollen. There was no gene flow from outside via seed. Also, there was no mutation during the formation of the regeneration.

For each replicate, the allelic richness (total number of alleles; A_R) relative to the original focus population of 200 adult trees was calculated for the combined population of regeneration and surviving adults. In addition to the relative allelic richness, the 'proportional allelic retention' (PAR) was also calculated, again relative to the original focus population of 200 adults. PAR measures the proportion of the alleles present in the original focus population of 200 adults that, after harvest, were still present in at least one copy in the combined population of regeneration and surviving adults. Note that in a closed system, with no gene flow via pollen or seed from outside the stand, relative A_R after harvest and PAR after harvest will be identical.

Our first objective was to show the response surface of relative A_R over the full range of harvest intensities (*i.e.*, from 0 to 100 percent harvest) with varying amounts of surviving regeneration (from 0 to 200 regenerants), in the absence of gene flow from outside the focus stand (fig. 1). Here, for each unique combination of the two parameters, harvest intensity and number of regenerants, we plotted the average relative A_R based on one thousand replicate harvests. We examined intervals of harvest intensity and number of regenerants of 1 percent and one regenerant, respectively (*i.e.*, $101 \times 201 \times 1,000 = 20,301,000$ simulated harvests in total for Figure 1).

Our next objective was to examine the effects of pollen flow on the minimum amount of surviving regeneration needed so that there would be no more than a 10 percent risk either of reducing allelic richness by more than 10 percent, or of retaining less than 90 percent of the alleles present in the original focus population. We examined levels of pollen flow from the outside pollen source of 0, 1, 2, 5, 10 or 20 percent. The minimum amount of regeneration needed was determined for harvest intensity levels varying from 0 to 100 percent, at 5 percent intervals. For this purpose, we developed a 'contour-finding' algorithm which we ran for each level of pollen flow, and for relative A_R and PAR

¹Microsatellite DNA markers are commonly used genetic markers consisting of tandem repeats of a simple sequence motif (*e.g.*, CACACACACA...). Such markers usually have numerous alleles differing in their repeat number (*e.g.*, CA₁₀ vs. CA₁₃).

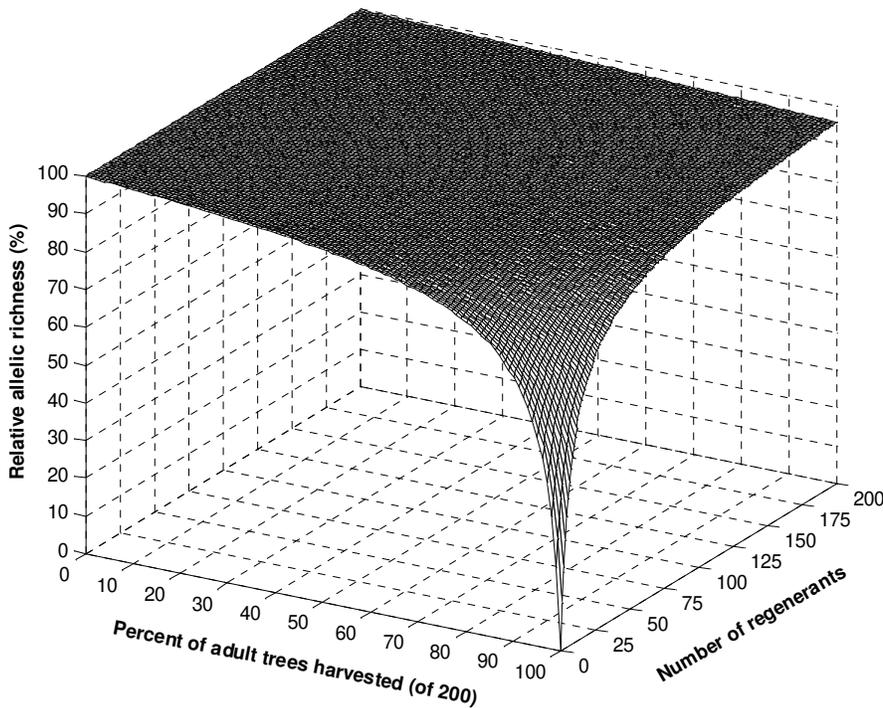


Figure 1.—Effects of harvest intensity and amount of regeneration on retained allelic diversity in a simulated, randomly mating forest tree population, in the absence of pollen flow from outside. Mean allelic richness (relative to the original adult population) is plotted based on 1000 replicate harvests.

separately. Ten thousand replicate harvests were performed for each unique combination of harvest intensity and amount of regeneration examined by our algorithm. Depending upon our purpose, either relative A_R or the PAR in the combined regeneration and surviving adult population was calculated for each replicate, and the resulting distribution of outcomes was then sorted from lowest to highest. The tenth percentile (*i.e.*, the one thousandth observation) was taken to represent the level of relative A_R (or PAR) corresponding to 10 percent risk; in other words, there was no greater than 10 percent risk that A_R (or PAR) would fall below that level, for that particular combination of harvest intensity and amount of regeneration.

Our ‘contour-finding’ algorithm began at 0 percent harvest and zero progeny (relative A_R or PAR of 100 percent, by definition), and then successively increased the harvest intensity by 5 percent, until the relative A_R (or the PAR) at the 10 percent risk level fell below 90 percent. At this point, the number of regenerants was successively increased by one until the relative A_R (or the PAR) at the 10 percent risk level either equaled or surpassed 90 percent. Once this occurred, the harvest intensity was again increased by 5 percent, and the number of regenerants (starting from the last value taken at the previous harvest intensity) was again successively increased by one until the relative A_R (or the PAR) at the 10 percent risk level again either equaled or surpassed 90 percent. At each level of harvest intensity, the minimum number of regenerants needed for no greater than a 10 percent risk of reducing A_R (or the PAR) by more than 10 percent was recorded. This process was repeated until, finally, the minimum number of regenerants was determined for a 100 percent harvest of all 200 adult trees in the focus stand.

Our simulation was written in the programming language C++, and carried out on PC computer running Microsoft Windows 2000.

Results

The response surface of the average (or ‘expected’) value of relative A_R , over the full range of harvest intensities, with varying amounts of surviving regeneration, and in the absence of gene flow from outside the focus stand, is shown in Figure 1. The 100 percent reference point for relative A_R lies at 0 percent harvest and no regeneration, and is equivalent to the A_R in the original focus stand of 200 trees

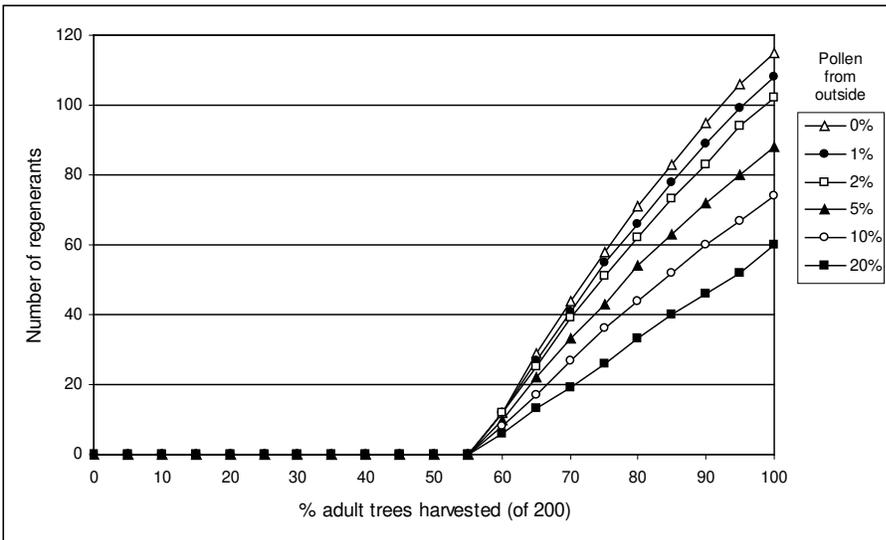


Figure 2.—The minimum number of regenerants required for no more than a 10 percent risk of reducing allelic richness by more than 10 percent. The effects of various levels of pollen flow from a genetically differentiated outside pollen source are shown.

prior to regeneration or harvest. When no adults are harvested, relative A_R does not rise with increasing numbers of regenerants, but, since there is no pollen flow from outside in this scenario, instead stays constant at 100 percent (note that, if there was pollen flow from a genetically differentiated outside pollen source, relative A_R would rise gradually with increasing numbers of regenerants). With increasing harvest intensity, A_R in the remaining trees is reduced, but this reduction does not become 'drastic' until high harvest intensities (*i.e.*, greater than 70 percent) in combination with low numbers of regenerants (*i.e.*, less than 75). Beyond this, A_R drops off dramatically. Obviously, at 100 percent harvest with no regeneration, relative A_R is zero. With 200 regenerants and 100 percent harvest, there is a slight reduction in expected A_R , in accordance with population genetic theory regarding genetic drift in finite populations (Hartl and Clark 1997). Overall, the results shown in Figure 1 suggest that there is a broad range of harvest intensities and amounts of regeneration over which there will only be a relatively mild impact upon the allelic diversity in a randomly mating forest tree population.

Each point in Figure 1 represents an *average* A_R based upon one thousand replicate harvests. However, it may be more pertinent for forest managers to consider gene conservation in a risk management context (Namkoong 1999). A forest manager or policy maker may wish to decide upon a target level of genetic diversity to conserve, and then set a maximum level of risk of falling short of this target. For the purposes of this study, we have chosen 90 percent of the allelic diversity in the original adult stand as the target level to conserve, and 10 percent as the acceptable level of risk of falling below this target.

To this effect, Figure 2 plots the minimum number of regenerants required to maintain a 10 percent or lower risk of reducing A_R by more than 10 percent, over the full range of harvest intensities (zero to 100 percent harvest). This relationship is shown for various levels of pollen flow from the genetically differentiated outside pollen source. In general, the risk level decreases as you move towards the upper left hand corner of Figure 2 (less harvest and more regeneration), and increases as you move towards the lower right (more harvest and less regeneration). The figure shows that, in the absence of regeneration, up to 55 percent of the adults can be harvested without exceeding the 10 percent risk level. Note that up to this point (55 percent harvest) the level of pollen flow from the outside stand is irrelevant.

Relative to the curve for 0 percent pollen flow in Figure 2, pollen from the genetically differentiated outside pollen source clearly has a marked effect, with as little as 1 percent pollen flow causing a discernable reduction in the amount of regeneration needed. When outside pollen flow is 20 percent,

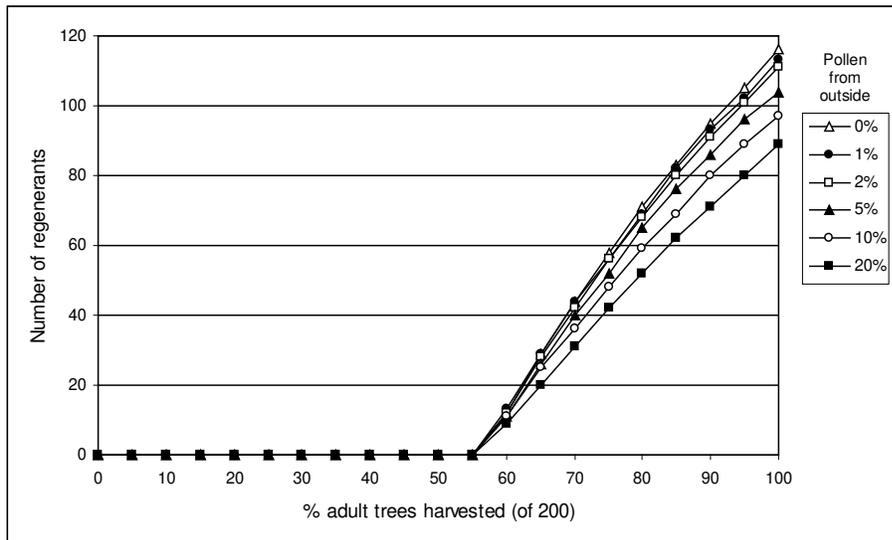


Figure 3.—The minimum number of regenerants required for no more than a 10 percent risk of retaining less than 90 percent of the alleles present in the original adult population. The effects of various levels of pollen flow from a genetically differentiated outside pollen source are shown.

the amount of regeneration needed to maintain relative A_R at 90 percent after all of the adults are harvested is nearly halved. This effect is primarily due to the novel alleles that are introduced to the focus population via pollen flow. This illustrates the limitation of using A_R as a sole measure by which to gauge the genetic effects of forest management. Allelic richness a non-discriminating measure, since it merely takes into account the number of alleles, but not their identity (Glaubitz 2003a, 2003b). If the outside pollen source was highly divergent (e.g., an ‘exotic’ plantation) it is conceivable that, under high levels of pollen flow, relative A_R could be maintained or even substantially increased, concomitant with loss of a large proportion of the alleles present in the original adult stand.

For this reason, we also plotted the minimum number of regenerants required to maintain a 10 percent or lower risk of reducing the *proportional allelic retention* (PAR; defined above) by more than 10 percent, for various levels of pollen flow from the outside pollen source (fig. 3). With PAR, in contrast to A_R , ‘novel’ or ‘foreign’ alleles introduced via pollen flow from outside do not contribute. Since in a closed system (without outside pollen flow), A_R and PAR are identical, the lines for 0 percent outside pollen in Figures 2 and 3 are equivalent. However, when PAR is used (fig. 3), pollen flow from outside has a much weaker reducing effect on the amount of regeneration needed, as compared to using A_R (fig. 2). In Figure 3, a low level of pollen flow (i.e., 1 or 2 percent) has an inconsequential effect. However, the reducing effect of 20 percent outside pollen on the amount of regeneration needed is still substantial. For example, when all of the adult trees are harvested, only 89 regenerants are needed with 20 percent pollen flow versus 116 with 0 percent pollen flow. This reduction indicates that pollen from the outside source is more efficient, relative to that from inside the stand, at replacing alleles that have been removed from the adult portion of the focus stand by harvesting.

Discussion

One purpose of this paper was to demonstrate the superiority of proportional allelic retention (PAR) over allelic richness (A_R) as a measure of the amount of allelic diversity conserved in a forest tree population after harvest. The weakness of A_R is that it does not take into consideration the identity of alleles; with this measure, alleles lost due to harvest of the adults can be compensated for by the gain of ‘foreign’ or ‘exotic’ alleles via gene flow from outside. This is not the case with PAR; here outside pollen flow can only contribute to the retention of allelic diversity via replacement of the actual alleles lost from the adults via harvest. The contrast in the two measures can be seen by comparing Figures 2 and 3.

Through the use of PAR, we have demonstrated that pollen from a genetically differentiated outside pollen source can be more efficient than pollen from within the stand at replacing alleles removed from the adult population by harvest, even when the regeneration has been derived from pre-harvest matings. In order to explain this seemingly paradoxical effect, we must first note that it is the low frequency alleles that are most likely to be lost due to partial harvest (Buchert and others 1997, Glaubitz 2003a, Glaubitz 2003b). Furthermore, if the outside pollen source is somewhat genetically differentiated from the focus population, then, among the set of alleles that are *rare in the focus population*, there will likely be a subset that are *more common in the outside pollen source*. It is these alleles that will be more efficiently replaced by pollen from outside relative to pollen from within the stand. If this is the case, then we would expect the level of outside pollen to have a 'diminishing returns' effect: once all the alleles in this class have been replaced, then additional pollen from the outside source will have no further reducing effect on the amount of regeneration required to obtain a PAR of 90 percent or more. If we allow even higher levels of pollen flow from outside (up to 100 percent) we find that this is indeed the case, with the maximum reducing effect nearly obtained at 40 percent pollen flow (results not shown).

With high levels of pollen flow from outside, a potential deleterious side effect accompanying more efficient allelic replacement might be the influx of a large number of foreign alleles into the stand. The magnitude of this effect will be reflected in the difference between relative A_R and PAR, with a large difference indicating influx of a large number of foreign alleles. It is difficult to say whether this would be detrimental or not from a conservation genetic standpoint, since preserving 90 percent of the original alleles found in the adult stand while adding additional alleles via pollen flow may actually enhance the adaptive potential of the focus population. If our goal is to preserve natural processes, then it would depend on the level of gene flow that would naturally occur in the absence of human disturbance or management. More empirical data is needed on this, particularly in the Central Hardwoods; we are currently gathering such data in our empirical studies of black walnut. Clearly, a large influx of foreign alleles from an *exotic* pollen source, such as a plantation derived from non-local germplasm, would be detrimental to gene conservation.

We will use the empirical data that we are gathering from black walnut using microsatellite DNA markers to help refine the basic computer simulation model that we have presented here, in order to make it more realistic. Refinements that we plan to incorporate include the following:

1. The influence of spatial position, wind direction, crown size and reproductive phenology on pollination success.
2. The influence of spatial proximity to gaps and gap size on female reproductive success.
3. The genetic distance to likely outside pollen sources (both 'natural' and plantation-based), and
4. The effect of alternative harvest practices and/or seed collection strategies, including consideration of the timing of harvest relative to reproduction.

It should be noted, however, that recent empirical data that we have collected from black walnut indicate that our basic assumption of random mating represents a reasonable first approximation. Based upon a sample of 89 black walnuts from two adjacent fragments in Carroll County, IN, we obtained an estimate of the panmictic index (F_{IS}) of -0.0042 across 15 microsatellite loci. This is very close to an F_{IS} value of zero, indicating random-mating genotypic proportions.

We eventually hope to use our model to project further into the future, accounting for overlapping generations, and larger spatial scales. Projection into the future may best be achieved by coupling our genetic model with succession-based models predicting future stand demography in relation to varying management practices (Porte and Bartelink 2002).

One potential application of the results of this study (and future extensions thereof) would be for the effective management of Genetic Resource Management Units (GRMU's; Ledig 1988, Millar and Libby 1991). The concept of GMRU's is to allow some level of timber extraction via partial logging, provided that this does not interfere with the overarching goal of conserving the local genetic resources

in situ. Future refinements of the basic model that we have presented here could lead to guidelines on what level of harvest would be appropriate in a GRMU, in relation to the target amount of allelic diversity to conserve and the acceptable level of risk, given the amount of regeneration expected to survive to maturity. Although GRMU's have not been adopted in the Central Hardwoods, considering the commercial value of many of our fine hardwood species, and the high levels of fragmentation, human disturbance and high-grading that have occurred in large portions of this region, the establishment of GRMU's for important timber species in the Central Hardwoods seems long overdue.

Acknowledgments

This research was funded by the Fred M. van Eck Forest Foundation, the USDA Forest Service North Central Research Station, the Institute For Sustainable Agricultural Systems, and Purdue University.

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CONSUMER PERCEPTIONS AND KNOWLEDGE OF COMMON FURNITURE WOODS

Scott A. Bowe and Matthew S. Bumgardner[†]

ABSTRACT.—We report findings from two studies that describe the perceptions consumers have of several common furniture wood species. A comparison is made between a 2001 survey of undergraduate students at a large midwestern university, and adult consumers intercepted at furniture stores and trade shows in the same and a nearby city in 2003. In both samples, participants were split into two groups and asked to rate six species on several semantic differential items, based either on word association or the appearance of wood specimens. Use in bedroom furniture was given as the frame of reference. The two methods of evaluation often produced different results, suggesting that the reputation a species has is not always based on its physical appearance. A high correspondence was found between the two groups concerning the species ratings. However, the adult consumers were better at identifying the specimens than were the college students. The results obtained from both studies should alert secondary wood manufacturers to the need for better understanding of the role perceptions play in consumer choice. This understanding can be used to enhance product design and communications decisions.

It has been demonstrated that people have differing perceptions of different wood species (Blomgren 1965, Swearingen et al. 1998, Bumgardner & Bowe 2002). If recognized and understood, these perceptions can be leveraged for marketing and product development advantages. In the furniture product development process, many decisions are made as to what new products will look like. One of the more important decisions involves what species to use (Bumgardner et al. 2001). Manufacturers have many species choices, and in many instances, multiple species will meet appearance, cost, and processing criteria for a given design. It is therefore important for furniture manufacturers to understand what their species selection decisions contribute to their products. The total product concept suggests that all attributes, both tangible and intangible, are part of what a consumer ultimately purchases when choosing a product (Levitt 1986).

There are indications that such decisions might play a role in determining the competitiveness of U.S. manufacturers as well. The furniture industry in the United States is undergoing substantial structural changes. Many domestic furniture companies have made the decision to move some or all of their manufacturing base offshore, either through outsourcing or direct investment in production facilities. Offshore manufacturers, particularly in China and Southeast Asia, enjoy considerable cost savings over their U.S. counterparts (Schuler and Buehlmann 2003). While the extent of loss of the domestic furniture manufacturing remains unclear, one thing seems certain: the industry of tomorrow will look quite different from the one of today.

Familiarity with U.S. species has been discussed as a possible source of competitive advantage for domestic manufacturers of secondary wood products (Lawser 2002), and research has shown that industry practitioners generally concur with this belief (Buehlmann et al. 2003). However, we found that most college students could not identify common hardwood species, although they maintained definite perceptual images of those same species (Bumgardner and Bowe 2002). The Appalachian

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Hardwood Manufacturers, Inc. (2002) report on a recent High Point Furniture Market noted that rubberwood (*Hevea brasiliensis* Muell. Arg) and other foreign species were often stained to look like cherry or referred to as “Asian oak” or “Asian cherry.” Combined with consumers’ perceptions but limited knowledge of wood species, these factors might contribute to confusion on the part of consumers and a missed opportunity for domestic manufacturers.

The objectives of this paper are to compare college students’ perceptions and knowledge of wood species as reported in Bumgardner and Bowe (2002) with a sample of older consumers surveyed at furniture stores and trade shows in the same general geographic area. Questions this study will help answer include: Is there a general perception and level of knowledge of different wood species among consumers, or does this change with experience? If there is a change, how much is “learned” through experience? Answers to these questions have implications for the design and promotion of wood products, the design of future studies investigating such topics, and the education of consumers.

Methods

Data Collection

As a comparison to the college student data from the 2001 study, an adult consumer sample was targeted for a follow-up study that took place from August 2002 through May 2003. Only individuals 25 years or older were included in the follow-up study. The University of Wisconsin Survey Center (UWSC) was contracted to collect the data. The survey instrument that was developed, pre-tested, and employed in the 2001 college student study was also used in the adult consumer study. The title *University of Wisconsin Survey Center: Wood Study* and center logo was added to the questionnaire booklet to identify the survey organization.

The data collection procedure utilized a modified mall intercept method in two midwestern cities, Madison and Milwaukee, WI. Initially, furniture stores were identified as the survey locations. This was to help insure that respondents had more experience with furniture purchases than did the college students previously surveyed. Two furniture retailers in Madison and one in Milwaukee offered space to conduct the survey. After several trials, it was determined that data collection volume was too low to achieve the data collection goals in a timely manner. Several relevant trade shows at local convention centers were then identified as alternate survey locations, and booth space was provided for the UWSC personnel. A simple display of forestry and forest products pictures was arranged in the booth space to pique interest in the study, and a UWSC banner was displayed to identify the organization. Incentives of candy bars and soda were used to encourage participation in the study.

Like in the college student study, the respondents were randomly split into two groups with approximately one-half completing a word-based perception questionnaire and one-half completing an appearance-based perception questionnaire. Respondents completing the word-based questionnaire were asked to evaluate six commonly used wood species based on the name of the species only. No visual cues were given. Respondents completing the appearance-based questionnaire were asked to evaluate six sample boards, which were identified only by question number. Two sample board sets were constructed for the 2001 study and reused for the current study. The sample boards consisted of six species samples measuring 0.5 inch by 4.0 inches by 6.0 inches mounted on plywood backing. The species evaluated included northern red oak (*Quercus rubra* L.), American mahogany (*Swietenia* sp.), black cherry heartwood (*Prunus serotina* Ehrh.), black walnut heartwood (*Juglans nigra* L.), sugar maple (*Acer saccharum* Marsh.), and eastern white pine (*Pinus strobus* L.). With the word-based perception questionnaire, the respondents were asked to evaluate each species based on the common names of *oak*, *mahogany*, *cherry*, *walnut*, *maple*, and *pine*. On the word-based perception questionnaire, respondents were asked to evaluate the species under the scenario that they had just seen a magazine advertisement for bedroom furniture made from the species in question. On the appearance-based perception questionnaire, the scenario for evaluation was being in a furniture store showroom and seeing bedroom furniture made from the wood specimen in question.

Table 1.—Theoretical factors describing household furniture and the semantic differential items selected to represent each factor.

Factor	Item
Quality	Fragile vs. Durable
Price	Expensive vs. Inexpensive
Visual Elements	Cold vs. Warm
Environmental Considerations	Sustainable vs. Depleting
Style	Casual vs. Formal
	Old-Fashioned vs. Modern
	Stately vs. Modest

Common components of both the word-based and appearance-based perception questionnaires were the semantic-differential scales employed and the theoretical factors they represented. Five theoretical factors used to describe wood household furniture were identified for the 2001 study (Ozanne and Smith 1996). One or more scale items were developed by the authors to describe each theoretical factor. The theoretical factors and corresponding items are shown in Table 1. Specific details on the factor/item development and instrument pre-testing can be found in Bumgardner and Bowe (2002).

To illustrate the word-based perception questionnaire, the respondent would consider the species *oak* and rate whether they thought it was *Fragile* or *Durable* on a seven-point scale (Fragile = 1 and Durable = 7). Likewise, on the appearance-based perception questionnaire, the respondent would examine the unlabeled oak sample block and rate whether they thought it appeared *Fragile* or *Durable* on a seven-point scale. The scales were treated as interval in nature (Coombs et al. 1970, Aaker et al. 1998), allowing for mean-based statistical analysis.

Sample Description

The number of usable questionnaires obtained was 871, which included 466 and 405 word-based and appearance-based questionnaires, respectively. Twenty-one percent of the responses were collected through furniture store interviews, and the remaining 79 percent were collected through trade show interviews. A large majority of the trade shows had a home and garden theme, the exception being one Corvette show. Fifty-three percent of the questionnaires were completed in Madison and the remainder were completed in Milwaukee.¹

Demographic data were collected, including information on age, gender, household income, home ownership status, and furniture purchases. The median age of respondents was 49. The sample was nearly equally split by gender with 51 percent female. Approximately 71 percent of respondents had a household income of more than \$50,000 per year. In addition, more than 87 percent of respondents owned their own house or townhouse. Thirty percent of respondents had been personally involved in a major furniture purchase in the past six months.

Results

Claimed vs. Ability to Identify Species

On the word-based perception questionnaire, respondents were asked if they *thought* they could correctly identify each of the six species if given the opportunity. On average across all species, 69 percent of adult consumers claimed they could identify each species, compared to 50 percent of college students from the previous study that claimed such ability. On the appearance-based perception questionnaire, the respondents were asked to identify each sample block in question. On average across all species samples, 36 percent of adult consumers correctly identified each sample, compared to 18 percent of college students that correctly identified each sample. As shown in Table 2, the adult

¹By comparison, the college student sample consisted of 146 word-based respondents and 107 appearance-based respondents. Ninety-one percent were 25 years of age or younger.

Table 2.—Claimed and actual ability to identify wood species, adult consumers and college students.

Species	Adult consumer:	College student:	Adult consumer:	College student:
	Claimed ability	Claimed ability	Actual ability	Actual ability
	%		%	
Oak	92	75	49	26
Mahogany	54	56 ¹	22	3
Cherry	70	52	20	11
Walnut	51	24	45	18
Maple	60	32	14	9
Pine	89	61	64	42

¹Numbers in bold italics denote proportions that are not significantly different between adult consumers and college students based on two-tailed z-tests, alpha=.05.

Table 3.—Summary of the adult consumer and college student word-based perception results. Adult consumer response is listed first, followed by the college student response. An asterisk denotes a mean not statistically different from the scale midpoint. A single entry indicates same response by both groups.

Species	Casual	Cold	Expensive	Fragile	Old-fashion.	Sustain.	Stately
	vs. Formal	vs. Warm	vs. Inexpensive	vs. Durable	vs. Modern	vs. Depleting	vs. Modest
Oak	*/formal	warm	expensive	durable	*/old-fash.	sustainable	stately
Mahogany	formal	warm	expensive	durable	old-fash.	depleting/*	stately
Cherry	formal	warm	expensive	durable/*	old-fash.	*/sustainable	stately
Walnut	formal/*	warm/*	expensive	durable	old-fash.	sustainable	stately
Maple	casual	warm	*/inexp.	durable	old-fash.	sustainable	modest/*
Pine	casual	*	inexp.	fragile	old-fash./*	sustainable	modest

Table 4.—Summary of the adult consumer and college student appearance-based perception results. Adult consumer response is listed first, followed by the college student response. An asterisk denotes a mean not statistically different from the scale midpoint. A single entry indicates same response by both groups.

Species	Casual	Cold	Expensive	Fragile	Old-fashion.	Sustain.	Stately
	vs. Formal	vs. Warm	vs. Inexpensive	vs. Durable	vs. Modern	vs. Depleting	vs. Modest
Oak	casual	cold	*/inexp.	durable	modern	sustainable	*/modest
Mahogany	formal	warm	*/expensive	durable	old-fash./*	sustainable	stately
Cherry	casual/*	warm/*	*	durable	*/old-fash.	sustainable	*/stately
Walnut	formal	warm/cold	expensive	durable	old-fash.	sustainable	stately
Maple	casual	cold	inexp.	*	modern	sustainable	modest
Pine	casual	cold/*	inexp.	fragile	old-fash./mod.	sustainable	modest

consumers were generally more successful in identifying the sample blocks than were the college students, the lone exception being maple. The average difference between *ability* and *claimed ability* was 34 percentage points for the adults and 32 percentage points for the students across all species. Additionally, the pattern of claimed vs. ability was quite similar between the adult consumers and college students. For example, with both groups, oak was the species generating the highest claimed ability, but pine was the species most correctly identified. The student group had a particularly difficult time identifying mahogany (3% could identify), while maple provided the greatest challenge for the adult group (14% could identify).

Word-Based and Appearance-Based Perceptions

The results for the word-based and appearance-based evaluations are shown in Table 3 and Table 4 and summarized below. A two-tailed *t* test (alpha = 0.05) was used to determine if the means were significantly different than the midpoint (4.0) for each semantic differential scale. Overall, few

Table 5.—Means from the word-based perception evaluations for adult consumers¹.

Species	Casual vs. Formal	Cold vs. Warm	Expensive vs. Inexpensive	Fragile vs. Durable	Old-fash. vs. Modern	Sustainable vs. Depleting	Stately vs. Modest
Oak	3.9	4.7	3.7	5.9	3.9	3.7	3.8
Mahogany	5.8	4.6	3.0	4.6	3.3	4.2	2.7
Cherry	5.7	5.3	2.9	4.5	3.7	3.9	3.0
Walnut	4.8	4.7	3.2	5.0	3.5	3.8	3.3
Maple	3.4	4.5	4.1	4.7	3.8	3.3	4.3
Pine	1.6	4.1	5.7	2.9	3.3	3.0	5.4

¹Means based on rating scales ranging from 1 to 7, anchored by the column headings. A lower number corresponds to the first word in the heading.

Table 6.—Means from the appearance-based perception evaluations for adult consumers¹.

Species	Casual vs. Formal	Cold vs. Warm	Expensive vs. Inexpensive	Fragile vs. Durable	Old-fash. vs. Modern	Sustainable vs. Depleting	Stately vs. Modest
Oak	2.8	3.2	4.0	4.5	4.2	3.6	4.1
Mahogany	4.3	4.8	3.9	4.5	3.7	3.6	3.7
Cherry	3.7	4.3	3.9	4.4	3.9	3.8	3.9
Walnut	4.7	4.5	3.6	4.9	3.5	3.8	3.5
Maple	2.6	3.0	4.3	4.0	4.5	3.7	4.5
Pine	2.2	3.7	5.0	3.5	3.7	3.4	5.0

¹Means based on rating scales ranging from 1 to 7, anchored by the column headings. A lower number corresponds to the first word in the heading.

differences were found between the adult consumer and college student groups. The two groups were in agreement about 75 percent of the time with both the word-based and appearance-based evaluation (76.2 and 73.8 percent, respectively). There were only two instances where the adult consumers and college students were on opposite sides of a scale, and both occurred with the appearance-based evaluations. One case involved walnut, with adult consumers rating this species as warm and college students rating it as cold. The other case involved pine, with adult consumers rating this species as old-fashioned and college students rating it as modern.

Results important for understanding how different species might impact product design and communication decisions are shown in Table 5 and Table 6. As shown in Table 5, based on the word-based perception evaluations, adult consumers² rated mahogany and cherry as the most formal species, and pine was rated as the most casual. Cherry was rated as the warmest species, but all species were rated as warm with the exception of pine. Cherry, mahogany, and walnut all rated highly as expensive species. Oak was rated as the most durable. Mahogany and pine were rated as the most old-fashioned. Pine was rated as the most sustainable. Mahogany was rated as the stateliest, followed closely by cherry. Pine was rated as the most modest.

The appearance-based perception evaluations for the adult consumers are shown in Table 6. Walnut was rated as the most formal, and pine was rated as the most casual. Mahogany was rated as the warmest, while maple and oak were rated as the coldest. Interestingly, walnut was the only species rated as expensive, and pine was rated as the most inexpensive. Walnut was also rated as the most durable. Maple was rated as the most modern, and walnut as the most old-fashioned. All species were rated as sustainable. Lastly, walnut and mahogany were rated as the stateliest, and pine the most modest.

²See Bumgardner and Bowe (2002) for similar results from the college student study.

A final consideration is the extent to which the word-based evaluations differ from the appearance-based evaluations for the adult consumer sample. Several such differences were found (a similar trend was noted among college students in the previous study). While several of these differences involved neutral ratings for either the word-based or appearance-based evaluations, those involving opposite ratings are noted here. Cherry was rated as formal on the word-based evaluations and as casual on the appearance-based evaluations, suggesting that cherry's formal reputation surpasses its appearance. Oak was rated as warm on the word-based evaluation and as cold on the appearance-based evaluation. Maple followed this same pattern. It seems wood in general, across species, is perceived as warm in name but might be perceived as cold in appearance, particularly lighter-colored species. Maple was rated as old-fashioned on the word-based evaluation and modern on the appearance-based evaluation. Interestingly, mahogany was rated as depleting on the word-based evaluation and as sustainable on the appearance-based evaluation; these findings perhaps suggest the difficulty of rating this attribute on appearance and a general perception that tropical woods are not being utilized in a sustainable fashion.

Discussion

This paper sought to determine if age and experience with household furniture affects perceptions and knowledge of common wood species. The answer seems to be no and yes. The word-based and appearance-based perceptions associated with the species investigated were quite similar for college students and adult consumers. By investigating college students, it seems possible to get a reasonable idea of adult consumers' perceptions of wood species. So when it comes to perceptions of wood species, age and experience seem to have little effect. Stated another way, by the time people arrive at college, they have already formed their species perceptions.

However, the adult consumers were better at species identification than were the college students. This suggests a greater level of knowledge on the part of the older and more experienced consumers. Still, pine was the only species to have at least 50 percent correct identification. Overall, there seems to be a lack of wood species knowledge even among more experienced consumers, which might be a troubling finding for some domestic manufacturers. The pattern of correct identification was similar between both groups, suggesting that both groups struggle with the same species, particularly maple and to a lesser extent mahogany and cherry. The gap between claimed and actual ability was very similar for both the adult and student consumers. While both groups substantially overestimated their abilities, this corresponded to their respective abilities. Perhaps respondents intentionally claimed an ability they did not possess; an alternative explanation is that a general familiarity with common wood species, if only in name, leads people to think they will know it when they see it.

One interesting finding regarding oak noted in the previous study of college students was that word-based evaluations of oak were often opposite to appearance-based evaluations, and that the word-based evaluations were more positive. This trend was not as evident with adult consumers, though the word-based perception was *warm* while the appearance-based perception was *cold*. This suggests that adult consumers are not as enamored with the reputation of oak as are less experienced students.

Acknowledgments

The authors wish to thank the University of Wisconsin Survey Center for their data collection efforts, including the following individuals: John Stevenson, Renee Roerdink, Amanda Pate, Renae Brincks, Dana Kiblawi, Mike Mankowski, and Tiffany Harper.

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DEVELOPMENT OF A SPATIALLY EXPLICIT LAND USE/ECONOMIC IMPACT ASSESSMENT MODEL

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ABSTRACT.—Land use decisions impact both the environment and economy of a region. Objectives of this study are: 1) to create a tool, integrating GIS and IMPLAN that will aid the prediction of land use change and quantification of the spatially-dependent economic impacts that result, and 2) to apply the new tool to compare the current allocation of land to corn, soybeans, hay, and forest in St. Clair County, Illinois with an income-maximizing land allocation scenario; and 3) examine the regional economic impacts of shifting to the income-maximizing scenario. The four land uses considered in this study are corn, soybeans, hay, and forest. A land use modeling tool is utilized to determine the income maximizing land use production capabilities for each pixel representing the research areas in St. Clair County. Impact Analysis for Planning (IMPLAN) is used to conduct an economic analysis and comparison of St. Clair County under the current and income-maximizing land use allocation. Results indicate that much of the land use within the county is utilized below its maximum potential. IMPLAN output reveals that St. Clair County is better suited for and would benefit from an expansion in production of hay and forest. Model results suggest that land use decisions in the study area are based on factors additional to economic returns.

Land use decisions impact both the environment and economy of a region. Computer tools developed through research in the field of economics allow the measurement of impacts that occur as a result of changes in an economy. Also, natural resource managers have enlisted prediction-based computer models of landscape change and environmental impact. While several researchers have sought to combine economic impact and land use change models (e.g. Irwin and Geoghegan 2001; Chomitz and Gray 1996; Baker 1989), these efforts have fallen short in their ability to simultaneously predict spatially explicit economic and environmental impacts. The purpose of this research is to develop a tool that will integrate the economic utility of input-output (I-O) analysis and the geo-spatial power of GIS to create a spatially explicit land use/economic impact model.

IMPLAN and Input-Output Analysis

While I-O analysis and economic land use change models have provided answers to key development questions, several deficiencies exist in each tool. Through I-O analysis, a framework can be developed to collect, categorize, and analyze information on interindustry and interdependencies throughout an economy (Shaffer 1989). This powerful tool allows the user to measure the industry-specific impacts resulting from a single transaction (Miernyk 1965). IMPLAN is an I-O analysis computer model developed by the United States Forest Service in 1982 (Siverts 1985). The major shortfall of I-O analysis is in the aspatial nature of the model's geographic range. IMPLAN I-O tables are first established at the national level and then divided by region, county, or zip code (a recent innovation) (Holland et al. 1997).

GIS and Land Use Models

Land use models created with the aid of GIS, are designed to predict the rate and direction of land use change resulting from economic aspects of human behavior (see Irwin and Geoghegan 2001; Baker

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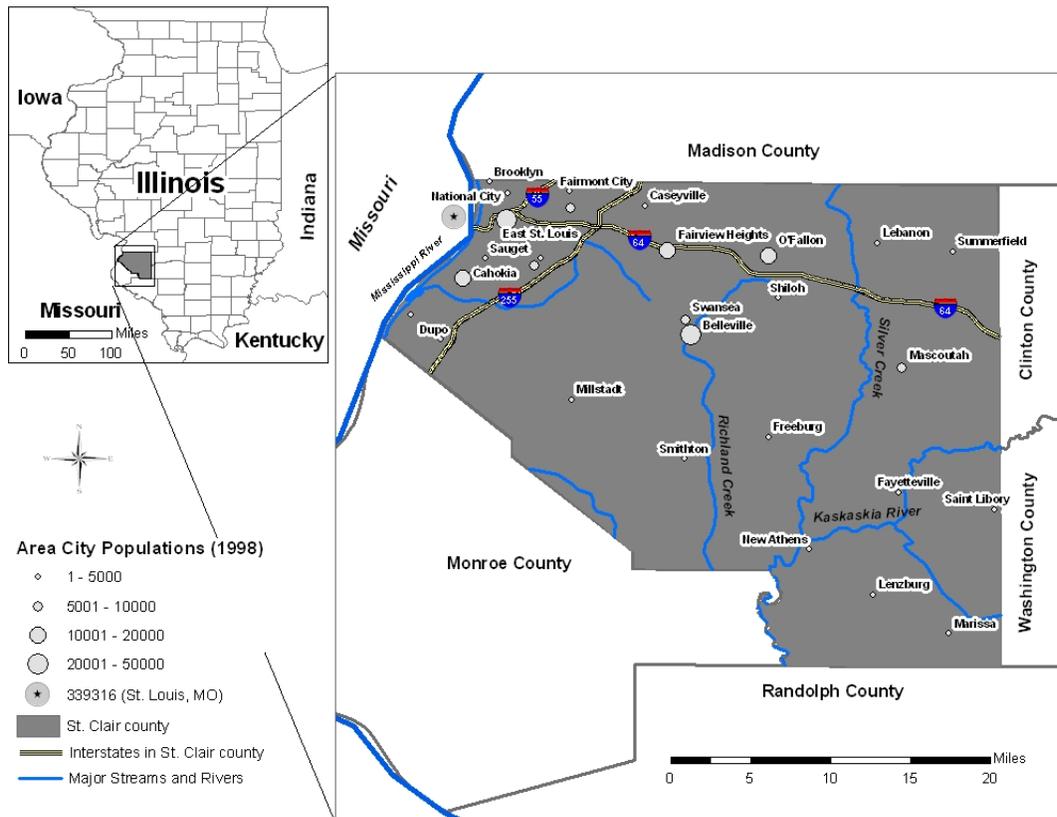


Figure 1.—Location of St. Clair County, Illinois.

1989). Irwin and Geoghegan (2001) categorize models of land use change as being either non-spatially explicit or spatially explicit. Non-spatially explicit economic models of land use change attempt to estimate where land use change is likely to occur based on the assumption of a homogeneous landscape (von Thünnen 1966). The available “resolution” of these land use change models is considered ineffective for identifying variation at the field or pixel level (Irwin and Geoghegan 2001). Spatially explicit economic models of land use change are characterized by the application of economic value as determined by a number of attributes that are unique to each pixel in the study area (Irwin and Geoghegan 2001).

Study Objectives

The purpose of this study is to create a tool, integrating GIS and IMPLAN that will aid the quantification of spatially-dependent economic and environmental impacts. Specific objectives of this study are; 1) to apply the new tool to compare the current allocation of land to corn, soybeans, hay, and forest in St. Clair County, Illinois with an income-maximizing land allocation scenario; and 2) examine the regional economic impacts of shifting to the income-maximizing scenario.

Study Area

St. Clair County is located in southwestern Illinois along the Mississippi River (fig. 1). The U.S. Census Bureau (2003) estimates St. Clair County’s 2002 population at 2.58 hundred thousand with its largest population centers in East St. Louis and the county seat, Belleville.

St. Clair County possesses a variety of land uses from urban to wetland. Productive soils and close proximity to major shipping ports in St. Louis, MO and the Mississippi River have also allowed St. Clair County to become a chief contributor to the Illinois agricultural sector. St. Clair County contains

over 245 thousand acres (57 percent of the county) allocated to the production of various agricultural goods, as well as over 56 thousand acres (13 percent) of forested and forested wetlands (USDA 2002). Soybeans and corn have been the main agricultural products with 92 thousand acres of soybeans and 72 thousand acres of corn being produced in 2001 (USDA 2002). The county is also home to Illinois' largest contiguous tract of forest maintained along the Kaskaskia River.

Methodology

Spatial Database Development

Spatial data were collected in this study to examine the current land use in St. Clair County and to evaluate the economic returns that would be expected from the current land uses (i.e., corn, soybeans, hay, and forest production). Geospatial analysis was conducted primarily in raster format utilizing ArcGIS 8.x (ESRI 2002).

The current allocation for each land use is derived from the USDA's cropland data layer (USDA 2002). From the cropland data layer, all areas with the value of 1, 3, 5, and 64 were identified as corn, hay, soybeans, and forest production respectively. These areas were individually classified and identified as the current land use allocation for this study. Determining the income maximizing land use allocation required the creation of production, gross income, and net income layers for each land use. The creation of each of these layers is described below.

Production levels for the three traditional agricultural crops (corn, soybeans, and hay) were taken directly from the soil survey geographic database (SSURGO) produced by the USDA (1999). This digital soil database identifies production levels for many agricultural crops at the pixel level for all of St. Clair County. According to the SSURGO data, the range of corn production in St. Clair County is between 57 and 163 bushels per acre (bpa) with the mean production at 102.9 bpa. Soybean production ranges between 20 and 54 bpa with a mean of 34.8 bpa. Hay production ranges between two and six tons per acre with a mean of 3.3 tons per acre. Forest production ranges between 2.9 thousand and 9.8 thousand with a mean of 3.8 thousand board feet fiber per acre per year.

Pixel level forest production levels were determined by the following procedure. Forest productivity in St. Clair County was determined by soil class using soil surveys provided by the USDA (1978) Natural Resource Conservation Service for St. Clair and neighboring Illinois counties (Washington, Clinton, Randolph, Monroe and Madison). The surveys list site index (SI) for soil types containing forest vegetation at the time of the survey. Additionally, site index data presented by Woolery, Olson, Dawson, and Bollero (2002) were also used where soil types in the study area matched soil types used in the Woolery study.

Tree species used to estimate site index varied greatly among the soil series. To find the best estimate an interspecies site index relationship was developed between species listed. This relationship was developed by computing a regression equation for two species occurring on the same soil class (Avery and Burkhart 2002). Site index values were recorded in eastern white oak (*Quercus alba*) equivalents. Where a site index score was only available for species other than white oak, a regression equation was used to correlate the appropriate tree species to white oak site index. Where site index measurements were not present and were not available, the State Soil Geographic (STATSGO) database was utilized (USDA 1994).

Along with site index information, USDA soil surveys include a cubic meters per hectare estimate of fiber production. Correlations were drawn based on the relationship stated by the USDA between site index and fiber production. The correlation equation was then used in Spatial Analyst to convert county-wide site index values to county-wide fiber production per acre values.

A gross income layer was created for each land use. Gross income was calculated in a GIS by multiplying unit production level of each pixel times the per unit price of each commodity. Per unit prices were taken from the Illinois Agriculture Statistics Service (IASS 2003) 2002 reports on 2001

Table 1.—Commodity price, consumer price index (CPI) conversion factor, and 1997 dollar equivalents.

Commodity	Unit of Measure	2001 Price Per Unit	CPI Conversion Factor	1997 Dollar Equivalents
Corn	Bushels	\$2.05	0.906	\$2.26
Soybeans	Bushels	\$4.50	0.906	\$4.97
Hay	Tons	\$88.00	0.906	\$97.13
Forest (uplands)	Mbf	\$180.00	0.906	\$198.68
Forest (bottoms)	Mbf	\$101.00	0.906	\$110.38

prices. 2001 prices were used to match available land use data and then deflated to reflect 1997 prices to run with the available IMPLAN model. Prices used in the analysis appear in Table 1.

Net income or return to farm layers, were calculated in a GIS by subtracting from the gross income layer, all expenses incurred with producing the respective land use. To measure incurred expenses, an enterprise budget was constructed for each land use. Corn, soybean, and hay budgets were created by the University of Illinois extension office (UIUC Farmlab 1999) for Illinois agriculture, and were then modified by researchers at Southern Illinois University to more accurately represent expenses for the southern Illinois region (see Peterson 2003). Forest budgets were constructed based on information obtained from the Illinois Department of Natural Resources (IDNR) district forester for St. Clair County. The IDNR provided estimates of inputs necessary to establish and manage a one-acre forest stand for a fifty-year rotation (Brown 2003).

Economic Analysis of St. Clair County

Impact Analysis for Planning (IMPLAN) is used to conduct an economic analysis and comparison of St. Clair County under the current and income-maximizing land use allocation. Total input, total output, total value added, and total employment were measured to compare economic impacts. Total input values were derived based on land use allocation, land use unit production, and price per unit. This value was then input in IMPLAN, which calculates output, value added, and employment for each land use.

A national IMPLAN model was used in this analysis to simulate a closed economic system within St. Clair County. Regional Purchase Coefficients (RPC) for each sector in IMPLAN identify the percentage of purchases that are made within the region of analysis. RPCs within the primary sectors of this analysis for the national model are 0.90 or greater.

Results and Discussion

GIS-Based Analysis

Graphical results showing a comparison between the current land use allocation and the income-maximizing allocation scenario are provided in Figure 2. GIS analysis of land use in St. Clair County reveals a large difference between current and income maximizing land use allocations. Currently, there are about 8.2 million bushels of corn and 3.4 million bushels of soybeans produced on a combined 1.64 hundred thousand acres, or 38 percent of the 4.30 hundred thousand acre county. 1.99 thousand tons of hay are currently produced on 925 acres. Annually, over 164 million board feet of fiber are produced on 42.4 thousand acres within St. Clair County.

The income-maximizing solution yields a reduction in row crops (corn and soybeans) and an increase in both hay and forest production. Corn production would be severely reduced to approximately four thousand acres, or a total of just under 63 thousand bushels. Reductions in soybean production are not as severe, however only 36 thousand acres would be allocated for the production of nearly 1.36 million bushels. A change toward the income maximizing land use would stimulate the greatest increase in hay production of 460 thousand tons on over 100 thousand acres. Forest production would also increase to 260 million board feet fiber per year on 62 thousand acres.

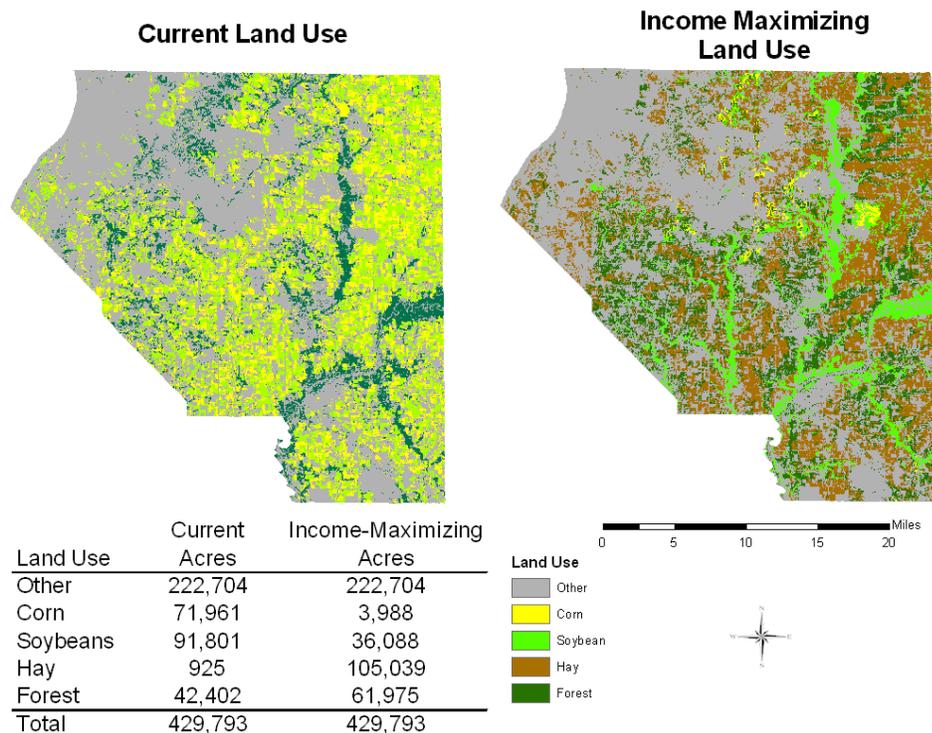


Figure 2.—Current versus income maximizing land use in St. Clair County.

Economic Impact Analysis of Current versus Income-Maximizing Land Use in St. Clair County

Economic impact analysis of the income-maximizing land use scenario indicates that, in the four categories of economic analysis (total input, total output, total value added, and total employment) great opportunities exist for increased income generation (table 2). Total input for corn and soybeans would be reduced by approximately \$27 million under the income maximizing allocation, however a \$44 million hay increase and a \$26 million forestry increase create a \$43.7 million net expansion in production.

IMPLAN results indicate that total output, or value of products created from the land use inputs, would expand by over \$32 million under the income-maximizing land use allocation. Total value added throughout the economy would also likely expand by more that \$14 million. Finally, under a shift to the income-maximizing land use allocation, St. Clair County would likely benefit in the creation of 724 jobs.

Conclusions

Land use allocation has a large impact on the economic and environmental health of a region. While most land use decisions are based primarily on economic returns, failure to maximize returns creates two economic dilemmas: 1) a reduction in income from primary production and value-added processing within the county, and 2) a decrease in economic exchanges among businesses within the county. All too often, landowners underestimate or overlook the productive potential (and income potential) of forest lands. In fact, results of this analysis suggest that forestland cover is substantially less than optimal in the St. Clair County study area.

While economic return-to-landowner values were central to the results in this analysis, other land use development strategies can easily be employed in the model. Values can be altered based a potential increase in demand for an agricultural crop, a need to maximize job creation, water quality objectives, aesthetic considerations, etc. Results in this analysis suggest that St. Clair County agriculture is better

Table 2.—Value and benefits of current and income maximizing allocation of land use.

Economic Measure	Land Use	Current	Maximizing	Net Benefit from switching to income maximizing
Input	Corn	\$18,363,801.00	\$1,423,687.00	-\$16,940,114.00
	Soybeans	\$17,020,227.00	\$6,751,784.00	-\$10,268,443.00
	Hay	\$191,274.00	\$44,669,115.00	\$44,477,841.00
	Forest	\$24,914,807.00	\$51,394,885.00	\$26,480,078.00
	Total	\$60,520,109.00	\$104,239,471.00	\$43,719,362.00
Output	Corn	\$43,924,849.00	\$3,793,748.00	-\$40,131,101.00
	Soybeans	\$44,468,663.00	\$17,640,352.00	-\$26,828,311.00
	Hay	\$491,842.00	\$30,005,964.00	\$29,514,122.00
	Forest	\$65,608,777.00	\$135,339,418.00	\$69,730,641.00
	Total	\$154,494,131.00	\$186,779,482.00	\$32,285,351.00
Value Added	Corn	\$22,523,107.00	\$1,945,300.00	-\$20,577,807.00
	Soybeans	\$24,162,291.00	\$9,585,061.00	-\$14,577,230.00
	Hay	\$260,408.00	\$15,886,767.00	\$15,626,359.00
	Forest	\$32,256,601.00	\$66,539,719.00	\$34,283,118.00
	Total	\$79,202,407.00	\$93,956,847.00	\$14,754,440.00
Employment	Corn	473.0	40.9	-432.1
	Soybeans	546.5	216.8	-329.7
	Hay	11.2	684.1	672.9
	Forest	765.7	1,579.5	813.8
	Total	1,796.4	2,521.3	724.9

suiting for the production of hay and rangeland in opposition to row cropping. Forestry is already well established within the county, and forest industry expansion, in light of modern water quality concerns, would likely provide economic and environmental benefits.

A major contribution of the methodology developed in this paper lies in the functionality of the model for creating spatially explicit input data for use in IMPLAN, a proven I-O economic analysis tool. This new generation of spatially explicit economic impact analysis will allow field and pixel-level analysis, resulting in the ability to simultaneously examine economic and environmental impact at a level of resolution previously unavailable.

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MANAGING MIXED SPECIES HARDWOOD STANDS FOR MAXIMUM FINANCIAL RETURNS

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ABSTRACT – Mixed yellow-poplar-oak stands were simulated on a computer using forward recursive dynamic programming to find financially optimal stand density, composition, and rotation length. Percent composition of oak was varied, and the marginal value of each thinning was determined as well as the marginal value of different types of thinning strategies. Improvement thinning and thinning from above were found to be the most valuable strategies, with the first thinning increasing soil expectation value by 11 to 19%, with subsequent thinnings improving SEV by less than 5%.

Introduction

Yellow-poplar-oak stands are among the most productive and valuable in the central and southern Appalachians and plateau regions of the eastern United States. This coverteype is dominated by fast-growing yellow poplar (*Liriodendron tulipifera*), and slower growing, but more valuable (in terms of per unit volume) white oak (*Quercus alba*) and northern red oak (*Quercus rubra*). Financial optimization of this forest type involves questions not only of rotation length and total stand density, but management of the composition of the forest over the rotation. General characteristics and management of the yellow-poplar-oak coverteype were presented by Beck and Della-Bianca (1981). The approach used in this research is based on computer simulation using dynamic programming to find optimal financial regimes.

Methods

Forward-recursive dynamic programming (DP) has been widely applied to solving optimal stand density and rotation length questions (Amidon and Akin 1968, Kao and Brodie 1980, Arthaud and Klemperer 1988, Pelkki 1998). The basic formulation of dynamic programming permits an efficient search of all stand states as defined in a network. Forward recursive dynamic programming starts at some set of initial conditions and moves forward (in time) towards final harvest.

Mathematically, dynamic programming starts with an objective function (1) which sums

$$f_N(Y_N) = \sum_{i=0}^N r_i(T_i) \quad (1)$$

where: Y_N = state vector describing stand state at age N

$r_i()$ = transformation function that returns value of management
action T taken at time i

the present value of all harvesting actions from the initial condition to the final harvest.

To move the stand between stages (the intervals at which management activities can take place) a transformation function (2), “grows” the stand from one stage to the next.

$$Y_i + G_i(Y) = X_{i+1} \text{ (for } i = 0, 1, 2, 3, \dots, N-1) \quad (2)$$

where $G_i()$ = the growth transformation function or vector that grows
a state in stage n to a state in stage n+1

X_i = a state that has been grown but has not had any management action (T_i) upon it.

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Table 1.—Stand tables with number of trees per acre by 1-inch diameter class by species for low-oak, medium-oak, and high-oak composition forests at age 20 years.

Diameter Class	Number of trees per acre at age 20 by initial oak composition class								
	Low oak			Medium oak			High oak		
	White oak	Red oak	Yellow-poplar	White oak	Red poplar	Yellow-oak	White oak	Red oak	Yellow-poplar
3	32	32	110	63	64	73	94	96	37
4	21	32	134	42	64	89	63	96	45
5	9	19	94	19	38	62	57	57	31
6	4	8	52	8	16	34	24	24	17
7	1	4	24	2	7	16	11	11	8
8	1	11		2	7	3	3	4	
9		4			2			1	
10		2			1			1	
11		1			1				

Connecting the states in a stage before management to their “after management” condition is equation 3, which applies the silvicultural activities (T_i) to the stand. Ending conditions are specified by equations 4 and 5 and simply states that the final management action is a clearfelling (even-aged management).

$$X_i - T_i = Y_i \quad (3)$$

$$X_N - T_N = 0 \quad (4)$$

$$Y_N = 0 \quad (5)$$

The dynamic programming network is a sequential decision making process that searches a network of all admissible defined stand states, stage by stage until an ending condition is reached. Linking the stages together is a recursive equation (6) that

$$f_N(Y_N) = \max_{(Y_{i-1}, T_i)} [r_i(X_i, T_i) + f_{i-1}(Y_{i-1})] \quad (6)$$

is based on the principle of optimality (Dykstra 1984), which states that “given the current state of the system, an optimal policy for the remaining stages is independent of any policy adopted in previous stages.” In other words, states are defined precisely enough so that the previous actions taken to arrive at a particular state do not affect future management decisions.

The practical application of dynamic programming requires that state neighborhoods (Kao and Brodie 1984, Arthaud and Klemperer 1988, Pelkki 1997) be used to change continuous state variables (number of trees per acre and net cubic foot volume per acre) to discrete state intervals. In this study, state neighborhoods were defined by basal area (ft.²/acre), net volume (ft.³/acre), and percent oak composition, determined by all oak basal area divided by total stand basal area. Intervals for the state variables were: ± 20 ft.² basal area per acre, ± 20 ft.³ volume per acre, and $\pm 2.5\%$ oak composition. The growth interval between stages was set at 5 years.

The objective function optimizes net present worth ($r_i()$ – NPW function), and the stopping condition was found by maximizing soil expectation value (the present value of an infinite number of rotations). Thus, once SEV declined, the DP algorithm stopped.

The growth transformation function was provided by the northeast variant of TWIGS, an individual-tree growth model (Miner et. al. 1988, Yaussy and Gale 1992, Yaussy 1993). Initial, 20-year old stand diameter distributions for site index 80 (base age = 50, base species = red oak) were derived from published data (Schnur 1937, Beck and Della-Bianca 1981) and are presented in table 1. Tree grade distributions were derived from the Forest Service permanent plot data from the Eastwide database and

Table 2.—Percent of stems by potential tree grade.

Species	G1	G2	G3	BG
Yellow-poplar	30	22	26	22
Red oak	22	31	26	21
White oak	21	26	22	31

Table 3.—Definition of harvest actions as executed by NESTER DP software.

Harvest Action	Abbreviation	Definition
Do nothing (no action)	N	No removal of stems beyond normal mortality from growth model.
Clearfelling	CC	All stems > 1" DBH are removed from stand. Merchantable stems (> 5") have positive value; pre-merchantable stems are removed with cost.
Thinning from above	TA	Stems in the stand list are removed from largest to smallest diameter until target basal area to be removed is reached.
Thinning from below	TB	Stems in the stand list are removed from smallest to largest diameter until target basal area to be removed is reached.
Thinning from above and below	TAB	Target basal area to be removed is determined. 50% of this target is removed in a thinning from below, 50% is removed in a thinning from above.
Improvement thinning	TI	Stems are removed until the target basal area is reached. First, below-grade trees are removed from largest to smallest diameter. Then trees are removed from smallest to largest diameter in grade three and grade two quality classes. Then non-grade trees are removed in ascending diameter. Finally, grade 1 trees are removed in ascending diameter.
High-grading	HG	Stems are removed until the target basal area is reached. Sawtimber trees are removed in descending grade (1 to 3), and harvested by descending diameter within each grade. Once all grade sawtimber is removed, the remaining trees are harvested by descending diameter.

are presented in table 2. Applying the TWIGS program to these initial conditions resulted in growth and yield projections that were consistent with other data (McGee and Della-Bianca 1967, Beck and Della-Bianca 1972 and 1975, Schlaegel et al 1969, Knoebel et. al. 1986) and suitable for internal comparisons.

Management actions (T_i) possible in the DP simulation included thinning from above (TA), thinning from below (TB), thinning from above and below (TaB), improvement thinning (TI), high-grading (HG), do nothing (N) and clearfelling (CC). Table 3 provides a brief definition of how each harvest treatment was modeled by the DP software. Thinning was permitted in 10% increments, from 10% to 50% of the stand's initial basal area. In each thinning operation, oaks could be strongly favored (all removals first in non-oak), slightly favored (two-thirds of removals in non-oaks), or not favored (thinning done regardless of species). Timber stumpage prices were obtained from Timber Mart-South and Kentucky State Timber price reports and are presented in table 4. A real price increase of 1% per year for sawtimber and 0.5% per year for pulpwood was assumed, and a real 5% discount rate was used to convert future returns to present dollars. A fixed cost of \$80 per acre was included for all intermediate and final harvests, and penalty was charged in the removing of pre-merchantable (dbh < 5 in.) stems. Annual management costs (taxes, interest charges, administration costs) were assumed to be equal to annual revenues (hunting and recreation leases). For a detailed description of the management parameters, see (Kirillova 2001).

Table 4.—Initial stumpage prices by species group and product class.

Species	G1 (\$ / MBF)	G2 (\$ / MBF)	G3 (\$ / MBF)	BG (\$ / MBF)	Pulp (\$/Mcf)
Yellow-poplar	160	110	60	30	240
Red oak	300	215	120	60	110
White oak	250	180	100	50	110

Table 5.—Optimal management regimes for high-, medium-, and low-oak composition stands under dynamic programming simulations with unrestricted thinnings.

Oak Composition	SEV (\$/ac)	Rotation Length	Year	Thinning regime Strategy	Oak Preference	Percent BA removed
High	\$ 207.7	70	30	TI	High	50
			40	TA	None	35
			45	TI	High	35
			55	TA	None	40
Medium	199.7	75	30	TI	High	50
			45	TA	None	25
			55	TI	High	45
			60	TA	None	30
Low	192.8	90	30	TI	None	20
			45	TA	None	45
			55	TI	None	30
			60	TA	None	30
			70	TI	Moderate	45
			80	TA	None	45

DP simulations were run to determine the value of thinning strategies, the number of allowed thinnings and the effects of thinning on the production of high-quality oak stems in stands with varying levels of oak composition.

Results

Without any intermediate harvests, stands with low-oak composition had rotation ages of 45 years and a soil expectation value (SEV) of \$158.8 per acre. Medium-oak composition stands had a rotation length of 45 years and an SEV of \$158.3 per acre. The high-oak composition stand had a rotation age of 40 years and an SEV of \$172.2 per acre under the “no thinning” options.

Optimal rotation schemes with thinning

Simulations were run without restricting either number of thinnings or the thinning strategy employed. Table 5 provides the optimal rotation value and harvest schedule for stands with high-, medium-, and low-oak compositions. Greater oak composition, results in greater SEV and shorter rotations. The general pattern of the thinning regimes is two or three sequences of improvement thinnings (TI) followed by thinnings from above (TA) without any special preference to retaining oak in the stand. In only one instance, low-oak composition and the third improvement thinning, was there slight preference given to retaining oak

Value of thinning strategies

Five DP simulations were run under each of the tree levels of oak composition allowing only one thinning strategy but not restricting the number or intensity of the thinnings. Table 6 provides the SEV and rotation age for each oak composition level allowing one particular strategy.

Table 6.—Value of individual thinning strategies on high-, medium, and low-oak composition stands.

Oak Composition	Thinning strategy	SEV	Rotation Age
High	TI	201.2	50
	TA	181.7	90
	HG	176.0	55
	No thinning	172.2	40
Medium	TI	188.4	65
	TA	184.3	85
	HG	175.2	75
	TaB	161.6	65
	No thinning	158.3	45
Low	TA	178.6	95
	TI	175.6	55
	HG	170.6	90
	TaB	162.9	60
	No thinning	158.8	45

Table 7.—Effect of number of allowed thinnings of any type on high-, medium-, and low-oak composition stands.

Oak Composition	Number of thinnings	Thinning strategies used	SEV (\$/ac)	Rotation Length
High	0	—	172.2	40
	1	TI	201.2	50
	2	TI	201.2	50
	3	TI	201.2	50
	4	TI, TA	207.7	70
Medium	0	—	158.3	45
	1	TI	184.3	50
	2	TI	188.3	65
	3	TI, TA	192.8	65
	4	TI, TA	199.7	75
Low	0	—	158.8	45
	1	TI	175.6	55
	2	TI, TA	179.9	75
	3	TI, TA	188.9	75
	4	TI, TA	191.1	80
	5	TI, TA	191.9	85
	6	TI, TA	192.8	90

For high- and medium-oak composition stands, improvement thinning is the single most valuable strategy, increasing SEV by 17% and 19%, respectively, while increasing rotation length by 10 and 20 years, respectively. The most valuable single thinning strategy for stands with low-oak composition was thinning from above, which increased SEV by 12% and rotation length by 50 years.

The second most valuable thinning strategy for high- and medium-oak stands is thinning from above, increasing SEV by 6% and 16%, respectively and rotation length by 50 and 40 years, respectively. For low-oak composition stands, the second most valuable thinning strategy is improvement thinning, which increased SEV by 11% and rotation length by 10 years.

After improvement thinning and thinning from above, high grading (HG) improves SEV in stands with all levels of oak composition. Increases in SEV for high-, medium- and low-oak stands are 2%, 10%, and 7%, respectively, while rotation length increases by 15, 30, and 45 years, respectively.

Thinning from above and below (TaB) improves SEV in medium- and low-oak composition forests. SEV gains are low, 2% for medium-oak and 3% for low-oak stands, while increasing rotation length by 20 years and 15 years, respectively. Thinning from above and below did not provide financial gains in high-oak stands.

Finally, the strategy of thinning from below (TB) did not result in any financial improvements in any of the stands.

Value of number of thinnings

Additional DP runs were made constraining the number of thinnings, but not restricting the thinning strategy employed. Table 7 provides the results of this sensitivity analysis.

For all levels of oak composition, the thinning strategy used when only one thinning was allowed was improvement thinning. Improvement thinnings increased SEV by 11% to 19%, while increasing rotation length by 5 to 10 years.

For high-oak composition forests, increasing the number of allowed thinnings to 2 or 3 has no effect on the optimal rotation scheme. Increasing the number of thinnings to four increases SEV by an additional 3% and rotation length by another 20 years, with improvement thinnings and thinnings from above employed in sequence as shown in table 5. Allowing more than 4 thinnings did not improve SEV for high-oak stands.

In medium-oak composition forests, a second improvement thinning increased SEV by an additional 2% over a single improvement thinning, and adds an additional 15 years to the rotation length (Table 6). Allowing three thinnings permits thinning from above to be employed, increasing SEV by another 2% without increasing rotation length. Finally, allowing four thinnings produces the management sequence for medium-oak shown in table 5, with a 4% increase in SEV and a ten year addition to the rotation length.

A second thinning in low-oak composition stands increases SEV by 2% while increasing rotation length by 20 years. Allowing 3, 4, 5, and finally 6 thinnings permits sequences of and improvement thinning followed by thinning from above, culminating with the low-oak rotation scheme shown in table 6.

Number of thinnings and output of quality timber

The effect of a single improvement thinning in high- and medium-oak stands is demonstrated by figures 1 and 2. A single improvement thinning results in a dramatic increase in the production of sawtimber volume from grade 1 and grade 2 trees. In high-oak stands, a single improvement thinning causes the percent of total sawtimber volume produced from grade 1 trees to increase from 8% to 32% and the production of sawtimber from grade 1 and grade 2 trees increases from 50% to 90% of total sawtimber production. In medium-oak stands, a single improvement thinning increased percent volume production from grade 1 trees to increase from 10% to 27%, and volume production from grade 1 and grade 2 trees increased from 47% to 85%.

In low-oak stands, the effects of the first improvement thinning are less dramatic (Figure 3). Production of volume from grade 1 trees increases from 7% to 15%, and volume from grade 1 and grade 2 trees increases from 41% to 53%. The number of thinnings that maximizes volume production from grade 1 trees involves an improvement thinning followed by a thinning from above (23% of volume from grade 1 trees). Three thinnings (two improvement thinnings and one thinning from above) results in the maximum percent of volume production from grade 1 and grade 2 trees (80%) in low-oak stands.

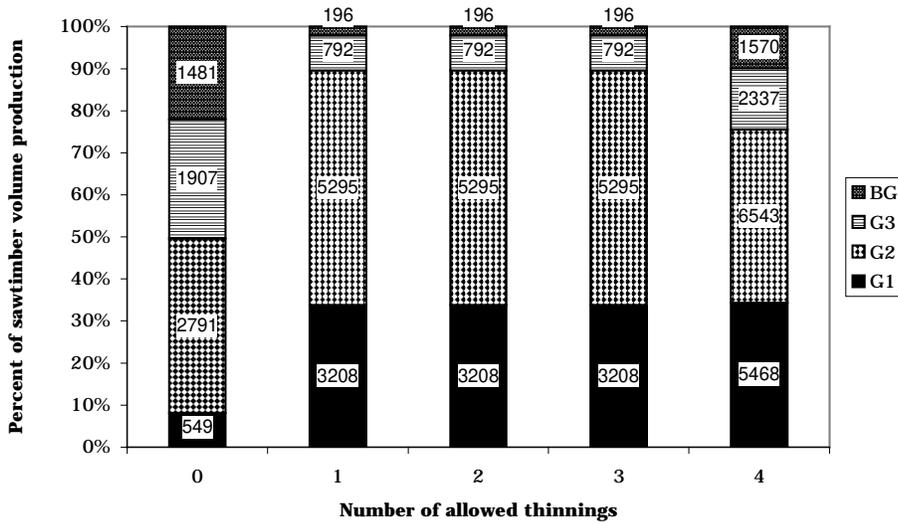


Figure 1.—Volume production for high-oak stands by number of allowed thinnings.

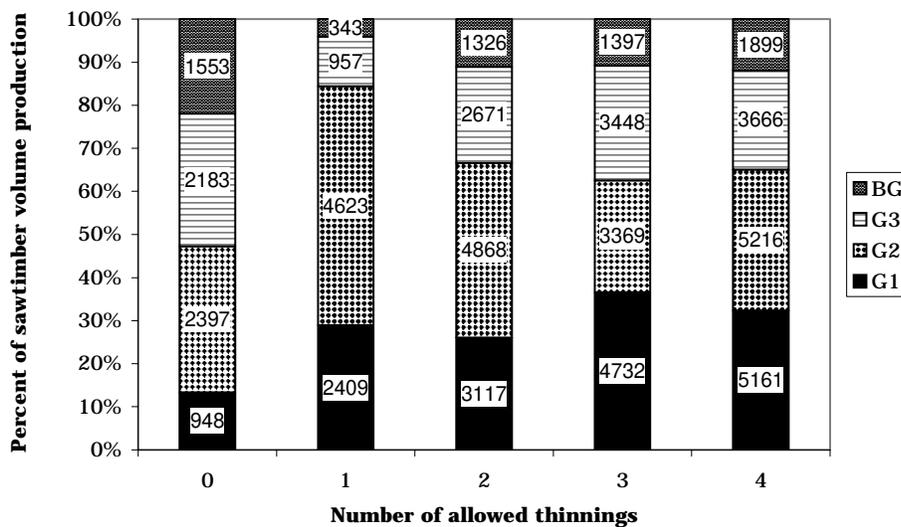


Figure 2. —Volume production for medium-oak stands by number of allowed thinnings.

Discussion

Clearly, the simulation results point to the potential gains from improvement thinning. Improvement thinning removes the lowest quality trees first, favoring future growth on the most valuable stems. In stands where oak is a dominant component of the forest, the price premiums for grade 1 and grade 2 oak trees (table 4) provide economic incentives for focusing volume production heavily on high-quality oak stems. However, even in stands dominated by yellow-poplar, which has a lesser price premium for high-quality stems (table 4), improvement thinning is still the best choice if only one thinning is to be employed.

All thinnings increase rotation age, but improvement thinning typically adds 5 or 10 years to the rotation while thinning from above adds from 20 to 50 years to the rotation. This can be explained by the strategies themselves. Improvement thinning removes the poorest quality stems (below grade) from largest to smallest, and if additional stems must be removed, stems are removed smallest to largest within each grade class, starting with grade 3 and moving towards grade 1. Thus, improvement thinning is similar to thinning from below, but favoring high-value stems, and as such, it will not lengthen the time the dominant stems in a stand reach financial maturity. Thinning from above, however, removes the largest diameter stems, regardless of quality, in an effort to release intermediate stems and capture economic returns early in the rotation. The residual stems are smaller and require a longer rotation to reach financial maturity.

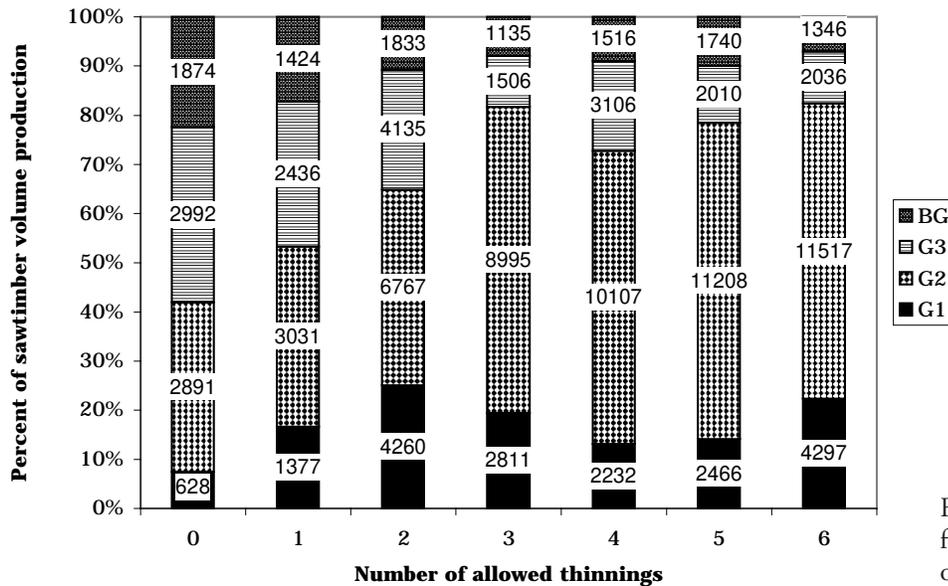


Figure 3.—Volume production for low-oak stands by number of allowed thinnings.

The advantages to this computer simulation approach lie in the ability to rapidly simulate a variety of management regimes quickly and conduct sensitivity (with and without) analyses. Weaknesses to this approach are that the input stands are hypothetical and presumed “typical” 20-year old yellow-poplar-oak stands. Also, the growth and yield data presented by this model should be applied with caution. While the yields were compared and found reasonable with other published data, the individual-tree models upon which they are based are not well suited for making long-term growth projections of more than 30-40 years. The outputs can be compared internally for judging the value of different actions. However, this is a caution that should be applied whenever modeling is used to make silvicultural recommendations.

When applying these results, it should be noted that the optimal simulations included 4-6 thinning entries, but the majority of economic gains were achieved with three or fewer thinnings. Considering that oak was retained in all simulations until the final regeneration felling, it possible, but by no means guaranteed, that successful natural regeneration of yellow-poplar-oak stand will ensue.

Acknowledgments

The authors would like to acknowledge the support of the Kentucky Agricultural Experiment Station and the Arkansas Forest Resources Center for supporting this research work.

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THE POST-HARVEST COMPETITIVE ENVIRONMENT IN 13 MIXED-OAK STANDS IN PENNSYLVANIA

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Abstract.—The post-harvest competitive environment in 13 mixed-oak stands in Pennsylvania was characterized using data from a repeated-measurement study. Competitors were defined as the tallest non-oaks that fell within milacre plots. Red maple was the most common competitor one and four years after overstory treatments on both high and low quality sites. Species that experienced rapid early growth became more common competitors after overstory treatments, accounting for about one-third of the competitors on high quality sites. The mean and median heights of competitors four years after treatment were 5.2 ft and 4.5 ft, respectively, on low quality sites, and 4.7 and 3.5 ft, respectively, on high quality sites. The percentage of oaks that remained competitive (oak height \geq non-oak height) declined on high quality sites, probably due to initially small advance oak regeneration. In total, the competitive environment in Pennsylvania appears to be less severe than that reported for other regions, suggesting that large stems of advance oak regeneration may not always be necessary to regenerate oak stands in this part of the Central Hardwood region.

Introduction

The recognition of an oak regeneration problem in the eastern United States has motivated forest managers to focus on obtaining adequate advance oak regeneration before harvesting oak stands (Lorimer 1993). The pursuit of adequate regeneration begs the question of what is “adequate.” In a general sense, adequate regeneration describes a population of seedlings that has a high likelihood of capturing a desirable portion of the growing space made available by an overstory removal (Larsen and Johnson 1998). An important factor influencing the amount of growing space captured by oaks is their ability to compete with other species after harvest. It is generally understood that oaks must grow at least as fast as neighboring trees to remain competitive. Consequently, the height growth of competing trees plays an important role in determining the post-harvest competitive environment.

Several influential works have engendered the belief that oaks experience a uniformly severe post-harvest competitive environment (Sander 1972; Sander et al. 1984; Loftis 1990). As a result, it is often assumed that advance oak regeneration must be large (e.g., \geq 4.5 ft) and grow rapidly to compete after harvest (Sander 1972). While this assumption appears valid on productive sites where oaks must compete with fast-growing associates, the importance of rapid early growth is less clear on the moderately productive sites that are common in Pennsylvania and other parts of the Central Hardwood region. Others have demonstrated that modest growth rates can lead to oak regeneration success in environments with low levels of competition (Johnson et al. 1989).

The purpose of this paper is to characterize the post-harvest competitive environment in mixed-oak stands in Pennsylvania. Three questions are addressed: 1) Which species commonly compete with oaks after an overstory removal, and do these species have the potential to exclude oaks during early stand development? 2) Does the competitive status of oak regeneration generally improve, decline, or remain stable after an overstory removal? 3) What rate of post-harvest growth is required for oaks to remain competitive after harvest?

Methods

Data were collected in 13 mixed-oak stands as part of a repeated-measurement study focused on regeneration success on State Forest lands in Pennsylvania. Stand descriptions are given in Table 1.

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Table 1.—Description of the 13 stands included in the study. Site classes are site index < 70 (low) and site index ≥ 70 (high).

Stand	Site Class	Mean DBH (in)	Basal Area (ft ²)		% BA in Oak	Dominant <i>Quercus</i> Species
			Pre-Harvest	Post-Harvest		
9601	Low	9.4	93	22	89	<i>Q. prinus</i>
9701	Low	7.6	90	27	76	<i>Q. rubra</i> – <i>Q. prinus</i>
9706	Low	10.1	112	19	82	<i>Q. rubra</i>
9717	Low	7.2	82	18	73	<i>Q. prinus</i> – <i>Q. velutina</i>
9806	Low	7.6	120	6	91	<i>Q. prinus</i> – <i>Q. velutina</i>
9603	High	8.3	108	27	78	<i>Q. rubra</i>
9604	High	7.3	110	35	60	<i>Q. prinus</i>
9607	High	6.6	93	11	78	<i>Q. prinus</i> – <i>Q. alba</i>
9612	High	7.3	106	13	80	<i>Q. prinus</i> – <i>Q. coccinea</i>
9711	High	5.8	77	16	55	<i>Q. prinus</i>
9715	High	5.5	105	19	53	<i>Q. rubra</i> – <i>Q. prinus</i>
9716	High	6.9	79	15	70	<i>Q. prinus</i> – <i>Q. rubra</i>
9804	High	6.4	108	17	73	<i>Q. prinus</i>

Eleven of the 13 stands are located in the Ridge and Valley physiographic province, while the other two (stands 9706 and 9711) are located in the Appalachian Plateaus province. Harvests removed from 70 to 95 percent of the stands' initial basal area.

Permanent plots were established in each stand before harvest. Plots were placed on regular grids, with plot centers falling at least 209 ft apart. Within each plot, four circular milacre (43.56 ft²) subplots were established 16.5 ft from plot center along the N, E, S, and W azimuths. Regeneration data were collected in the subplots. In total 1,291 plots were measured approximately one year before harvest, and remeasured one year and four years after harvest (each plot was measured three times).

Although a great deal of data were collected on each subplot, this paper focuses on the dominant (tallest) oak and non-oak regeneration. Hypothetically, these stems have the best chance of occupying the milacre subplot as the new stand develops. During the 1999 – 2003 field seasons, the dominant oak and non-oak were identified in each subplot and the species and heights were recorded. During the 1996-1998 field seasons, only a single dominant stem was measured and the species recorded. The missing data (either dominant oak or non-oak) were reconstructed from tallies of all regeneration by species and height class. Height classes were in 1 ft, or smaller, increments (smaller increments were used for stems < 1ft), which allowed heights to be reconstructed to the nearest foot. If only a single oak or non-oak (other than the dominant seedling) occupied the largest height class, this species was selected to augment the existing data. In cases where more than one species occupied the largest height class, the more abundant species was selected. Ties were broken at random.

For data analysis, stands were assigned to either a “low quality” class (site index < 70) or a “high quality” class (site index ≥ 70) and data in each class were analyzed separately. The high quality class included 765 plots and the low quality class included 526 plots. Six of the thirteen stands were fenced after harvest to exclude white-tailed deer (*Odocoileus virginianus* Boddaert). Data from fenced and unfenced stands were not analyzed separately because stand were fenced only if deer browsing was perceived to present a regeneration problem, and initial analysis failed to show significant differences between fenced and unfenced stands.

The composition of non-oak competitors was summarized for each period (pre-harvest, 1-yr post, and 4-yrs post) by calculating the frequency of occurrence, mean height, and median height of common non-oak species. The frequency of occurrence, mean height, and median heights of dominant oaks were also calculated. The competitive position of oaks was examined by tabulating, by period, the difference in height between dominant oaks and dominant non-oaks. The height growth required for oaks to

Table 2.—Frequency of occurrence, mean height, and median height of oaks and common competitors before and after overstory treatments on low quality sites.

Period	Species	Frequency (percent)	Height (ft)	
			Mean	Median
Pre-Harvest	Oaks	78.3	0.9	0.8
	All Competitors	93.0	1.4	0.5
	<i>Acer rubrum</i>	78.0	1.3	0.5
	<i>Acer pennsylvanicum</i>	3.6	3.6	2.5
	<i>Carya glabra</i>	3.0	2.2	0.6
	<i>Amelanchier spp.</i>	2.4	0.9	0.8
	<i>Pinus strobes</i>	1.5	2.2	2.0
1-yr Post	Oaks	83.9	2.2	1.5
	All Competitors	86.4	2.0	1.5
	<i>Acer rubrum</i>	67.7	2.2	1.5
	<i>Nyssa sylvatica</i>	2.4	2.0	2.5
	<i>Prunus serotina</i>	2.3	2.2	2.0
	<i>Amelanchier spp.</i>	2.1	1.8	1.5
	<i>Carya glabra</i>	1.9	1.5	1.1
4-yr Post	Oaks	87.2	5.0	4.5
	All Competitors	92.0	5.2	4.5
	<i>Acer rubrum</i>	70.2	5.4	4.5
	<i>Prunus serotina</i>	3.6	5.6	6.5
	<i>Acer pennsylvanicum</i>	2.4	8.4	7.5
	<i>Nyssa sylvatica</i>	2.1	7.6	7.5
	<i>Betula lenta</i>	1.9	5.5	3.5

maintain a competitive position was examined by calculating the proportion of subplots where oaks were competitive (i.e., oak height \geq non-oak height) four years after harvest for each dominant oak height class.

Results

Advanced oak regeneration occurred on 78 percent of subplots on low quality sites (Table 2). The percentage of subplots containing oak regeneration increased to 84 percent, one year after harvest, and to 87 percent, four years after harvest. Dominant oaks averaged 0.9 ft tall before harvest, and increased to 2.2 ft, one year after harvest, and 5.0 ft, four years after harvest.

On high quality sites, advanced oak regeneration occurred on 73 percent of subplots. This percentage generally remained stable one and four years after harvest (69 and 71 percent of plots, respectively). Dominant oaks averaged 1.3 ft tall before harvest, and increased to 1.6 ft, one year after harvest, and 3.2 ft, four years after harvest.

The composition of competitors changed little throughout the study period and differed little between site classes (Table 2 and Table 3). A competitor was present on nearly all of the subplots (86 – 99 percent) at all times in both site classes. Red maple (*Acer rubrum* L.) was consistently the most common competitor by a substantial margin. The occurrence of red maple as a dominant competitor on low quality sites ranged from 78 percent of subplots, before harvest, to 67 percent, one year after harvest. On high quality sites, red maple was the dominant competitor on 64 percent of subplots before harvest. This declined to approximately 50 percent of subplots one and four years after harvest.

Four years after treatment, species that experienced rapid early growth, such as black birch (*Betula lenta* L.), blackgum (*Nyssa sylvatica* Marsh.), and black locust (*Robinia pseudoacacia* L.), became more common competitors on the high quality sites (totaling 32 percent of subplots) (Table 3), but

Table 3.—Frequency of occurrence, mean height, and median height of common competitors before and after overstory treatments on high quality sites.

Period	Species	Frequency (percent)	Height (ft)	
			Mean (std. dev.)	Median
Pre-Harvest	Oaks	72.8	1.3	0.8
	All Competitors	96.0	2.0	0.5
	<i>Acer rubrum</i>	63.9	1.4	0.5
	<i>Betula lenta</i>	6.9	4.4	1.5
	<i>Acer pennsylvanicum</i>	6.1	5.7	4.5
	<i>Nyssa sylvatica</i>	5.8	1.5	0.8
	<i>Amelanchier spp.</i>	4.3	1.0	0.8
1-yr Post	Oaks	69.1	1.6	1.5
	All Competitors	95.0	1.8	0.8
	<i>Acer rubrum</i>	49.7	1.6	0.8
	<i>Nyssa sylvatica</i>	18.3	2.2	1.5
	<i>Betula lenta</i>	8.0	2.1	0.8
	<i>Sassafras albidum</i>	4.8	1.1	0.8
	<i>Robinia pseudoacacia</i>	4.5	3.4	4.4
4-yr Post	Oaks	70.9	3.2	2.5
	All Competitors	96.5	4.7	3.5
	<i>Acer rubrum</i>	50.8	4.0	2.5
	<i>Betula lenta</i>	14.0	5.2	4.5
	<i>Nyssa sylvatica</i>	12.4	6.7	6.5
	<i>Robinia pseudoacacia</i>	5.8	8.1	8.5
	<i>Sassafras albidum</i>	3.1	2.2	1.5

remained less common on the low quality sites (4 percent) (Table 2). Black cherry (*Prunus serotina* Ehrh.), which also experienced rapid early height growth, appeared among the common competitors on the low quality sites (4 percent).

The mean and median heights of competitors differed little between site classes (Table 2 and Table 3). The mean pre-harvest height was somewhat greater on the high quality sites versus low quality sites (2.0 ft versus 1.4 ft, $P = 0.001$ for two sample t-test), but differences between mean heights were not statistically significant one and four years after harvest. On high quality sites, the pre-treatment mean and median competitor heights were 2.0 ft and 0.5 ft, respectively. These heights remained almost unchanged one year after treatment. Four years after treatment, the mean competitor height reached 4.7 ft, with a median height of 3.5 ft. A similar pattern occurred on low quality sites, and four years after treatment the mean competitor height reached 5.2 ft, with a median height of 4.5 ft.

The competitive status of oaks remained stable on low quality sites (Table 4), but declined on high quality sites (Table 5). On low quality sites, dominant oaks were taller on approximately 40 percent of subplots throughout the study period. On high quality sites, the percentage of subplots where dominant oaks were taller decreased from 43 percent before harvest to 26 percent, four years after harvest. The percentage of plots where dominant competitors were taller increased on both high and low quality sites. Four years after harvest, dominant oaks and competitors were equal in height on a sizable percentage of subplots (about 15 percent on both high and low quality sites).

Figure 1 illustrates the percentage of dominant oaks that were competitive four years after harvest, by height class. There is little difference in the general trend between high and low quality sites. In both cases, there was a rapid increase in the percentage of competitive oaks with increasing height from 0.5 to 2.5 ft. Oaks > 3 ft were generally competitive on at least 50 percent of subplots. Oaks in the largest size class (>7 ft), were competitive on over 80 percent of subplots.

Table 4.—Distribution of height differences between dominant oaks and dominant competitors on low quality site.

Period	Oak Taller	Oak = Competitor	Competitor Taller
		— Percentage of Subplots —	
Pre-Harvest	39.2	29.4	31.4
1-yr Post	44.8	20.8	34.4
4-yr Post	39.1	15.1	45.9

Table 5.—Distribution of height differences between dominant oaks and dominant competitors on high quality site.

Period	Oak Taller	Oak = Competitor	Competitor Taller
		— Percentage of Subplots —	
Pre-Harvest	43.2	9.5	47.3
1-yr Post	39.3	18.0	42.7
4-yr Post	26.7	14.7	58.7

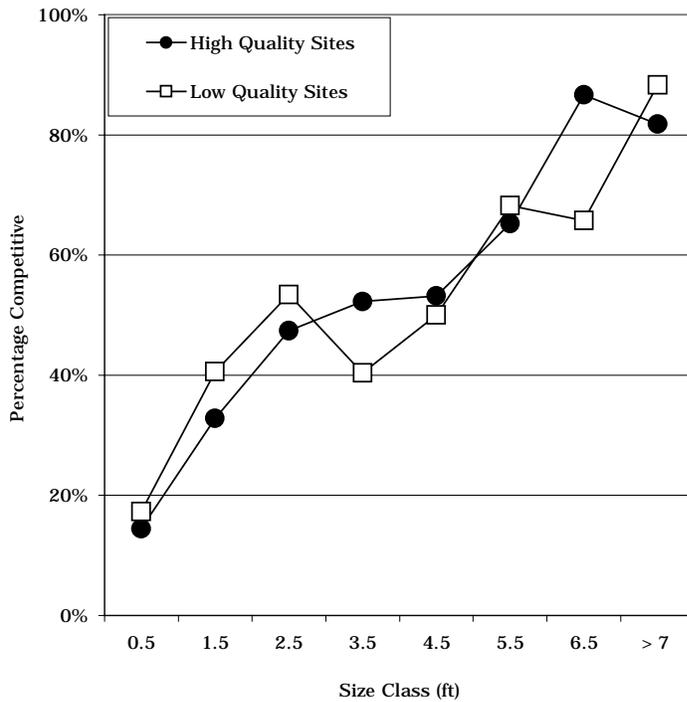


Figure 1.—Percentage of dominant oaks that remained competitive (oak height \geq non-oak height) four years after harvest, by size class.

Discussion

Our results suggest that the influence of competition on oak regeneration success may not be as great in some Pennsylvania stands as has been reported elsewhere. Although non-oak competitors were almost always present before harvest, they were generally small, averaging only 2.0 ft on high quality sites and 1.3 ft on low quality sites. The median competitor heights are also important to consider, since early competition is most likely to occur between trees sharing the same subplot. Before harvest, dominant competitors were ≤ 0.5 ft tall on one-half of subplots on both high and low quality sites. The small competitor sizes support the conclusions of others (McWilliams et al. 1995; McWilliams et al. 2002) that the condition of advance regeneration in Pennsylvania is generally depauperate.

Although the growth rate of oaks was not high, the percentage of subplots with oak regeneration did not decline after harvest on either high or low quality sites. Four years after harvest, oaks averaged only 5 ft tall on low quality sites and were smaller on high quality sites. However, dominant competitors were not much larger, averaging 5.2 ft on low quality sites, and 4.7 ft on high quality sites. Oaks experienced some loss of competitive position harvest on high quality sites, while their competitive position remained stable on low quality sites. The loss of competitive position on high quality sites appears to be the result of poor oak growth, rather than greater competitor growth. It is possible that advance oak regeneration was initially smaller in many subplots on high quality sites, a condition that has been reported elsewhere (Host et al. 1987). In addition, competition with non-tree vegetation may have been greater on high quality sites, slowing the growth of oak regeneration.

The composition of competitors appears favorable for the long-term success of oaks in these stands. Red maple and black birch, the two most common competitors, typically form a sub-canopy beneath oaks after the second decade of stand development (Oliver 1978). Overstory treatments did result in a shift in dominance towards species with rapid initial growth, particularly on high quality sites. However, this shift was of a much lesser magnitude than has been reported on more productive sites (Beck and Hooper 1986; Loftis 1990). Notably absent among common post-harvest competitors were species, such as yellow-poplar (*Liriodendron tulipifera* L.), that are clearly capable of sustaining height growth rates superior to those of oaks. Several common competitors (e.g., striped maple (*Acer pensylvanicum* L.) and serviceberry (*Amelanchier* spp.)) that established as advance regeneration are understory obligates, and their influence on stand development will be short-lived if other species succeed in growing past them. Consequently, most competitive exclusion of oaks by non-oaks is likely to be limited to a relatively short period following overstory removal.

The relationship between dominant oak heights and the percentage of oaks that were competitive four years after treatment suggests that a range of height growth rates may result in regeneration success. It appears plausible that some oaks that are relatively small, yet larger than any nearby competitor, will ultimately become part of the next stand. In similar stands, Ward and Stephens (1999) found oaks ranging from 5 ft to < 1ft tall before harvest reached a dominant or codominant crown position 12 yrs after harvest. Small oak seedlings have generally been discounted as a source of regeneration and are generally expected to succumb to the effects of competition. While this may happen in these stands, it remains unclear which trees would out-compete the oaks.

Conclusions

Although this study only spans a four-year period, initial results suggest that the post-harvest competitive environment in some mixed-oak stands in Pennsylvania is less severe than has been reported for other regions. With this in mind, the successful regeneration of some stands may depend more on the initial establishment and early growth of advance oak regeneration than on post-treatment competition. Oak stands surely cannot be regenerated without well-distributed advanced oak regeneration, or when competing non-tree vegetation, such as hayscented fern (*Dennstaedtia punctilobula* (Michx.) Moore), inhibits early seedling growth (Steiner and Joyce 1999). These factors, along with browsing by white-tailed deer, continue to be major impediments to oak regeneration in Pennsylvania. However, once these impediments are overcome, only modest height growth may be required for a substantial percentage of oaks to remain competitive during the years following harvest. Additional research is required to determine whether oaks will maintain their competitive position after four years, but the composition of competitors is generally favorable for the long-term success of oaks.

Acknowledgments

This research was funded in part by the Pennsylvania Department of Conservation and Natural Resources, Bureau of Forestry.

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THIRTY-ONE YEARS OF STAND DEVELOPMENT IN A NORTHERN RED OAK (*QUERCUS RUBRA* L.) PLANTATION, NORTH-CENTRAL OHIO, USA

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ABSTRACT.—Northern red oak stand development was monitored in a plantation at the Ohio Agricultural Research and Development Center (OARDC), Wooster, OH. By comparing stand development in this plantation with other stands (including those naturally regenerated), we attempt to evaluate the potential for ecosystem restoration of northern red oak forests. Approximately 4,000 seedlings were planted on this 1.9 ha site in November 1962 as a provenance test. Mortality rates were monitored during the autumns of 1962, 1963, and 1965, and were initially relatively high. Stem density (921.2 ± 40.3 stems ha^{-1} in 1968), basal area, and diameter were also periodically sampled until 1993, and these parameters exhibited expected patterns over the thirty-one year observation period. Examining long-term trends in plantations can provide insights into stand development patterns, expected growth rates and stand structures, and the potential for successful forest restoration.

Introduction

Across the Central Hardwoods Region, stands currently dominated by oaks are becoming less common due to regeneration failures. Long-term fire suppression and the increased mortality of oaks caused by gypsy moth (*Lymantria dispar* L.) defoliation and oak wilt [*Ceratocystis sogalearum* (Bretz) Hunt.], have resulted in many oak species being displaced by more shade-tolerant species (Hix and Lorimer 1991; Abrams and Nowacki 1992; Isebrands and Dickson, 1994). Additionally, urban expansion, conversion of mixed-oak forests to pasture and cropland, and accelerated harvesting has resulted in additional losses of oak stands across the Region (Abrams and Nowacki 1992; Johnson, 1993). This decline is particularly true for northern red oak (*Quercus rubra* L.) stands, which currently are being heavily harvested by private landowners because of the species' high lumber value and susceptibility to pests such as the gypsy moth. Consequently, the establishment and restoration of northern red oak stands are important issues facing resource managers in the Central Hardwoods Region.

Unfortunately, the silvical characteristics of northern red oak are such that it is often difficult to successfully regenerate these stands. As an intermediate species in shade tolerance with heavy seeds (Burns and Honkala 1990), northern red oak will not rapidly invade open areas such as old fields or out-compete other species following harvest operations or in closed canopy conditions. Acorn production is also often highly erratic from year to year, and predation losses due to insects, birds, and mammals can be very high. These characteristics when coupled with fairly high moisture requirements for early growth make northern red oak a prime candidate for restoration planting.

Although considerable research has been conducted over the past half-century on the regeneration dynamics and establishment patterns of northern red oak (e.g., Martin and Hix 1988; Nowacki et al. 1990), relatively little information on the early stand development of planted northern red oak stands is available. Since the goal of ecosystem restoration is to return a disturbed area to a successional trajectory that is similar to the area prior to disturbance (SER 2002), it is not clear whether planted stands of northern red oak over time begin to emulate the composition, structure and function of naturally regenerated northern red oak stands. In an effort to address this issue, we have begun to analyze growth and development data collected over the past 31 years from northern red oak

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plantations initially established as provenance tests at the Ohio Agricultural Research and Development Center (OARDC) located at Wooster, Ohio.

The objective of this paper is to summarize the early stand development patterns of one such stand of planted northern red oak located at OARDC, and compare these patterns with those of naturally regenerated northern red oak stands and plantations of the Central Hardwoods Region. With this information, we can begin to understand how these planted northern red oak stands develop over time and determine whether they are beginning to emulate the patterns of development exhibited in naturally regenerated stands.

Study Site and Methods

Mortality and growth have been monitored in the Apple Creek Provenance Test (AC-9) at the Ohio Agricultural Research and Development Center, Wooster, Ohio (40°45'N, 81°55'W) over the past 31 years. The 1.9 ha site is located along Apple Creek on fluvial terraces with Bogart loam, Euclid silt loam, and Orrville silt loam soils. These soils are classified as either moderately well drained or somewhat poorly drained.

In 1961, northern red oak seeds from 35 locations in 15 states and two Canadian provinces were collected and sown at a nursery in Green Springs, Ohio. In 1962, approximately 4,000 of these nursery-grown northern red oak seedlings were planted in seven randomized blocks totaling 1.9 ha (with double border rows). Sixteen trees were planted per plot at a 2.7 m by 2.7 m spacing. Seedlings that died in 1962-64 were replaced annually. As part of the provenance test maintenance, each block was fertilized (ammonium nitrate) in spring 1962, 1967 and 1968. Additionally, each block was treated with herbicide (Simazine and Atrazine) in 1965 and in 1968, mowed in 1967, and the spaces between rows were disked in the fall of 1968 in an effort to reduce competition.

Mortality was recorded in November 1962, October 1963, April 1965, and October 1965. However, seedlings that were dead were replaced until 1965. Consequently, we only present mortality rates since 1968. Stem diameter was measured in December 1968 (at 1 ft above the ground as many individuals were not yet 1.4 m tall), while the diameter at breast height (dbh; 1.37 m from the ground) of each stem was recorded in March 1983 and April 1993.

Results and Discussion

Mortality – After the first ten years of stand development (1962-1973), seedling survival was 54% (Kriebel 1976). Most of the mortality occurred prior to 1968. This level of survivorship is considerably lower than other studies from the region. McNeel et al. (1993) report first-year survival rates of 88% for planted northern red oak stems in northern West Virginia, and Pubanz et al. (1989) reported first-year survival rates of planted northern red oak seedlings in Wisconsin ranging from 85% in untreated stands to > 95% in cut and herbicide-treated stands. Johnson et al. (2002) report mean annual survival rates range from 96-99% for the red oak group (which includes northern red oak).

Most models of stand development predict that mortality rates peak during the stand initiation stage and then decline to 1-2% year⁻¹ at maturity (Buchman 1983; Bell 1997; Allen and Sharitz 1999). However, the specific mortality rates vary by species depending on specific life history traits (e.g., species longevity or shade tolerance). For example, Bell (1997) found that mortality rates were size- and species-specific, with smaller stems having a higher mortality rate. In the AC-9 stand, mortality rates have declined over time as the stand has progressed through the stand initiation and stem exclusion stages of development (Fig. 1). From 1968 (the point at which dead seedlings were not replaced) to 1983, the mean (\pm 1 SD) mortality rate was 1.5 (0.3) % year⁻¹. Over the next ten years (1983-1993), the mortality rate further declined to 0.8 (0.3) % year⁻¹. These rates are comparable to those reported by Lorimer et al. (1994) for treated (all understory trees >1.5 meters cut and stumps treated with herbicide) northern red oak stands on mesic sites in southern Wisconsin (1.8% year⁻¹ over the first four years following planting). However, it should be noted that in the untreated northern red oak stand sampled by Lorimer et al. (1994) seedling mortality was extremely high (72% over four

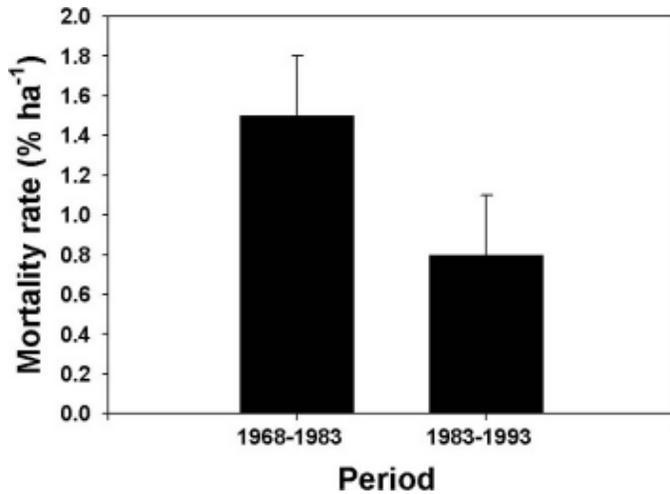


Figure 1.—Average (± 1 SD) mortality rate (% stems yr⁻¹) of northern red oak from 1968-1983 and 1983-1993 for the AC-9 northern red oak plantation, OARDC, Wooster, Ohio.

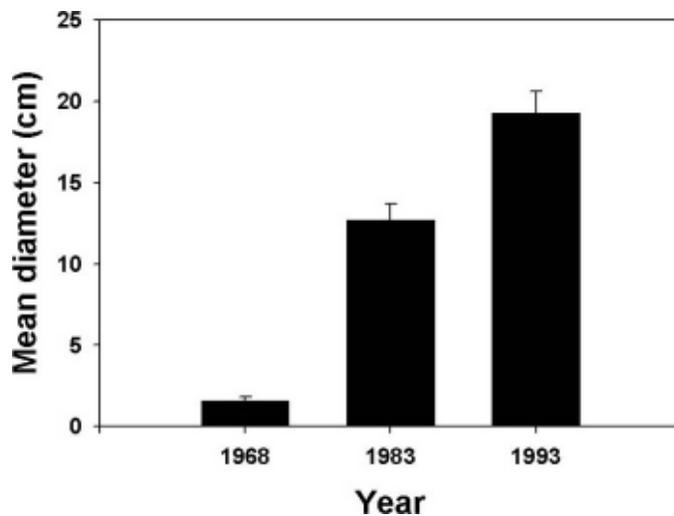


Figure 2.—Changes in mean (± 1 SD) diameter (dbh) from 1968 to 1993 for the AC-9 northern red oak plantation, OARDC, Wooster, Ohio.

years or 18% year⁻¹). In Illinois, Bell (1997) found a mortality rate of 3.1% year⁻¹ for trees ≥ 4 cm dbh in streamside forests along Hickory Creek and associated the high mortality with slow tree growth. Upland forests at this site were dominated by *Quercus rubra* that had a mortality rate of 1% year⁻¹ over 18 years. As a comparison, in old-growth forests of the Congaree Swamp National Monument in central South Carolina, background mortality rates for stems >10 cm dbh ranged 0.5 to 1.0% year⁻¹ (Allen and Sharitz 1999).

Diameter growth – In 1968, six years after planting, the mean dbh of the planted northern red oak stems was 1.6 (± 0.2) cm (Figure 2). By 1983, the mean dbh of the AC-9 stand had increased to 12.7 (± 1.0) cm, and by 1993 the mean dbh was 19.3 (± 1.3) cm. While the dbh has increased over time, diameter growth rates have declined. During the first twenty years of stand development, mean diameter growth was 0.74 (± 0.06) cm year⁻¹, and then over the next ten years (1983-1993) it averaged 0.66 (± 0.06) cm year⁻¹. Northern red oak growth rates in the AC-9 plantation greatly exceeded growth in forest understories in Wisconsin where the average dbh was 4.8 cm 26 years after a shelterwood cut (Martin and Hix 1988). More rapid growth would be expected where the overstory density is low and competition has been reduced by planting at wide spacing and there has been control of the competing vegetation. Northern red oak diameter growth rate, averaged over a wide range of initial tree diameters, as compiled from several studies by Johnson et al. (2002), was found to be 0.41-0.74 cm year⁻¹.

In natural northern red oak stands, somewhat similar patterns of diameter growth have been observed. For example, Abrams and Nowacki (1992) found that radial growth in northern red oak fluctuated in natural stands, initially increasing with tree size, followed by slower dbh growth rates until releases from suppression have occurred.

Density and basal area – Six years following the initial planting and four years following the last replacement of dead seedlings, mean stand density was 921.4 (± 40.3) stems ha⁻¹ (Fig. 3). Fifteen years later (in 1983), stem density had declined to 708.5 (± 37.2) stems ha⁻¹, and by 1993 density declined further to 617.5 (± 34.9) stems ha⁻¹ (Fig. 3). Thus, there was a decline of 14.1 stems ha⁻¹ year⁻¹ during the early portions of the stem exclusion stage, and a decline of 9.1 stems ha⁻¹ year⁻¹ during the later portions of the stem exclusion stage. Mean basal area in 1968 was 0.27 (± 0.05) m² ha⁻¹ and increased to 10.37 (± 1.97) m² ha⁻¹ by 1983 (Fig. 3). Thirty-one years following establishment, mean basal area had increased to 20.32 (± 2.71) m² ha⁻¹. Over the first 15 years of the stem exclusion stage (1968-1983), basal area accumulation averaged 0.72 (± 0.18) m² ha⁻¹ year⁻¹. Over the next 10 years (1983-1993), the rate of basal area accumulation had increased to 1.00 (± 0.08) m² ha⁻¹ year⁻¹.

In comparison, Schnur (1937) indicated that even-aged upland oak forests at age 30 years typically had basal areas of approximately 19.5 m² ha⁻¹, on average sites. The decline in mean density and concurrent increase in mean basal area are similar to patterns observed in other forest types in the stem exclusion stage of development (Oliver and Larson 1996, Johnson et al. 2002).

Conclusions

Early stand development patterns observed in the AC-9 provenance test plantation are comparable to those observed in natural northern red oak stands of the Central Hardwood Forest Region. Mortality rates after the first five years were initially higher than those of some naturally regenerated stands. However, the wide initial spacing may have delayed the onset of competition, and the monitored mortality rates are similar to those observed in natural stands, after the first five years. Diameter growth rates greatly exceeded those of northern red oak in natural stands. The lack of an existing overstory and removal of competition as part of plantation management accelerated dbh growth and hence stand development. The basal area after 30 years reflects the rapid diameter growth of the stand. Thus, our initial results suggest that the AC-9 plantation has followed the expected trajectory of stand development, with a relatively accelerated diameter growth rate. Given these outcomes, this plantation serves as a successful model of how it is possible to plant the desired species and restore the northern red oak forests of the region.

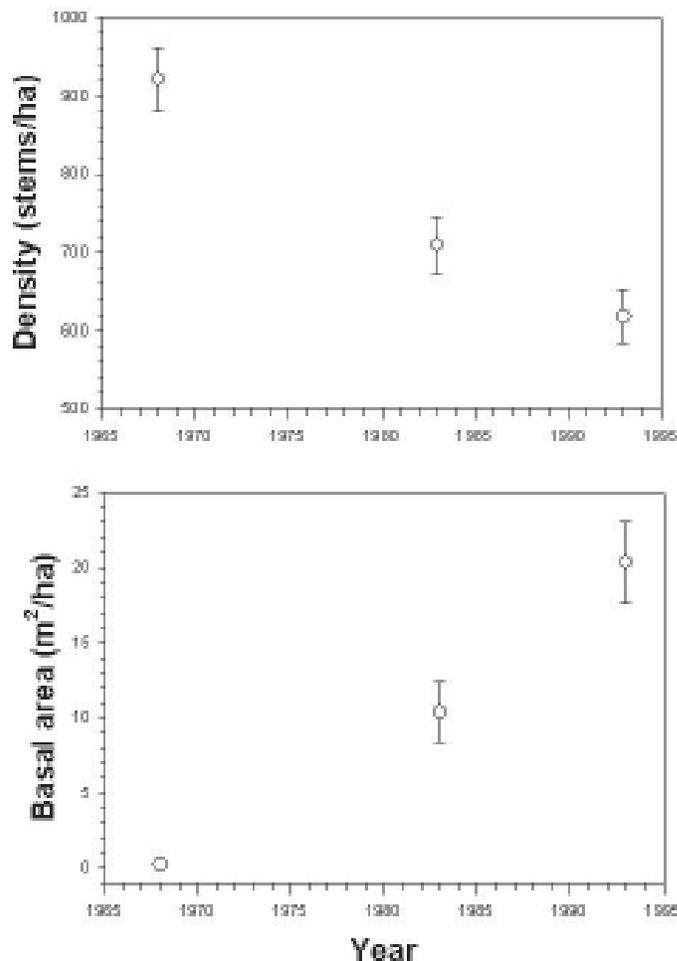


Figure 3.—Changes in mean (± 1 SD) stem density (stems ha⁻¹) and basal area (m² ha⁻¹) from 1968 to 1993 for the AC-9 northern red oak plantation, OARDC, Wooster, Ohio.

Acknowledgments

We wish to thank Marie Semko-Duncan for her help entering the original provenance test measurements. Additionally, we wish to thank those researchers at OARDC, including Dr. Howard Kriebel, who established and collected growth measurements at the AC-9 plantation since 1962. Helpful suggestions were obtained from two anonymous reviewers of the manuscript.

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DIMENSIONALITY FROM OBSCURITY: REVISITING HISTORICAL SOURCES OF BIG TREE SIZE

Don C. Bragg[†]

ABSTRACT.—Maximum tree dimensions are becoming increasingly sought-after information. However, scientific literature contains little on big trees, and champion tree registers often feature atypical individuals. Obscure historical sources can supplement estimates of maximum tree size. Many of these outlets are promotional and hence biased towards large trees. For example, a railroad company booklet contained a photograph of a white oak (*Quercus alba*) almost 7 feet in diameter, 125 feet tall, with a 100-foot wide crown. The *American Lumberman*, a trade journal from the early 20th Century, specialized in timber company narratives. Their articles are valuable for dimensioning trees because they often highlight “trophy” individuals like an oak from an Arkansas bottomland that scaled 10,000 board feet. Less commercially prominent taxa were sometimes mentioned, such as a persimmon (*Diospyros virginiana*) 108 inches in circumference and 120 feet tall from a pamphlet by Arkansas Commission of the Panama-Pacific International Exposition. Other potential sources include public land surveys, old journals, historical society memoirs (especially those before 1900), 19th Century state geology reports, and early soil surveys. Obscure historical resources benefit from their proximity to presettlement periods, but may also be affected by exaggeration, selection bias, or imprecise measurement and thus should be carefully evaluated.

Introduction

There are old-growth forests in eastern North America for which the trees are of very modest stature. For instance, the Cross Timbers region of Oklahoma and Texas is exemplified by post (*Quercus stellata*) and blackjack (*Quercus marilandica*) oak-dominated upland forests in which 200 to 500 yr old individuals rarely exceed 25 inches in diameter at breast height (DBH) and 50 feet tall (Therrell and Stahle 1998). Cliff faces and talus slopes also harbor very old (up to 1000 yr) and stunted northern white-cedars (*Thuja occidentalis*) (Kelly and others 1992). In addition to limiting tree size, harsh site conditions often restrict access and produce trees of poor form, thus greatly decreasing their economic value. This has helped protect some ancient forests from commercial exploitation, resulting in numerous examples of poor site old-growth surviving to modern times. Since many of these stands are largely untouched, they can act as suitable models for defining reference conditions. Unfortunately, virtually all accessible and productive eastern forests have experienced extensive logging and land clearing.

In recent decades, public land managers have invested considerable time and resources to the restoration of good sites to stands with old-growth-like attributes (for example, Vora 1994). To assist these efforts, a number of diagnostic old-growth features have been described (for example, Hunter 1989, Hunter and White 1997, Franklin and others 2002, Keddy and Drummond 1996, Rusterholz 1996). A fundamental characteristic of old-growth on good sites is an abundance of big trees (Martin 1992, Gaines and others 1997). However, the nature of tree dimensionality in old-growth leads to a number of critical questions. For example, if size is to be a defining factor in presettlement forest restoration, how big is big enough? Will dimensions gathered from modern forests correspond to those found in virgin landscapes? What are the best sources of maximum species dimensions, and how reliable are they?

A number of contemporary reports on big tree size are available. *American Forests* publishes a champion tree register listing native and naturalized tree species in the United States (American Forests 2003). In

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addition to this national register, other organizations like the Eastern Native Tree Society (ENTS) and many states have begun maintain their own champion lists. However, big tree registers often feature atypical (for example, multi-stemmed) individuals and are frequently incomplete. Furthermore, their scoring systems tend to favor open-grown individuals, which are architecturally different than trees from closed forests. With the possible exception of the ENTS database, there is also a degree of uncertainty in some of the dimensions of champion lists because of inadequate measurement techniques.

The scientific literature contains limited information on maximum tree dimensions. A number of journal publications provide data on big trees in some study areas (for example, Baldwin 1951, Bromley 1935, Jones 1997, Laughlin 1947, Lindsey and others 1961, Rood and Polzin 2003). May (1990) produced a list of big trees from the USDA Forest Service Forest Inventory and Analysis databases. May's forest survey sample is very extensive and inclusive, but since the FIA plots are systematically located, they will miss most of the biggest trees in the highly fragmented forests of the eastern United States. Other sources like silvics guides (for example, Folwells 1965) or dendrology textbooks (for example, Harlow and others 1979) report selected big trees, usually gathered from published national champion lists.

Historical sources from the 19th and early 20th Centuries can supplement dimensionality for many species. These resources tend to be obscure, often promotional, yet can be surprisingly informative. They also have the benefit of being from a period much closer to presettlement times. General Land Office (GLO) survey notes, promotional brochures, trade journal articles, and even postcards can provide at least qualitative information on the dimensions of big trees. This effort evaluates a number of examples of the forest giants mentioned in these outlets.

Methods

The information for this paper was developed in an effort to restore old-growth-like conditions to upland loblolly (*Pinus taeda*) and shortleaf (*P. echinata*) forests in southeastern Arkansas. Even though this study focused on pine old-growth, champion-sized hardwoods were frequently encountered while describing the reference conditions of this area (Bragg 2002, 2003). Arkansas GLO survey notes were transcribed from the compact disk archives they are stored in, entered into a spreadsheet, and then summarized to produce maximum tree diameters by surveyor tree identification. All other dimensional information was taken from captions, photographs, or articles.

Results

GLO Survey Notes

Table 1 lists the largest individuals from selected species used by GLO surveyors in the Ashley County, Arkansas area (Bragg 2003) and the national and state champion trees listed by American Forests (2003) and Arkansas Forestry Commission (2002), respectively. It should be noted that the GLO information in Table 1 came from a single county in Arkansas, and that some species (for example, *Betula nigra*, n = 11) had a very limited sample size.

Only a few species in the Ashley County GLO notes proved to be larger than the big tree lists. A pine (almost certainly loblolly) from the Ouachita River flatwoods had a larger diameter (72 inches) than either the current national champion loblolly pine (59.2 inches DBH) or the former state champion (58.6 inches DBH). The largest white oak (*Quercus alba*) (70 inches), pin oak (probably *Q. nigra* or *Q. phellos*) (78 inches), and baldcypress (*Taxodium distichum*) (144 inches) from the GLO records were bigger than the current Arkansas champions.

Other Government Publications

Concerned with a looming timber "famine" and severe environmental problems related to uncontrolled logging and land clearing, government agencies issued a number of advisory reports to Congress or the President in the latter parts of the 19th Century and the early 20th Century. Some of these volumes (for instance, Wilson 1902) contain historical photographs of impressively large trees. As an example, Ayres

Table 1.—Diameters of big trees extracted from the Ashley County GLO survey (Bragg 2003) and compared to current national and Arkansas champions.

Surveyor tree name	Probable species ^a	Ashley County GLO		American Forests (2003)	AFC big trees (2002) ^c
		n ^b	GLO max.	DBH (in.)	
Pine	<i>Pinus echinata</i> or <i>P. taeda</i>	2200	72	55.4 ^d -59.2	35.7-58.6
White oak	<i>Quercus alba</i>	1167	80	121.6	73.2
Sweet gum	<i>Liquidambar styraciflua</i>	872	70	88.5	n/a ^e
Pin oak	<i>Q. phellos</i> or <i>Q. nigra</i>	675	78	82.4-91.0	74.2-75.4
Overcup oak	<i>Q. lyrata</i>	588	54	82.1	n/a
Black gum	<i>Nyssa sylvatica</i>	408	40	73.8	64.3
Red oak	<i>Q. falcata</i> or <i>Q. pagoda</i>	344	60	99.3-108.9	87.9-95.8
Pecan	<i>Carya illinoensis</i>	206	40	85.0	77.3
Cypress	<i>Taxodium distichum</i>	173	144	205.0	143.6
Persimmon	<i>Diospyros virginiana</i>	127	24	28.0	46.2
Holly	<i>Ilex opaca</i>	65	16	39.8	36.9
Dogwood	<i>Cornus florida</i>	61	12	36.3	19.4
Sassafras	<i>Sassafras albidum</i>	58	30	83.4	73.8
Tupelo gum	<i>Nyssa aquatica</i>	18	72	107.0	n/a
Water birch	<i>Betula nigra</i>	11	36	66.2	47.4

^a Probable contemporary species name identified by the surveyor.

^b Number of tree records from Ashley County GLO of this species.

^c Unpublished list of state big trees maintained by the Arkansas Forestry Commission (AFC).

^d Where two species are probable, both diameters are provided in the order listed.

^e No current state champion.

and Ashe (1902) included a picture of men standing next to a chestnut (*Castanea dentata*) that may exceed 9 feet in diameter (fig. 1).

Other potentially valuable government reports may include “working plans” developed by professional foresters for landowners to help them learn how to sustainably manage their forests. These reports (for example, Foster 1912, Olmsted 1902, Reed 1905) focus on the commercial aspects of forestry, but often contain valuable pictures, graphs, and tables of virgin timber.

Promotional Literature

A booklet produced by the St. Louis, Iron Mountain, and Southern Railway (SLIMSR) touting their Arkansas lands contained a photograph (fig. 2) of a white oak that was 6.8 feet in diameter, 125 feet tall, with a 100-foot wide crown (SLIMSR 1892). This booklet is a classic example of self-promotion since the SLIMSR was interested in selling as much of their timberland as possible. However, there is no evidence to suggest that the size of this white oak had been exaggerated or otherwise falsified.



Figure 1.—Giant chestnut in the southern Appalachians, circa 1900 (picture from Ayres and Ashe (1902)).

Even though most historic photographs concentrate on commercial timber species, less prominent taxa are sometimes included. As an example, a pamphlet from the Arkansas Commission of the Panama-Pacific International Exposition contained an image of a persimmon (*Diospyros virginiana*) 108 inches

in circumference and 120 feet tall near Luxora, Arkansas (Hutchins 1915). While no crown width was reported in the caption, its other dimensions are easily as large as the current national champion (American Forests 2003).

Postcards were often used as a marketing tool. This compact medium of local features were inexpensively produced, and hence made popular advertisements. Two postcards from a recently published collection (Hanley 2000) showed massive oak logs cut from the Mississippi River bottomlands of eastern Arkansas. One included a caption describing a 12-foot log that scaled 2,160 board feet. Assuming the Doyle scale of sawtimber volume, it is possible to back-calculate a 58 inch small-end diameter for this log. While probably not typical of oak in the virgin forest, the soundness of these large sawlogs shows that it is possible to grow very large, quality hardwoods.

Early Journals

The *American Lumberman*, a trade journal from the early 20th Century, specialized in timber company narratives. *American Lumberman* articles usually included photographs of mill staff, products, equipment, and forestlands. Their descriptions are valuable in dimensioning large trees because many highlight “trophy” individuals. For instance, a single red oak (*Quercus falcata* or *Quercus pagoda*) from an Arkansas bottomland was estimated to scale 10,000 board feet (Doyle) (Anonymous 1909). Other brief notes on big hardwoods are sprinkled throughout this magazine, including an Indiana white oak log 88 feet long and nearly 4 feet in diameter at the butt, with very little taper (Anonymous 1903a) and an Ohio walnut (probably *Juglans nigra*) that yielded a 75-foot long clear bole 8 feet in diameter at the base (Anonymous 1903b).

Another helpful feature of *American Lumberman* photographs is that many were geographically referenced. According to its accompanying caption, the red oak from the preceding paragraph was located in section 31 of township 6 south, range 14 west, Grant County, Arkansas. Many stands are located even more precisely than this, sometimes down to a 40-acre parcel.

Other old periodicals can contain useful big tree information. *Garden and Forest*, one of the earliest forestry journals, would occasionally publish captioned photographs of large trees like the American beech (*Fagus grandifolia*) in Figure 3. An accompanying article (Sargent 1895) stated that this beech had a circumference of 9.5 feet at 6 feet above the ground and a crown spread over 130 feet wide. Assuming a conservative height of 80 feet, this tree had a “bigness index” of approximately 470 points, besting the current national champion by almost 10% (American Forests 2003). Langtree (1867) described very large hardwoods in the presettlement forests of Arkansas, including cottonwood (probably *Populus deltoides*) that reached 6 feet in diameter and bottomland oaks greater than 4 feet in diameter. Leech (1939) described an eastern white pine (*Pinus strobus*) cut from the virgin forests of Alger County, Michigan. This five-forked pine was 7.5 feet in diameter on the stump and yielded sixteen logs 16 feet long and a 12-foot long butt log.

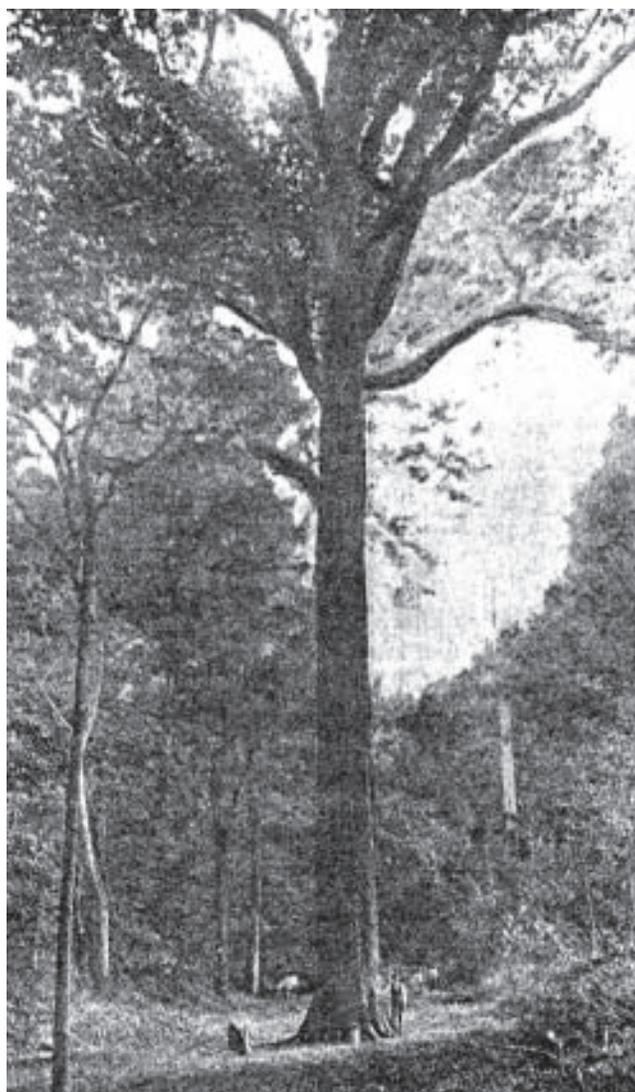


Figure 2.—A large white oak on the lands of the St. Louis, Iron Mountain, and Southern Railway (picture from SLIMSR (1892)).



Figure 3.—A massive American beech from South Hingham, Massachusetts. This individual scored a “bigness index” of 470 (picture from Sargent (1895)).

The JSTOR and *Making of America* online archives have also provided new venues for tree dimensionality. JSTOR’s botany and ecology collection of key scientific journals includes volumes that date back well into the 19th Century. These old journals often documented peoples’ travels and experiences in a format that would not be considered by most modern technical outlets. For example, a well-traveled former land surveyor named Jonathan T. Campbell imprecisely described an oak near Rockville, Indiana that “...was over six hundred years old...[and] between six and seven feet in diameter...” (Campbell 1885, p. 843). Campbell also reported very large sycamore (*Platanus occidentalis*) and a water elm (possibly *Planera aquatica* or some species of *Ulmus*) five feet in diameter along the Wabash River bottom. Another historical reference, adapted from an early silvics text

by D.J. Browne, mentions a white oak estimated to be “twenty-four to twenty-seven feet in circumference at the smallest part of the trunk.” (presumably, above the buttress) (Anonymous 1837, p. 345).

Discussion

Historical information sources are valuable additions to the data available on the maximum dimensions of tree species in eastern North America, but they must be viewed cautiously. One concern is that they may be more fiction than fact, and since they were described many decades ago, there is very little chance of ever validating a claim. However, exaggeration is not simply a historical phenomena, as it is not uncommon to find modern statements that cannot be supported. Second, the accuracy and reliability of the dimensions reported are unclear, as rarely was any mention made of how the measurements were taken. Even the observations made by professionals trained in some aspects of mensuration (like land surveyors) can be imprecise.

GLO notes can prove very useful in developing maximum tree diameter information. However, it is important to recognize some of the limitations of GLO notes (Bragg 2003). First, tree species identifications are sometimes vague, making the notes unavailable for a number of taxa. Second, witness and line trees were not selected for their size, but rather to facilitate the surveying process. Hence, there may be biases against very large trees or understory taxa that rarely reached sufficient size for scribing. Finally, diameter was estimated rather than measured using techniques inconsistent with current forestry standards.

Many historical outlets are promotional and biased toward large trees, but so long as exaggerated reports are avoided, this propensity is advantageous. A classic example of a fraudulent claim would be the rail car-sized produce commonly used in tongue-in-cheek postcards (even to this day). The use of “boosterism” arose from the desire of interested parties to sell and settle their lands, or to increase their prominence. Hence, when it came time to take expensive pictures, the photographer would often be led to particularly large (trophy) individuals. Fortunately, since people, horses, or other familiar objects were usually placed in the image as well, a de facto scale was provided to ensure claims of tree size are not too unbelievable (although even historical photographers were capable of doctoring images). The proclivity for showcasing the biggest of the trees, while detrimental for determining average or typical stand conditions, can help define maximum size.

Other potentially valuable but rarely used sources of big tree size include historical society memoirs (especially those dating to before 1900), early state geology reports, and possibly the first soil surveys. In reality, any reliable source that dates back a century or more could prove useful because of how much closer in time they were to virgin conditions. It is also likely that, almost without exception, these giants still represent underestimates of maximum tree size.

Acknowledgments

I would like to recognize the support of O.H. "Doogie" Darling and Jim Guldin. Without their assistance, many of these historical references would have gone uncovered. Mike Shelton and two anonymous reviewers helped improve the quality of this paper.

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LONG-TERM GROWTH AND CLIMATE RESPONSE OF SHORTLEAF PINE AT THE MISSOURI OZARK FOREST ECOSYSTEM PROJECT

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ABSTRACT.—Shortleaf pine (*Pinus echinata*) is one of the most of important conifers in the Central Hardwood region both ecologically and economically. In the Ozark Highlands of Missouri and Arkansas, the presence of shortleaf pine provides an important conifer component in otherwise hardwood dominated forest types. Regional forest health issues such as oak decline and red oak borer as well as pine restoration efforts have increased interest in the species and the management of oak-pine forest types. The purpose of this paper is to provide information about the growth of shortleaf pine using one of the longest existing growth records available. Using tree-ring data collected at the Missouri Ozark Forest Ecosystem Project (MOFEP), we provide a long-term perspective of shortleaf pine growth, climate response, and regeneration. Tree-ring data from shortleaf pine remnants and cores of live trees illustrate the temporal variability in growth of shortleaf pine over the past 386 years. We developed a climate response function based on instrumental climate data from a 70 year period (1931-2001). Variation in shortleaf pine ring-width was most strongly correlated with growing year Palmer Drought Severity Index (PDSI) months, extreme minimum winter temperatures, and previous fall season PDSI ($r^2 = 0.43$, $p < 0.0001$). Further analysis of the relationship between tree-ring indices and PDSI suggest that the correlation was higher in the early half of the 20th century than the latter half. Late winter (January and February) may be becoming increasingly important in determining shortleaf pine growth. Spectral estimates of the tree-ring chronology suggest that shortleaf pine growth has a near 21 year periodic variation that is likely an influence of larger scale climatic cycles. Comparisons between growth of shortleaf pine in three age classes both pre- and post 1880 show that the species' growth rate was significantly greater during the 20th century than for the two prior centuries. Management implications for the long-term variations in shortleaf pine growth are discussed.

Shortleaf pine (*Pinus echinata*) is a valuable component of the Central Hardwoods region for notable reasons. Dendrochronologically, the species' high resistance to decay (Harborne 1994,) and successful regeneration on drought prone aspects (Parker 1982, Stambaugh 2001) makes it one of the best sources for studying the relationship between tree growth and climate in the Central Hardwoods region. Ecologically, the species provides diversity in a hardwood dominated forest region; in part because it is a conifer, its successful competition and occupation on xeric landscape positions (Guldin 1986, Murphy and Nowacki 1997, Stambaugh 2001) and its regenerative success during frequent disturbance (Keeley and Zedler 1998, Guyette and Spetich 2003). Economically, the species played a key role in the initial development of lumber industries in the Ozark Highlands of Missouri and Arkansas and remains an important commercial species today.

During the last century, many large-scale land management projects have been undertaken with objectives of promoting shortleaf pine success (USDA 2003, Buekenhofer and others 1994, Masters and others 1996, Rimer 2003). Interest in the management of shortleaf pine has increased because of the dramatic decrease in the acreage of the species since Euro-American settlement (Liming 1946, Hamilton 2003), including the decrease of pine savanna communities. Additionally, Ozark region forest health issues of red oak borer and oak decline have underlined the importance of maintaining a conifer forest component. In this paper we reconstruct the growth of shortleaf pine using one of the longest existing growth records available. Using tree-ring data collected at the Missouri Ozark Forest Ecosystem Project (MOFEP), we provide a long-term perspective of shortleaf pine growth, climate

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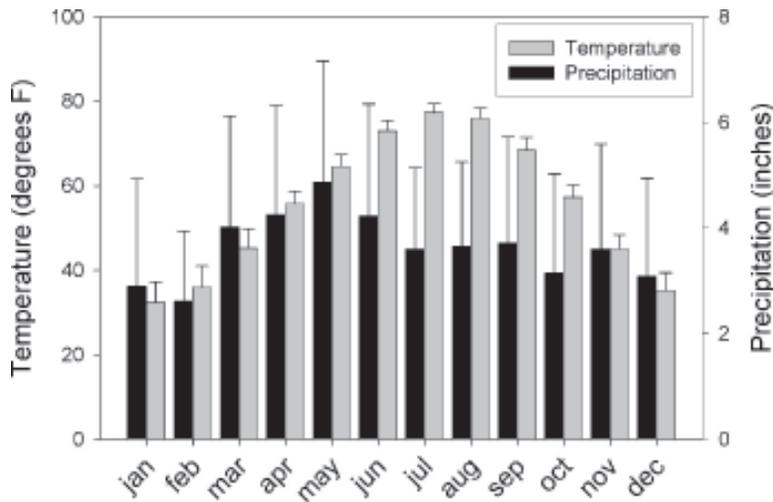


Figure 1.—Monthly averaged surface data for Missouri Climate Division 5 (NCDC 1994) for the period 1895 to 2002 with error bars equal to one standard deviation.

response, and regeneration. The information here provides managers a standard in long-term growth against which to compare current growth and measure future growth variability.

Study Areas

Shortleaf pine samples were collected from the southeastern portion of Missouri within site boundaries of the Missouri Ozark Forest Ecosystem Project (MOFEP) (approx. 91°4'9"W, 37°9'43" N). MOFEP is a landscape experiment comparing the effects of even-aged management, uneven-aged management, and no harvesting on multiple ecosystem components (Brookshire and others 1997a). Nine sites, each a minimum of 600 contiguous acres, comprise MOFEP. Sites are nearly entirely forested with approximately 9 percent of the total tree species importance values in shortleaf pine (Brookshire and others 1997b). Although shortleaf pine occurs on all MOFEP sites, its abundance is highly variable and attributed to competition with hardwoods, forest history, and topographic and soil properties (e.g., slope, aspect, parent material). Throughout the MOFEP study sites and the greater Ozark Highlands region ample remnant shortleaf pine wood (e.g., standing dead snags, stumps preserved from lumbering era (Hill 1949, Cunningham and Hauser 1989)) provides excellent opportunity for dendrochronological research and construction of long time series (Guyette and Dey 1997).

The climate and weather of the Ozark Highlands region is determined primarily by the mid-continent location in the middle latitudes and has been classified as humid-continental and humid subtropical (Rafferty 1980). The climatic patterns of the Ozark Highlands are consistent with those of the Central Plains, which infers that general circulation features dominate the local climate (Su 1997). Su (1997) found that the temperature and climate of the Ozark Highlands is significantly related to sea surface temperatures particularly of the north Pacific and tropical Pacific Ocean currents. In the region of MOFEP, mean monthly precipitation, primarily in the form of rain, is 3.64 inches (9.25 cm) (fig. 1).

Methods

During the years 1996 to 2002 over 200 shortleaf pine cross-sections were cut from remnant stumps and dead standing snags from all MOFEP sites. Over 150 increment cores from live trees were collected during the years 2002 and 2003 from MOFEP sites 1 through 7. Samples were collected non-randomly because of their limited quantity and tree-ring quality. Live trees were selected for sampling if 1) their age was likely greater than 50 years, 2) they had a live crown ratio greater than 30 percent and no signs of dieback, and 3) they occupied dominant and co-dominant canopy positions. Growth rate of the tree and the aspect and slope of the tree's geographic position were not considered in selection. Cross sections of remnants were collected from 15 to 100 cm above the ground and increment cores were collected from breast height (1.37 m). Cross sections and cores were prepared by sanding their surfaces with increasingly finer sandpaper down to 600 grit. All annual rings were measured to 0.01 millimeter precision. Tree-ring measurements of remnants were cross-dated using live

tree cores (i.e., known tree-ring dates) and a previously constructed shortleaf pine tree-ring chronology from Shannon County, Missouri (Guyette 1992). Cross-dating is a method whereby individual tree-rings are absolutely dated to calendar years by matching variation in ring-width patterns to those of known dates (Stokes and Smiley 1968, Baillie 1995). Crossdating identifies errors in measurement caused by missing or false rings. We used COFECHA software (Holmes and others 1986) for quality control of ring measurements and crossdating through inter-series correlations and t-tests.

Long-term growth

Shortleaf pine chronologies were constructed for the purpose of quantifying the long-term growth of the species at MOFEP. Shortleaf pine remnants (n=70) and cores (n=38) used to construct the MOFEP shortleaf pine tree-ring chronology were chosen based on circuit uniformity, the lack of missing rings, and crossdating. We constructed an ARSTAN tree-ring chronology utilizing Turbo ARSTAN software (Cook 2002). ARSTAN derives a standardized chronology using a robust mean value function to reduce the effect of statistical outliers. Standardized chronologies are pre-whitened by reincorporating the derived pooled auto-regressive model to form the auto-regressive standardized (i.e., ARSTAN) chronology. ARSTAN chronologies are described in detail in Cook (1985) and Cook and Holmes (1984).

Tree-ring series were interactively detrended in two steps (Holmes and others 1986). First, a negative-exponential curve or linear regression line was fit to individual series to remove age related growth trend. Second, we interactively chose flexible spline curves (i.e., 20-50 yr length) to remove higher frequency variation in growth due to stand disturbances (e.g., mid-1940's logging event). All tree-ring series from remnant wood (i.e., outside ring date prior to 1880) were fit with 50-year spline curves as we had no knowledge of dates of prior stand disturbances. The stationary tree-ring chronology was analyzed for the purpose of identifying common high frequency (e.g., < 30 yrs) components or climate cycles influencing ring-width variation. Using SAS software, we tested for significant white noise components in the time series by Fisher's Kappa and Bartlett's Kolmogorov-Smirnov test statistics. SAS software uses a Fast Fourier Transform to generate periodograms. A centrally weighted Daniell filter (bandwidth = 5) was used to smooth the periodogram. It was taken into consideration that excessive smoothing can obscure the important spectral detail and insufficient smoothing leaves erratic unimportant detail in the spectrum. The power spectral density was estimated for the purpose of defining which periodic components likely contribute the most variation in the frequency of ring width (i.e., tree growth). The greater the power spectral density the more variation is accounted for by that period or climate (e.g., drought) cycle.

We analyzed the differences in growth rates of shortleaf pines that grew at MOFEP over approximately the past 386 years. Only cores and cross sections of remnants that had an estimated 15 or fewer rings missing (number of missing rings was estimated using methods of Duncan 1989) to the pith were used for the analysis. Near exact pith dates were necessary because we wished to separate and compare the growth rates of shortleaf pine both pre- and post-1880. Pith dates likely incorporate some error due to differences in sampling heights between cross sections and cores. We chose to compare the growth rate of shortleaf pine post- and pre-1880 because of the time period's association with increases in lumbering (Cunningham and Hauser 1989), decrease in fire frequency (Guyette and others 2002), and increases in human population (Stevens 1991) and agricultural land use (Jacobson and Primm 1997). From radial measurements of tree-rings we calculated tree diameter at 10, 30, and 50 years of age. Radial tree-ring measurements were converted to tree diameter by multiplying by two. We used t-tests to test for differences in the growth of shortleaf pine pre- and post-1880.

Climate response

Increment cores from live trees, ranging from 50 to over 200 years old, were used to analyze the climate response of shortleaf pine. Considering the land use history of MOFEP and larger Ozark region (Rafferty 1980, Stevens 1991), many trees likely developed following logging disturbance, grew under the influence of specific stand developmental processes (i.e., stem initiation, stem exclusion (Oliver and Larson 1996), and currently occur in closed canopy conditions. For the climate response analysis we chose sections of cores with the least amount of growth suppression and anomalous wood formation. For the purpose of maximizing the climate signal of the chronology, we chose the most highly inter-

correlated trees (n=38) growing during the period of divisional instrumental climate record (1895-2002). This resulted in a minimum of 28 trees represented in the indexed growth value for any year for the period 1895-2002.

Divisional climate data (1895-2002) from the National Climate Data Center (NCDC 1994), which included monthly precipitation, monthly temperature, and monthly Palmer Drought Severity Index (PDSI) values, were used for the climate response analysis. We also used individual station data (Salem, MO, 1931-2001) for the purpose of accounting for more localized weather events such as extreme minimum winter temperature. SAS (SAS/STAT, 2002) and DendroClim2002 software (Biondi 2002) were used to correlate climate data and tree-ring chronology indices and analyze and choose the response variables. Although climate response functions typically employ principal components regression, we used stepwise regression so to keep monthly variables distinctively defined. The stepwise regression may cause an inflation of the correlation coefficients when multi-colinearity exists. In model development, we chose climate variables (and their combinations), based on their biological relevance to shortleaf pine tree-growth. A multiple regression equation defined the response function with all variables significant ($\alpha = 0.1$).

Correlations were calculated between the ARSTAN chronology and monthly precipitation, temperature, and PDSI data for August of the year prior to growth through December of the year of growth. We calculated the mean correlation between the ARSTAN chronology and PDSI using moving intervals of 36 years on one year time steps (e.g., time step 1: 1895-1929; time step 2: 1896-1930; etc.) The length of moving interval (36 years) was chosen so to maximize the number of time steps compared, include an adequate number of years for correlation analysis, and because it is the minimal length allowed by DendroClim2002 software. Slight differences in correlation results likely occur as the length of the moving interval is adjusted by a few years, however too short of intervals may be unrelated to larger scale climate variability and too long of intervals decreases the number of comparisons between intervals.

Results

Long-term growth

We constructed a 386-year tree-ring chronology to demonstrate the long-term year to year changes in growth of shortleaf pine (fig. 2). For the 20th century some of the notable years and periods of relatively low growth were 1901, 1912-1913, 1930, 1936, 1952-1954, 1960, 1978, 1980, 1984; a total of 12 years. In retrospect, approximately 11 years of comparable low growth (ring-width index < -0.20) occurred in each of the 18th and 19th centuries. Calendar year 1936 had the lowest ring-width index value during the 20th century and from 1700 to 2001 only four years had a less or equal ring-width index value. Prior to 1700 the chronology is poorly replicated and changes in growth should be viewed with caution.

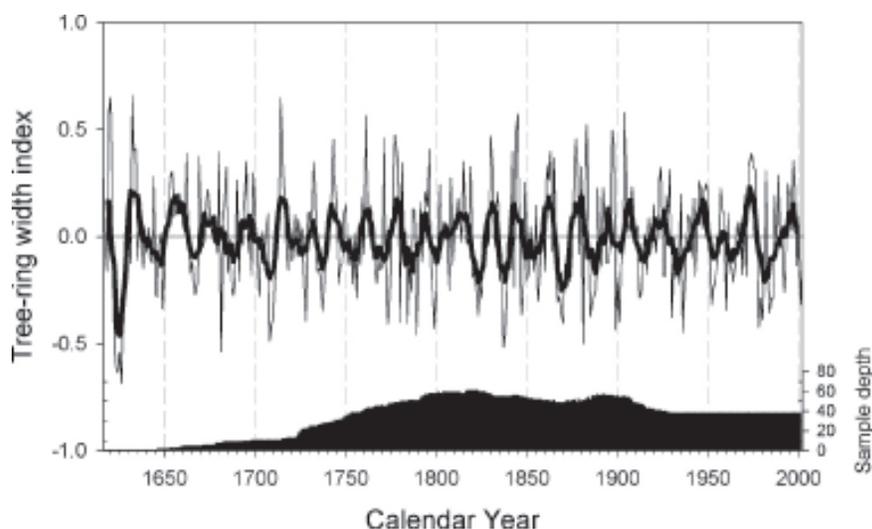


Figure 2.—A 386-year shortleaf pine tree ring chronology from the Missouri Ozark Forest Ecosystem Project (MOFEP) with the sample depth at calendar years plotted on the right y-axis. A 7-year moving average (bold line) demonstrates the lower frequency variation in the tree-ring indices. Variation in tree-ring indices were most highly correlated to mean summer (May through August) Palmer Drought Severity Indices (PDSI)(NCDC 1994).

We arbitrarily chose and fit a 7-year moving average to the tree-ring width index in order to enhance the visibility of the lower frequency variation in shortleaf pine growth (fig. 2). A spectral analysis of the time series revealed peaks in variance at periods of 3.5, 9, and 21 years (fig. 3). The hypothesis that the

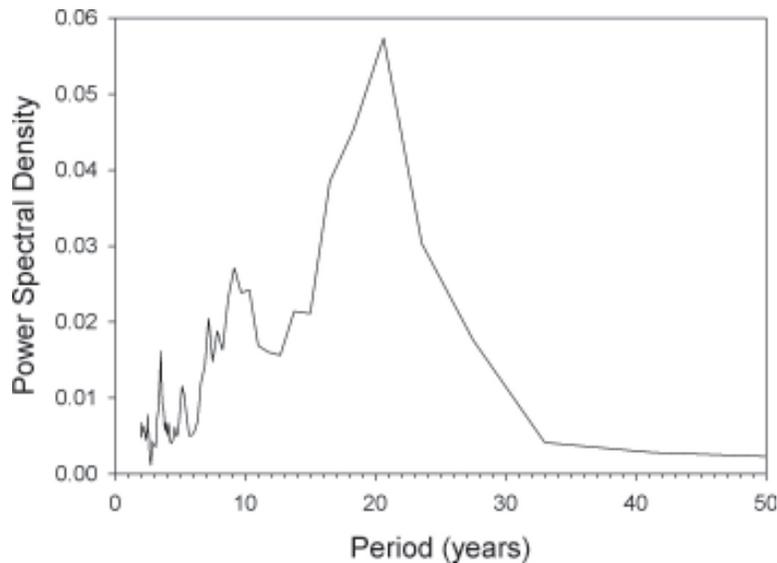


Figure 3.—A plot of the power spectral density of the 386-year tree-ring chronology with notable peaks in variance at periods of 3.5, 9, and 21 years. The tree-ring index time series was significantly different from white noise.

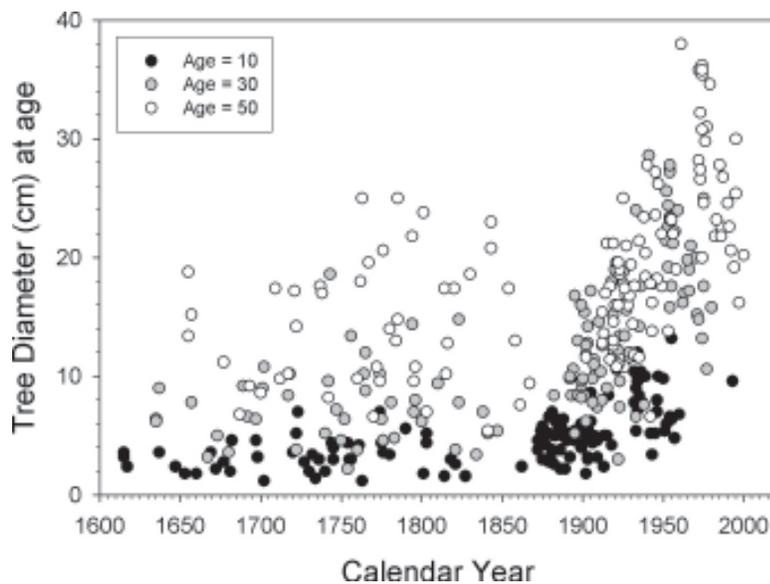


Figure 4.—A scatter plot of shortleaf tree diameter by calendar year. The plot shows the diameter and year of individual trees at ages 10, 30, and 50 years. On average, the growth at age of shortleaf pine trees currently growing at MOFEP have been unsurpassed in comparison to trees growing prior to 1880.

21-year peak was not significantly different from the white-noise spectrum was rejected (Fisher's Kappa = 10.46, $p < 0.01$; Bartlett's Kolmogorov-Smirnov test statistic = 0.29, $p < 0.0001$).

We quantified tree diameter at ages 10, 30, and 50 years for all tree-ring series with estimated pith dates. Series were divided into two periods; pre- and post-1880. T-tests showed that the growth of trees between the two periods is significantly different for all tree age classes (table 1). Trees post-1880 grew faster than those grown pre-1880 (fig. 4).

Climate response

We used a highly replicated tree-ring chronology that spanned the instrumental climate record of 1895–2002 to analyze climate response variables. The highest correlations between the 107-year tree-ring index and a single monthly climate variable was for July PDSI ($r = 0.51$, $p < 0.0001$). All individual growing season (May through July) PDSI variables were significantly correlated ($r = 0.43$ to 0.51 , $p < 0.0001$) with the tree-ring index. Mean PDSI variables for a growth year (January through July) and

Table 1.—Important changes in forest history occurred circa 1880 for the study site region (see text). A comparison of shortleaf pine mean tree diameters at age between the pre- and post-1880 periods. We hypothesized that there is no significant difference in growth at ages 10, 30, and 50 years between the two periods. The null hypothesis was rejected for each of the age comparisons.

Tree age	Period	n	Mean diameter (cm)	Variance	t-stat	p
10	Pre-1880	72	3.77	2.37	-7.34	<0.0001
	Post-1880	57	6.75	7.49		
30	Pre-1880	72	8.79	13.12	-9.04	<0.0001
	Post-1880	56	16.97	35.63		
50	Pre-1880	72	14.96	23.7	-6.95	<0.0001
	Post-1880	56	23.15	59.42		

mean annual PDSI values also yielded significant correlations ($p < 0.0001$). Although several individual months of precipitation and temperature (e.g., June precipitation, $r = 0.40$, $p < 0.0001$; July temperature, $r = -0.44$, $p < 0.0001$) were significantly correlated to the tree-ring index, combinations of these variables (e.g., PDSI) can better represent the climatological influence on growth. Extreme minimum winter temperature (February and March, 1931-2001) was also significantly related to the ring-width index.

The independent variables predicting the tree-ring index were growth year PDSI values, extreme minimum winter temperature, and previous year fall PDSI. Because of the inclusion of extreme minimum winter temperature, our response function was developed from data taken during the period 1931-2001. The response function was defined as:

$$\text{TRI} = 0.92 + 0.07 * (\text{PDSI}_G) + 0.01 * (\text{ExT}_W) - 0.02 * (\text{PDSI}_F)$$

where: TRI = tree-ring chronology index

PDSI_G = mean of growth year PDSI values (January through July),

ExT_W = extreme minimum temperature (February and March), and

PDSI_F = mean PDSI values of previous year fall season (October through December)

(model $r^2 = 0.43$, $p < 0.0001$)

Correlations between monthly PDSI and the tree-ring index using 36-year moving intervals were also greatest for June PDSI, particularly during the periods with moving interval end dates between 1954 (i.e., 1918-1954) and 1967 (i.e., 1931-1967) (fig. 5). The individual months May, June, and July PDSI were generally highly correlated to tree-ring indices for all 36-year intervals, except those intervals with end dates after 1987 (i.e., 1951-1987). Correlations between monthly PDSI and the tree-ring index for months prior to the growing season (i.e., prior to May) were less correlated with PDSI than growing season summer months (May through August). Pre-growing season months (January to April) had the greatest portion of significant correlations around the period of approximately 1909 to 1970, suggesting that, at least during some years of growth, conditions prior to the period of cambial expansion are important for current year growth. This result, coupled with a potential trend towards higher correlations between ring-width indices during months prior to the growing season (e.g., January and February) and the lack of correlation late in the growing season (e.g., July and August) may point to changes in climate. Six of the last eight moving intervals (i.e., 1994-2002) had significant correlations during growing season January. No significant correlations existed between PDSI of growing season months and the tree-ring index after the interval with end date 1985 (i.e., 1949-1985).

Discussion

In the last few decades advances in tree-ring research, its methodology, and computer technology have contributed significantly to our understanding of the relationship between tree-growth, climate, and anthropogenic and natural disturbances in the Midwestern U.S. (Meko and others 1985, Cleaveland and Stahle 1989, Cook and Cole 1991, Cleaveland 1995, LeBlanc 1998, Woodhouse and Overpeck 1998, Guyette and others 2002, Hughes 2002, Fye and others 2003). The 386-year tree-ring chronology described here provides important information about long-term variability in shortleaf pine tree growth and the influence of climate at one of the largest landscape scale ecosystem projects in the country. MOFEP, an ecosystem project with an expected 100-year duration, can not only utilize the information here to gain a historical perspective of the growth of their current shortleaf pine forests, but also to apply the long-term variations recorded by tree-rings to the long-term variation in other ecosystem components. Relative to other MOFEP ecosystem components, trees are likely the best resource for quantifying and identifying long-term temporal climate and other environmental changes because trees are relatively long-lived, stationary in geographic location, relatively abundant, contain inter- and intra-annual growth information, and extensively studied thus comparable to many other locations.

Our results suggest that shortleaf pine growth at MOFEP is influenced by continental scale atmospheric general circulation patterns as suggested by Su (1997). The 21-year drought cycle is very similar in length to the 22-year drought cycle that has been found to influence tree-growth across the other parts of western (Mitchell and others 1979, Ammons and others 1983) and Midwestern United States (Meko and others 1985). In more detailed studies (Mitchell and others 1979, Stockton and others 1983) attributed the near 22 yr period (i.e., drought cycle) to the Hale solar cycle and commented about its somewhat imprecise rhythm. Guyette (1981) also found shorter periodicities represented in variance of tree-rings of white oak (*Quercus alba*) and eastern redcedar (*Juniperus virginiana*) growing in the Ozark region. In a qualitative sense, a near 22-year drought cycle fits many of the notable droughts of Missouri during the 20th century from 1911, 1912 to the 1930's dust bowl droughts to 1952-1954 to 1978 to 2000-2002. Most of these years had tree-ring index values < -0.20 (fig. 2). If the 21-year cycle continues as it has over the last 300 or more years, shortleaf pine indexed growth at MOFEP is expected to be entering a negative phase (fig. 2).

The correlation between the tree-ring index and individual monthly PDSI values using moving interval techniques showed that the strength of the correlation over the past century has important characteristics for understanding tree-climate response. One explanation for the high correlation between the tree-ring index and PDSI during the early half of the 20th century is that this was a period of abundant drought years, thus a growth-limiting factor common among all trees. In comparison, the latter half of the twentieth century has few intervals with significant correlation to PDSI which, at least in part, explains the low variance explained in the response function. During relatively wet periods, as was the case throughout much of the 1960s and through the 1990s, growth is not limited by moisture therefore, tree-rings, particularly shortleaf pine on xeric aspects, can be poor predictors of climate conditions.

An arguable explanation for a change in correlation between the tree-ring index and PDSI over the last century is that climate in the region is changing. Cleaveland and Stahle (1996), in a tree-ring reconstruction of Ozark hydrologic drought using oaks (*Quercus stellata*, *Quercus alba*), eastern redcedar (*Juniperus virginiana*), and baldcypress (*Taxodium distichum*), identified a marked decrease in Palmer Hydrologic Drought Index (PHDI) variance occurring in the mid 20th century. A notable feature of the correlation between the tree-ring index and PDSI is the increase in the number of significantly correlated intervals for months prior to the onset of growth (fig. 5). Our climate response function supports that PDSI and extreme minimum temperatures prior to the period of wood formation have important influence on growth occurring primarily May through July. Minimum monthly winter temperature data from the Ozark region shows a trend of continual small increases in mean minimum winter temperature. Increases in winter temperature could benefit the growth of shortleaf pine, particularly in Missouri where the species' range and growth potential may be limited by temperature

(Fletcher and McDermott 1957, Lawson 1990). Assuming minimum winter temperatures will continue to increase and that the growth of shortleaf pine will positively respond to further increases, management to promote the species would be of increased value. Furthermore, increased growth rates by shortleaf pine would likely increase the species' ability to compete with hardwoods. This is particularly important under conditions of small-scale forest disturbances (e.g., canopy gaps), where shortleaf pine success is very low (Stambaugh 2001, Stambaugh and others 2002).

Despite the limitations on growth by climatic factors, our analysis shows that shortleaf pines at MOFEP are growing faster than they have over the past 300 years. Significant land use changes circa 1880 were used to separate our two sets of growth data. Explanations for the result that trees prior to 1880 grew significantly slower than post-1880 are numerous. Likely the most important event was lumbering and its effect of releasing advanced shortleaf pine regeneration, reducing competition, and increasing available light. Despite the effects of lumbering on pine growth, the difference in fire disturbance regimes between the two periods have likely also been important. Because the effects of fire on tree-growth were likely different during each stage of the regional anthropogenic fire regime (Guyette and others 2002) it is difficult to measure the influence of repeated fire disturbances. Future studies of the effects of overstory removal and fire disturbances on pine growth would not only enhance our understanding for shortleaf pine management but also aid in interpreting historic changes in tree growth.

Acknowledgments

We thank two anonymous reviewers for comments on earlier versions of the manuscript. This research was made possible through support from the Missouri Department of Conservation and Missouri Ozark Forest Ecosystem Project.

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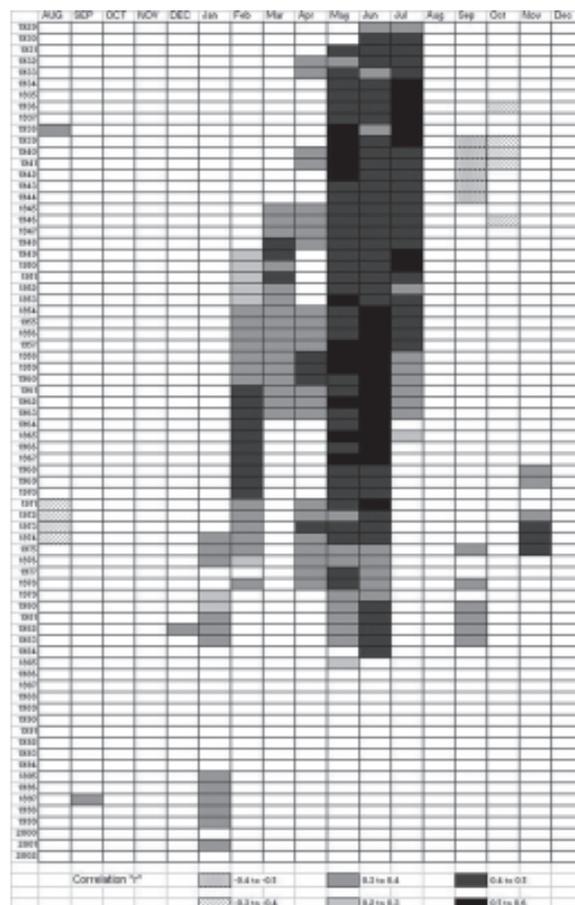


Figure 5.—Correlation matrix of ring-width index and monthly PDSI values. Months in capital letters are prior to the year of growth and smaller case months are during the year of growth. Each box represents a 36-year moving interval that ends on calendar year represented on the y-axis (e.g., 1950 represents the 36-year interval 1914 to 1950). Box shade represents the correlation coefficient between the ring-width index and PDSI values. Empty (white) boxes are intervals of no significant correlation ($p \geq 0.010$). See legend for correlation value ranges.

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A GENERAL ECOLOGICAL MODEL FOR INTERPRETING FOREST LANDSCAPE PATTERNS AND PROCESSES IN THE CENTRAL HARDWOOD FOREST REGION

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Based on previous research on forest communities in Illinois, Kentucky and Tennessee, a general ecological model has been developed for interpreting forest community patterns and processes across the landscape. Past ecological and physiological research that has shown three major environmental resource factors control the primary response of most terrestrial (upland and wetland) woody plant species. These resources are 1) light or lack of it, 2) soil water (too little, too much, optimum) and 3) soil oxygen or a lack of it. The assumption here is that the forest plant is within its normal geographic range and is free to interact with its normal associates. A fourth factor, disturbance, has a major effect on species distribution through damage to the photosynthetic component and forest structure which effectively rearranges the distribution of the resources or changes their amount or level of intensity. The general ecological model for the central hardwood forest region is a construct that combines species distribution (normal or Gaussian) curves (continuous) with a soil moisture/oxygen gradient from xeric to mesic to hydric. The model integrates the ecological/physiological characteristics responsible for species response to variations in environmental resources. Disturbance, or lack of it, may be responsible for major changes in the landscape patterns, and the landscape model has been modified to reflect this effect. The submodel integrate and show the consistency in patterns of species characteristics such as functional and realized niches, shade tolerance, stem size, growth rates, r and k reproductive strategies as well as a third strategy type, seed size, and type of seed dispersal. Stand composition, basal area and diversity also show consistent patterns. The effect of disturbance on stand and species characteristics (e.g., species richness) changes with location along the gradient.

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SIX-YEAR GROWTH RESPONSE OF GREEN ASH TO RELEASE AND FERTILIZATION TREATMENTS

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One major obstacle in the forestry profession is public resistance to traditional silvicultural practices. Non-industrial private forests are expected to be the major source of hardwoods in the future. Because of the increased demands on these forests, it is important to provide landowners with an attractive alternative for managing to retain forest productivity. Crop tree enhancement promotes stand growth and quality while also providing an opportunity to maintain wildlife habitat and aesthetics of hardwood forests. This study examines a bottomland hardwood site at Ames Plantation near Grand Junction, Tennessee. The site was clearcut in 1980, and a significant component of green ash emerged. Other species on the site include black willow, elm, cottonwood, box elder, river birch, cherrybark oak, yellow-poplar, red maple, and sweetgum. Green ash was designated as the crop tree species with the greatest potential value in the future. The goal of crop tree enhancement at this site is to examine which treatments will retain green ash as a significant component of the stand. Crop tree enhancement treatments included release, release with fertilization, and control, and were replicated 5 times. In 1996, 25 crop trees in each treatment plot were selected and initial measurements of total height, height to green crown, dbh, and crown spread in 2 directions were obtained. All of the trees included in a 10-factor prism sweep around each crop tree were designated as competitors. Total height, height to green crown, dbh, crown projection, azimuth, and distance from the crop tree was recorded for each competitor tree. The site was remeasured during the winter of 2002-03. Preliminary results indicate differences among crop tree enhancement treatments in height and diameter growth in the first 6 years after treatment. The differences will be examined with respect to several distance-independent and distance-dependent expressions of competition, live crown ratio, crown width, and soil factors.

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WHITE-TAILED DEER HERBIVORY ON FOREST REGENERATION FOLLOWING FIRE AND THINNING TREATMENTS IN SOUTHERN OHIO MIXED OAK FORESTS

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ABSTRACT—The effects of white-tailed deer browsing on species richness, density and height of forest regeneration were examined on three oak dominated forests in southern Ohio. Each study site consisted of four - 20 ha silvicultural treatments, resulting in 12 experimental units. Mean estimated post harvest deer densities were 6 deer km⁻². Three pairs of plots were established within each treatment at each location, and a 2.4 m tall deer exclosure fence was installed around one plot from each pair in the summer 2001. Species and diameter (cm) of all overstory and midstory vegetation were recorded. All woody stems between 10 and 150 cm tall were identified, and height (cm), basal diameter (mm) and evidence of deer browse were recorded in 2001, 2002 and 2003 on sub-plots within each plot. Overall mean height of regeneration was 16.1 percent lower on un-fenced plots, but species richness and seedling density did not differ between fencing treatments. Blackgum, which had 40 percent lower mean heights on unfenced treatments, was the only species tested that was significantly affected by fencing treatment. Silvicultural treatment had no significant effect on overall mean seedling height; however, treatment effects were significant for chestnut oak, red maple and greenbrier.

Oak (*Quercus* spp. L.) dominated forests occupy approximately 46 percent of the forestland in the eastern United States (Mc Williams et al. 2002). In Ohio, oak dominates 59 percent of the forestland, and this percentage is even greater in the southeastern portion of the state (77 percent; Griffith et al. 1993). Most of the mature second-growth forest in southern Ohio established following severe disturbance from past agricultural or industrial usage (Hutchinson et al. 2003). Iron furnaces, fueled by charcoal that was produced from repeated cutting of forests, have had a lasting effect of the forest cover of southeastern Ohio (Stout 1933; Hutchinson et al. 2003). Also, during establishment these forests were exposed to repeated fire, with fire return intervals ranging from 5 to 15 years (Sutherland et al. 2003). These frequent fires continued in southern Ohio until organized fire suppression was established in 1923 (Sutherland 1997). At the time these forests became established (mid 1800's to early 1900's) white-tailed deer (*Odocoileus virginianus* Zimm.) populations in southern Ohio were likely to have been in decline (Iverson and Iverson 1999). In fact, from 1904 to 1923 white-tailed deer were absent from Ohio (ODNR 2003).

Today, advanced regeneration in these oak dominated forests ranges from high percentages of red maple (*Acer rubrum* L.) and other shade tolerant species on sites with minimal canopy disturbance to regeneration dominated by pioneer species like yellow-poplar (*Liriodendron tulipifera* L.) on sites with considerable canopy disturbance (Lorimer 1993). In fact, numerous studies have reported a distinct lack of oak regeneration throughout the oak-hickory region on all but the driest of sites (Clark 1993). Many studies including McCarthy et al. (1987), and Goebel and Hix (1997) have found that advanced oak regeneration is probably inadequate to successfully maintain a significant proportion of oak in these forests following canopy disturbance. Reasons most commonly cited for this lack of oak regeneration include: 1) cutting practices which remove individual trees and often do not supply adequate canopy disturbance to provide sufficient light (Burns and Honkala 1990), and 2) the lack of repeated disturbance from fire since oaks are typically less susceptible to injury and are able to repeatedly produce sprouts (Huddle and Pallardy 1999, Dey 2002, Brose and VanLear 1998).

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In a review of the literature on the effects of white-tailed deer on plants and plant communities, Russell et al. (2001) cite numerous studies which have been conducted to elucidate the effects of high densities of white-tailed deer on forest regeneration in northern hardwood forests. These studies indicate that deer can significantly influence the morphology and growth rates of tree seedlings (Jacobs 1969, Tierson et al. 1966). Effects of deer on survival are less obvious, but on sites with high densities (> 8.5 deer km^{-2}) deer have prevented the recruitment of tree seedlings into the sapling and larger size classes (Russell et al. 2001, Healy 1997, Trumbull et al. 1989), and since deer feed selectively, species diversity can also be significantly affected (Harlow and Downing 1970, Tilghman 1989). Because white-tailed deer browsing has been linked with much of the failure of forest regeneration in Pennsylvania and other northern states (Marquis and Brenneman 1981), it is often assumed to be a contributing factor to oak regeneration problems in the oak-hickory region (Lorimer 1993, Boerner and Brinkman 1996). However, very few studies have been conducted in these oak dominated forests to verify this assumption (Russell et al. 2001), and even less is known about the combined effects of timber harvest, prescribed fire, and white-tailed deer browsing on oak dominated forests with deer densities that more closely resemble pre-European settlement levels (approximately 3 to 8 deer km^{-2} ; McCabe and McCabe 1997, Russell et al. 2001, Horsley et al. 2003).

The objective of this study was to examine the effect of white-tailed deer browsing on the height, density, and composition of woody regeneration in oak hickory forests following thinning and prescribed fire treatments. Long-term results from this study should provide much needed information to help forest and wildlife managers in southern Ohio and the Central Hardwood Forest to make sound decisions about the management of these valuable forest and wildlife resources.

Study Area

This study was conducted on the Ohio Hills site of the Fire and Fire Surrogate Study (FFS; Prasad 2003) in mature oak-dominated forests at Tar Hollow (TAR) and Zaleski (ZAL) State Forests, and the Raccoon Ecological Management Area (REMA) in Ross and Vinton Counties of southern Ohio. These forests are located in the Unglaciated Allegheny Plateau of southern Ohio and are characterized by high hills, sharp ridges, and narrow valleys (McNab and Avers 1994). Elevations range from over 300 m to below 210 m MSL. Soils on all three forests are dominated by the Gilpin series and consist of mostly loams and silt loams that are acidic and well drained (Sutherland et. al. 2003, Boerner and Sutherland 2003).

Pre-harvest inventories (2000; data on file at Delaware Forest Service, Delaware, OH) indicate that oaks (*Q. prinus* L., *Q. alba* L., *Q. velutina* Lam., *Q. rubra* L. and *Q. coccinea* Muench.) represented 74.6, 86.0 and 88.5 percent of the basal area of canopy trees at TAR, REMA and ZAL, respectively. The greatest percentage of oaks occurred on the xeric and intermediate sites. Dendroecological analysis of dominant trees harvested from these sites, indicate that these forests established from 1850 to 1870, 1870 to 1905 and 1895 to 1905 at REMA, ZAL and TAR, respectively (data on file at Delaware Forest Service, Delaware, OH). Ohio Department of Natural Resources, Division of Wildlife, estimates of post-harvest deer densities from 2000 to 2002 averaged 6.3 and 5.4 deer km^{-2} for Vinton (REMA and ZAL) and Ross Counties (TAR), respectively (Data on file at the Waterloo Wildlife Research Area, Athens Ohio).

Materials and Methods

The Ohio Hills Site is comprised of four silvicultural treatment units (20 ha) at each location (REMA, TAR and ZAL). Treatments include the following: no treatment control (C), thin (T), thin and prescribed burn (TB), and prescribed burn only (B). Harvests in treatments T and TB consist of selective removal of canopy and mid-story trees in the fall and winter or 2000-2001 resulting in approximately $19.5 \text{ m}^2\text{ha}^{-1}$ of residual basal area (data on file at Delaware Forest Service, Delaware, OH). Oaks and hickories (*Carya* spp. Nutt.) were favored when possible, while red maple and other shade tolerant species were targeted for removal. Prescribed burns were conducted on the B and TB sites in March and April of 2001. Fire behavior varied within and among treatments, but flame heights averaged less than 1 m at all sites.

One pair of 400 m² (20 × 20m) plots was established within each of three moisture classes (mesic, intermediate and xeric) based on an integrated moisture index (IMI; Iverson and Prasad 2003) in each of the four treatments at all three locations during the summer of 2001. One plot from each pair was randomly chosen for installation of a deer proof fence in the summer of 2001. Each enclosure consists of 2.4 m black mesh barrier made of UV stable polypropylene installed just beyond the perimeter of the plot. A total of 72 plots (36 fenced and 36 unfenced) were established. Species and DBH of mid-story and overstory vegetation were recorded at the time of establishment.

Twenty-0.5 m² sub-plots were systematically spaced within each plot at the time of fencing (June/July 2001). Woody regeneration between 10 and 150 cm in height was identified to species, and heights (cm) and basal diameters (mm) were recorded at the time of plot establishment. Relative importance values (RIV = [relative density + relative frequency]/2) were calculated for all woody regeneration (10 to 150 cm height). Each woody stem was examined to determine the percentage of branches browsed and a browse class from 0 (0 percent) to 4 (> 75 percent) was assigned. Woody regeneration measurements were repeated in late May to early June of 2002 and 2003.

Correspondence Analysis (CA) was used to evaluate the changes in species composition before, and two years following silvicultural and fencing treatments. An Analysis-of-Variance (ANOVA) was performed on overall mean height of all woody regeneration and on twelve individual woody species by silvicultural treatment (TRT), integrated moisture index class (IMI), and fencing treatment (FENCE). All treatments were treated as fixed effects except location, which was treated as a random effect and dropped from the analysis of higher order interactions. Data were tested for normality and no transformations were warranted.

Results

Basal area of overstory and midstory trees (> 5.0 cm DBH) across all location and silvicultural treatment combinations was dominated by oak, with red maple, yellow-poplar, and hickory comprising a majority of the remaining basal area. Total basal area was 23.1 percent lower on thinned (T and TB; 21.85 ± 1.6 m²ha⁻¹; mean ± SE) than on unthinned plots (C and B; 28.4 ± 2.0 m²ha⁻¹). The greatest species difference in basal area between thinned and unthinned plots occurred with red maple which had 70 percent less basal area on the thinned plots (1.0 ± 0.2 m²ha⁻¹) than on unthinned plots (3.3 ± 0.5 m²ha⁻¹). In contrast, total oak basal area was only 23.2 percent lower on thinned plots (16.1 ± 1.1 m²ha⁻¹) than on unthinned plots (20.9 ± 1.8 m²ha⁻¹).

Relative importance values (RIVs) of three species of woody vines and shrubs accounted for over 44 percent of the total importance of woody regeneration (10 to 150 cm height) at the time of fencing in 2001 (table 1). Mean RIV's (across all treatments) for greenbrier (*Smilax rotundifolia* L.; 23.6) were over two times greater than those of any other species. Ericaceous shrub species (*Vaccinium* spp. L. and *Gaylussacia* spp. HBK.) had a combined mean RIV of 10.6, followed closely by viburnums (mostly *V. acerifolium* L., 9.9). Among tree species, sassafras (*Sassafras albidum* Nees.) had the greatest mean RIV (10.2) which was followed closely by red maple (9.3) and oaks (all spp. combined, 8.9). Chestnut oak had the highest mean RIV (3.9) of the oaks followed by white oak, which had a RIV of 1.9.

Correspondence analysis indicated that while the thinning treatments resulted in considerable compositional change, presumably due to increased light to the understory, the deer enclosure treatments yielded little change (fig. 1). Thus, within this period, deer were unable to produce a compositional change in the community (i.e., did not selectively browse certain species to depletion).

ANOVA results reveal significant IMI ($P = 0.025$) class and FENCE ($P = 0.035$) effects on the mean height of woody regeneration (table 2). Seedlings were significantly taller on plots in xeric locations (47.6 ± 2.9 cm; mean ± SE) than on intermediate (37.3 ± 2.9) and mesic (38.1 ± 2.9) locations (fig. 2). Mean height of regeneration was also significantly greater on fenced (44.6 ± 2.3 cm.) than on unfenced plots (fig. 3). There were no significant effects due to TRT ($P = 0.078$), and none of the interactions among TRT, IMI and FENCE were significant.

Table 1.—Relative Importance values (Relative Density + Relative Frequency/2) of woody regeneration at the time of fencing in 2001 by location and treatment.

Species	REMA				TAR				ZAL			
	Control	Thin	Thin/ Burn	Burn	Control	Thin	Thin/ Burn	Burn	Control	Thin	Thin/ Burn	Burn
<i>Acer rubrum</i>	9.07	15.18	6.96	11.87	13.89	14.92	2.60	9.47	4.09	4.54	4.54	14.76
<i>Ailanthus altissima</i>	0.00	0.00	0.00	0.00	0.00	0.00	13.88	0.00	0.00	0.00	0.00	0.00
<i>Carpinus caroliniana</i>	0.87	1.93	2.32	0.82	0.74	2.25	0.00	0.00	3.31	0.38	0.00	0.00
<i>Carya glabra</i>	1.31	0.88	0.00	2.38	0.74	0.00	0.00	0.00	2.58	1.76	2.56	1.54
other <i>Carya</i> spp. ¹	0.87	0.88	0.00	1.15	0.00	0.00	0.43	0.40	0.00	0.00	0.00	0.99
<i>Corylus americana</i>	1.72	0.00	0.00	1.75	0.00	0.47	0.00	0.00	0.00	0.00	0.00	0.00
<i>Cornus florida</i>	1.88	0.00	3.33	4.13	0.00	0.75	0.00	1.05	1.63	0.38	0.00	6.19
<i>Fraxinus</i> spp.	0.87	0.88	1.00	0.00	0.37	9.59	0.34	1.12	1.12	0.88	0.00	0.99
<i>Liriodendron tulipifera</i>	0.44	0.00	0.00	11.34	2.54	0.75	2.21	1.45	0.00	0.00	0.40	0.00
<i>Nyssa sylvatica</i>	0.87	3.86	2.16	6.49	3.75	0.37	1.87	1.20	1.12	0.00	8.12	2.74
<i>Oxydendrum arboreum</i>	2.02	2.41	0.66	0.00	0.00	0.75	0.00	6.65	3.14	0.00	0.92	0.00
<i>Parthenocissus</i> spp.	3.46	1.01	3.48	0.00	0.37	1.32	1.78	1.99	2.41	0.00	0.52	0.77
<i>Quercus alba</i>	2.18	6.66	4.81	0.97	0.74	1.12	0.34	0.00	1.68	2.25	0.81	1.76
<i>Quercus coccinea</i>	0.00	0.00	0.00	0.00	0.74	0.00	0.00	0.00	0.00	2.79	5.71	1.54
<i>Quercus prinus</i>	2.18	1.89	0.00	2.53	8.36	5.27	0.67	0.80	0.56	12.08	3.28	9.65
<i>Quercus rubra</i>	0.57	3.34	1.99	0.48	0.00	1.12	0.00	0.00	4.65	0.38	0.00	0.00
<i>Quercus velutina</i>	2.02	0.00	0.00	0.97	0.37	1.50	0.00	0.80	2.41	1.99	0.00	2.74
<i>Rhus</i> spp.	0.00	0.44	1.32	0.00	0.00	0.00	5.68	0.00	0.00	0.00	0.40	0.00
<i>Rosa</i> spp.	0.57	0.88	3.31	0.00	0.00	0.37	0.00	0.51	0.00	1.26	1.21	5.06
<i>Rubus</i> spp.	0.00	0.88	3.15	0.48	1.95	1.87	6.39	8.37	0.00	0.88	0.40	5.49
<i>Sassafras albidum</i>	6.45	9.97	4.64	15.54	5.87	3.30	17.38	14.63	17.51	15.39	9.80	1.97
<i>Smilax</i> spp.	22.25	21.06	27.04	10.91	45.60	39.61	18.24	32.73	12.62	13.53	27.41	15.26
<i>Tilia americana</i>	0.44	0.96	4.65	0.00	0.00	1.62	0.00	0.00	0.00	0.00	0.00	0.00
<i>Ulmus rubra</i>	0.00	0.00	0.00	0.00	0.74	6.42	2.64	2.72	2.36	0.00	0.00	6.23
<i>Vaccinium</i> spp. ²	19.91	6.05	5.31	21.89	3.18	0.00	2.02	2.57	0.00	30.86	19.87	15.75
<i>Viburnum</i> spp.	11.90	17.77	17.92	3.42	6.05	0.00	3.03	7.49	33.31	7.14	10.25	1.63
<i>Vitis</i> spp.	2.18	3.08	1.32	1.19	1.95	1.52	14.78	2.32	0.56	3.51	1.37	0.77
Others ³	5.99	0.00	4.64	1.67	2.06	5.07	5.72	3.73	4.94	0.00	2.41	4.16

¹Includes: *Carya ovata*, *Carya tomentosa* and *Carya cordiformis*

²Also includes: *Gaylussacia baccata*

³Includes: *Acer saccharum*, *Aesculus glabra*, *Amelanchier arborea*, *Castanea dentata*, *Cercis canadensis*, *Gleditsia triacanthos*, *Hamamelis virginiana*, *Lindera benzoin*, *Ostrya virginiana*, *Prunus serotina*, *Toxicodendron radicans*, *Ulmus americana*

Percentage of woody regeneration with evidence of browse in the unfenced plots increased from 13.8 in 2003 to 16.7 in 2003. In both years the percentage of stems browsed varied greatly among species. In 2003 these percentages ranged from 0.0 for few species which include sugar maple (*Acer saccharum* Marsh) and tree-of-heaven (*Ailanthus altissima* Desf.) to 46.8 for flowering dogwood (*Cornus florida* L.). Of the twelve woody species tested in the ANOVA, mean percentages of stems browsed (unfenced plots), across all locations and silvicultural treatments varied from 1.7 ± 1.7 for white oak to 27.1 ± 3.5 for greenbrier. Sassafras had the greatest percentage of stems browsed among tree species (20.4 ± 4.2). Blackgum was the only species tested that revealed significant fence effects ($P = 0.003$; fig. 4A; stems were 57.1 ± 5.4 and 34.9 ± 4.6 cm in height (mean \pm SE) in the fenced and unfenced treatments, respectively. FENCE also had an influential effect on sassafras and chestnut oak mean heights ($P = 0.082$ and 0.068 , respectively; table 3.).

Red maple, chestnut oak and green brier were the only species that exhibited significant TRT effects. Red maple seedlings were on average 14.9 ± 6.4 cm (mean \pm SE) in the control (C) as compared to 38.7 ± 6.4 cm and 38.3 ± 6.6 cm in height the thin only (T) and the burn only (B) treatments,

Table 2.—Analysis of Variance table for mean height of 2003 woody regeneration (10 to 150cm) all species combined.

Source Term	DF	Sum of Squares	Mean Square	F	P	Power
Location	2	5369.905	2684.952	13.60	0.000023*	
Treatment (Silvicultural)	3	1422.606	474.2019	2.40	0.079762	0.563277
IMI (Moisture Class)	2	1575.386	787.693	3.99	0.025243*	0.685947
LI	6	2245.833	374.3056	1.90	0.101811	0.641655
Fence Treatment	1	932.2346	932.2346	4.72	0.034966*	0.566550
TF	3	532.35	177.45	0.90	0.449053	0.231320
IF	2	354.1499	177.075	0.90	0.414834	0.195310
TCF	6	232.579	38.76317	0.20	0.976238	0.095232
S	46	9081.547	197.4249			
Total (Adjusted)	71	21746.59				
Total	72					

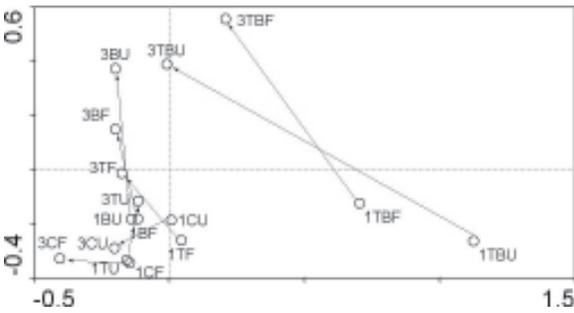


Figure 1.—Correspondence analysis of combined silvicultural (C, T, TB and B) and fence (F and U) treatment effects on species composition from 2001 (1) to 2003 (3).

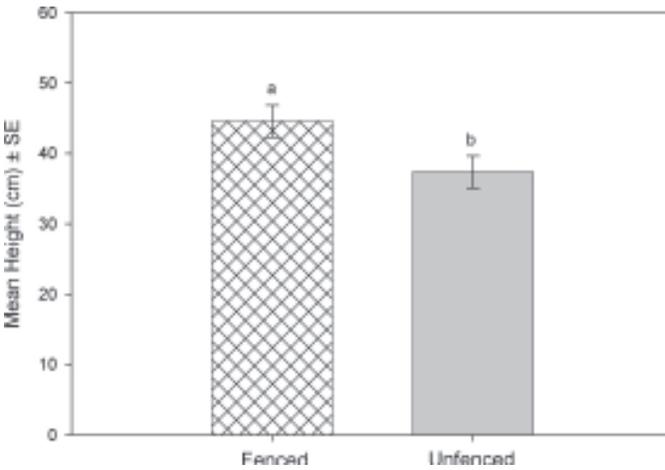


Figure 2.—Mean height of stems 10 to 150 cm by fencing treatment. Error bars represent 1 standard error.

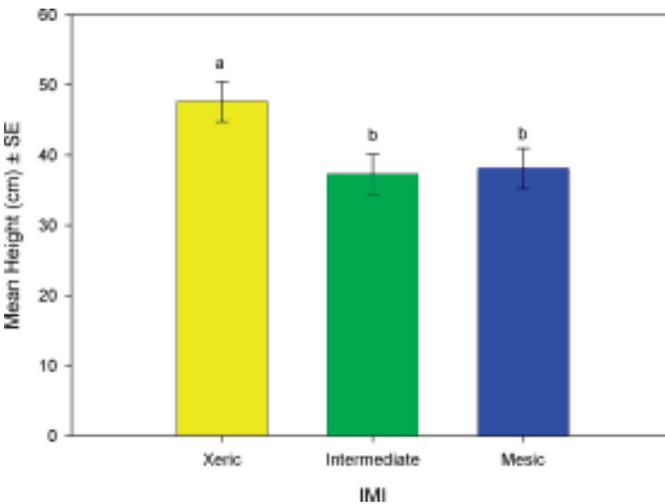


Figure 3.—Mean height of stems 10 to 150 cm by Integrated Moisture Index (IMI) Class. Error bars represent 1 standard error.

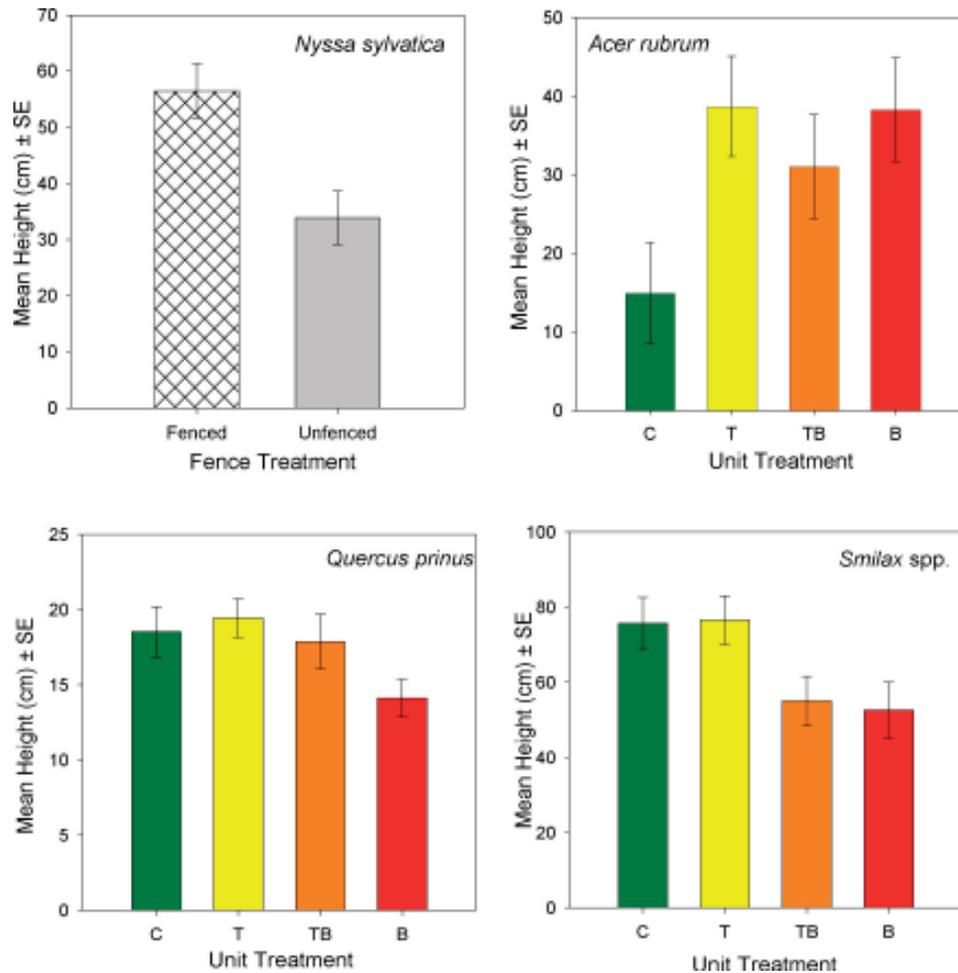


Figure 4.—Mean height of stems 10 to 150 cm by Silvicultural Treatment. C=control, T= thin; TB=thin and burn; and B=burn only. Error bars represent 1 standard error. A) blackgum (*Nyssa sylvatica*); B) red maple (*Acer rubrum*); C) chestnut oak (*Quercus prinus*); D) green brier species (*Smilax* spp.).

respectively (fig. 4B). Chestnut oak seedlings were taller in T (19.4 ± 1.3 cm) than in the B treatments (14.1 ± 1.2 cm; fig. 4C). Greenbrier stems were taller in C (75.7 ± 6.9 cm) and T (76.5 ± 6.5 cm) treatments than in TB (55.1 ± 6.5 cm) and the B treatments (52.6 ± 7.4 ; fig. 4D).

Discussion

There was a significant over-all increase in height of woody stems (all species combined) with the exclusion of deer, albeit this may be largely driven by the response of one species (blackgum). However, we found no evidence to show that browsing affected species richness, community composition, or density of hardwood regeneration within two years following harvest. This is not surprising since negative effects of deer on plant survival are more difficult to document than effects on growth rates especially on sites with less abundant deer populations (Russell et al. 2001, Inouye et al. 1993, Stang and Shea 1998). Russell et al. (2001) suggest that the under representation of saplings in some tree populations may reflect reductions in seedling survival; however, in a review of white-tailed deer browse studies he indicates that all of the studies that documented failures of recruitment of seedlings into larger classes were conducted on sites with deer densities above 8.5 deer km^{-1} . Our estimated deer densities (approximately 6.0 deer km^{-1}) are considerably lower than this potential threshold.

Table 3.—Mean percentage of stems browsed (unfenced plots) and mean height (cm) of selected species (10 to 150 cm) in 2003 by fencing treatment for all locations and silvicultural treatments.

Species	Percentage of stems browsed	Height (mean ± SE)	
		Fenced	Unfenced
<i>Acer rubrum</i>	13.1 ± 4.1	35.8 ± 5.7	25.7 ± 3.5
<i>Liriodendron tulipifera</i>	3.4 ± 1.6	24.5 ± 4.6	18.3 ± 1.4
<i>Nyssa sylvatica</i>	17.7 ± 7.1	57.1 ± 5.4	34.9 ± 4.6
<i>Oxydendrum arboreum</i>	3.1 ± 3.1	64.3 ± 11.9	54.8 ± 17.5
<i>Quercus alba</i>	1.7 ± 1.7	16.8 ± 2.1	20.0 ± 1.5
<i>Quercus coccinea</i>	15.5 ± 6.6	23.8 ± 4.0	21.9 ± 2.0
<i>Quercus prinus</i>	3.1 ± 1.8	23.7 ± 3.6	19.5 ± 3.4
<i>Quercus rubra</i>	23.8 ± 11.3	32.0 ± 8.2	23.4 ± 3.2
<i>Quercus velutina</i>	7.9 ± 4.3	17.8 ± .4	16.4 ± 1.7
<i>Sassafras albidum</i>	20.4 ± 4.2	32.0 ± 3.2	26.4 ± 2.7
<i>Smilax spp.</i>	27.1 ± 3.5	61.6 ± 6.2	60.1 ± 5.3
<i>Vaccinium spp.</i>	15.0 ± 5.4	22.2 ± 1.28	22.5 ± 1.4

Even if white-tailed deer densities are large enough to have a significant effect on growth and survival of woody regeneration on our sites, it is likely that it is too early in this study to detect them. In Wisconsin, Jacobs (1969) documented a 10 percent reduction in sugar maple survival over a 5 year period, but no saplings died until the fourth year. Other studies only demonstrate infrequent effects of deer. For instance Inouye et al. (1994) were only able to detect reduced rates in height growth of red oak and white pine (*Pinus strobus*) in years two and four of a nine year study. According to Russell et al. (2001) studies of short duration (1 to 3 years) may not be long enough to detect changes in rates of overstory regeneration.

Despite frequent suggestions that browsing contributes to poor oak survival and recruitment (Clark 1993, Inouye et al. 1994, Boerner and Brinkman 1996, Stange and Shea 1998, Healy 1997), oak seedlings do not appear to be directly affected by deer browsing after two years of exclusion. Only about 2.5 percent of oak stems were browsed in 2002 (Apsley 2002) and seedling heights were not significantly affected. The percentage of oak stems browsed increased to 10.6 in 2003 with red oak having the greatest percent browsed (23.1). Even with this increase in browsing, which is likely attributed to a greater than average number of days with snow cover in the winter of 2002-2003, there is still no significant effect of deer browsing on oak seedling height. However, there is evidence that oak stump sprouts have been affected. Since oak stump sprout density is relatively low and stump sprouts only occurs on the cut treatments, they are not well represented in the sampling using 0.5 m² sub-plots. In the summer of 2003 oak stump sprouts within the 400 m² plots on the thin treatments at ZAL were observed and 83 percent of stump sprouts in unfenced areas had evidence of browse damage. The dominant oak stump sprout was 212.7 ± 11.2 and 159.8 ± 17.9 cm in height (mean SE) in fenced and unfenced plots, respectively. It is unclear whether this will have a significant effect on the ability of these sprouts to become established within the stand. Sander et al. (1984) indicate that stump sprouts can be a significant source of oak regeneration and that stems greater than 1.5 m in height are capable of becoming established if the canopy is reduced enough to allow them to compete with more shade tolerant species.

At lower densities white-tailed deer may have an indirect effect on oak regeneration, and it is not yet clear whether this effect will be positive or negative. Red maple, which is often cited as the major impediment to oak regeneration in upland oak forests (Heiligmann et al. 1985, Lorimer 1993) does not appear to be significantly influenced by deer browse. Only 10.8 percent of red maple stems have been browsed and average heights of fenced and unfenced red maple are not significantly different. However, several other non-commercial vine, shrub, and tree species are being more heavily browsed, and the over all height of stems has been reduced. This may eventually have an indirect positive effect

on the establishment of oak stems. This effect is most apparent on intermediate and xeric sites, but it has not yet resulted in increased height of oak seedlings. This is not surprising since the silvicultural and fencing treatments were only established for two growing seasons at the time of the last sampling.

Finally, several studies indicate that it can take several years with multiple stage canopy removal for oak seedlings to become established (Sander and Graney 1993). The harvest treatments were prescribed to reduce the basal area to 14 m²·ha⁻¹; however, the goal was not reached and an average residual basal average of 19.5 m²·ha⁻¹ remains on harvested treatments (data on file at Delaware Forest Service, Delaware, OH). These treatments may not have provided adequate light for oak regeneration to become established (Hodges and Gardiner 1993). Since fire has been effectively excluded from these sites since 1923 (Sutherland 2003), red maple and other shade tolerant species have had the opportunity to become well established. It may take repeated burns or other silvicultural treatments to reduce the mid-story canopy enough to stimulate oak regeneration (Brose and VanLear 1998).

The second year results from this study indicate minimal direct and indirect effects of deer browse on oak seedlings and their competitors. However, oak seedlings are relatively sparse and deer populations are relatively low (Horsley et al. 2003). Since it will likely take several years for these sites to fully regenerate (Sander and Graney 1979) and since deer densities fluctuate annually, the effect of deer herbivory may not be fully realized for some time on these southern Ohio sites. We plan to continue monitoring these plots through at least 2005.

Acknowledgments

This research was funded by the National Fire Plan. Although the authors received no direct funding from the U.S. Joint Fire Science Program (JFSP) for this research, it could not have been accomplished without JFSP support of existing Fire and Fire Surrogate project sites.

Thanks to Joni Downs and Chris Giesting for their leadership in the field; Shelly Hesson and Lois Campbell for data entry; and several forestry students from Hocking College, too numerous to mention individually, for field assistance.

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LANDSCAPE AND LOCAL INFLUENCES OF FOREST MANAGEMENT ON CERULEAN WARBLERS IN PENNSYLVANIA

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ABSTRACT.— Populations of cerulean warblers (*Dendroica cerulea*) have declined rangewide, and conservation of this mature-forest species is poised to become a major forest management issue in central hardwood forests. Although cerulean warblers are associated with large trees in mature forest, cerulean warblers also use edges of timber harvests, roads, and ridgetops within heavily forested areas – all habitats with heterogeneous canopy structure. Thus, certain types of forest management may be compatible with conservation of this declining species and may even create habitat by creating canopy gaps. In this paper, I examined data from the forested landscapes of Pennsylvania (1997-1999) and asked how local and landscape-level characteristics influenced the occurrence of cerulean warblers. More specifically, I examined the extent to which amount of forest cover within 1 km, distance from recent harvests (<5 years old), and habitat structure were associated with presence of cerulean warblers. Occurrence of cerulean warblers on 17 mature oak-hickory forest stands was best explained by distance to the nearest timber harvest. Within forested landscapes of Pennsylvania, cerulean warblers were more likely to occur closer to, rather than farther from recent silvicultural activity. However, this model was closely ranked to other univariate models containing forest cover, canopy cover, large trees, and snags. Results suggest that cerulean warblers respond to a suite of local and landscape-level factors.

Conservation of cerulean warblers (*Dendroica cerulea*) is poised to become a major forest management issue in eastern deciduous forests. Cerulean warblers were once fairly common birds in Appalachian hardwood forests, but populations of this mature-forest warbler have declined rangewide since 1966, with annual declines of –4.4 percent (Sauer et al. 2003). The Ohio Hills and Northern Cumberland Plateau Physiographic Provinces still hold the greatest breeding concentrations of this warbler (Rosenberg et al. 2000). Even in these areas, though, steep declines are evident (Sauer et al. 2003), making population trends especially alarming. Cerulean warbler is listed as a species of concern by U.S. Fish and Wildlife Service, a high priority species by Partners-In-Flight, and a WatchList species by the National Audubon Society. Cerulean warbler also was petitioned for federal protection (threatened status) in 2002. If this species is federally protected under the Endangered Species Act (ESA), forest management in eastern deciduous forests will be dramatically impacted. Even if the bird is not formally protected under ESA, conservation of cerulean warblers will remain an important goal of agencies and conservation organizations.

Anthropogenic land use changes are probably the primary cause of population declines for cerulean warblers. In particular, fragmentation of mature, deciduous forest on breeding grounds is cited as an important contributing factor (Hamel 2000). Indeed, cerulean warblers are usually considered area-sensitive, and minimum patch sizes range from 10->8,000 ha (Hamel 2000). Such variation in area requirements is likely a consequence of different amounts of regional forest cover. The species also may be sensitive to land uses within landscapes. For example, cerulean warblers were more likely to occur in forested landscapes disturbed by silviculture than forested landscapes disturbed by agriculture (Rodewald and Yahner 2001).

Paradoxically, forest harvesting has the potential to be an important tool for the conservation of cerulean warblers, as it may improve habitat conditions by creating canopy gaps and heterogenous forest structure. Cerulean warblers often are associated with well-spaced large trees with high canopies

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(Jones and Robertson 2001). They also use edges of timber harvests, roads, and ridgetops within heavily forested areas. For example, cerulean warblers were detected in and adjacent to variable retention stands containing ca. 100 trees/ha in central Pennsylvania, (Rodewald and Yahner 2000). Similarly, these birds frequently sang in trees adjacent to clearcuts (4-20 ha) and forest roads in the forested landscapes of southern Ohio (A.C. Vitz, pers. obs). Due to their apparent preference for tall, uneven forest canopies, shorter rotation periods and even-aged silvicultural methods may negatively affect cerulean warblers because fewer stands reach maturity (Hamel 2000).

Presently, efforts to manage many forest wildlife species are constrained by inadequate knowledge about responses to silvicultural activities (Thompson et al. 2000), and this is especially true for cerulean warblers. Forest managers lack a thorough understanding of how this sensitive species responds to forest management at both local and landscape scales. I used data from the forested landscapes of Pennsylvania (1997-1999) to determine how local and landscape-level forest management activities influenced the occurrence of cerulean warblers. More specifically, I examined the extent to which amount of forest cover within 1 km, distance from recent harvests (<5 years old), and habitat structure were associated with presence of cerulean warblers.

Study Area

This study was conducted in Rothrock and Bald Eagle State Forests in central Pennsylvania. These state forestlands were primarily comprised of mature forest with scattered even-aged harvests (7-38 ha, 1-5 years old) and intermittently occupied residences. Thus, the landscape matrix was forested, and silvicultural disturbances occurred as patches within the forest matrix. Most roads in the study area were unpaved, narrow (<20 m wide) roads covered by a relatively unbroken forest canopy. Sites had little slope, were 250-500 m in elevation, and did not occur along ridgetops. Tree species common in the study area included white oak (*Quercus alba*), northern red oak (*Q. rubra*), chestnut oak (*Q. prinus*), red maple (*Acer rubrum*), sugar maple (*A. saccharum*), black gum (*Nyssa sylvatica*), black cherry (*Prunus serotina*), and hickories (*Carya* spp).

Methods

I selected 17 25-ha forest sites within contiguous mature forest (approximately 80-110 years old) and delineated 1-km radius landscapes centered on each study site. Because cerulean warbler territories average 1 ha (range = 0.38-2.4 ha; Oliarnyk and Robertson 1996), a 314-ha area was thought to represent a landscape in that it could contain multiple conspecific territories and multiple land cover types. I calculated forest cover from classified thematic mapper imagery using ARC/INFO geographic information system software (ERSI 1997). Even-aged harvests, such as clearcuts less than 15 years old, were classified as non-forest. Older sapling or pole-stage forests could not be distinguished from mature forest by thematic mapper imagery. Uneven-aged techniques had not been widely applied in the study area. Recent harvests were digitized manually using ARC/VIEW geographic information system software. Because most (>80 percent) non-forest cover within the landscape consisted of recent (<15 years) even-aged harvests, percent forest cover was used as an inverse indicator of the amount of silvicultural activity within the landscape. I also calculated distance (m) to nearest recent harvest (<5 years old) directly from maps. This measure indicated the proximity of intense silvicultural activity to the study site, which is relevant to short-term displacement and disturbance. Sites averaged 76 percent \pm 13.7 forest cover within 1 km (range = 49-95 percent) and 550 m from a recent harvest \pm 348 m (range = 150-1000 m).

Cerulean warblers were sampled at 3 points located at 150-m intervals along a transect bisecting each study site from 30 May-25 June 1997-1999. Points were at least 150 m from habitat edges (e.g., road or clearcut), and all were located in mature forest. Twice per year, 10-minute, 50-m radius point counts were conducted between 0545-1045 on days without strong wind (> 20 mph) or rain. To minimize temporal effects, I surveyed sites in reverse order on the day of the second visit. Abundance data for cerulean warblers were summed over the three point counts per site and averaged over the two visits in each year (to produce one measure of abundance for each site per year). I later converted these data to simple presence-absence (1,0) for each site (over all 3 years combined).

Habitat characteristics were measured in a 0.04-ha (11.4-m radius) circular plot centered on each of the three bird-sampling points in late June 1997. Numbers of live trees by species in three dbh (diameter breast height) classes (8-23 cm, 23-38 cm, and > 38 cm dbh) and number of snags (standing dead trees \geq 15 cm dbh) were recorded. I also established two 20-m perpendicular transects through the center of each sampling point in north-south and east-west directions. The forest canopy was measured at 2-m intervals along these transects using an ocular tube (hit, no-hit) and used to calculate percent canopy cover (total number of hits/20 points).

Because use of traditional hypothesis-testing in observational studies has been criticized recently, I used an information-theoretic approach, which is considered to be more robust than traditional hypothesis-testing for many ecological studies because it emphasizes fitting and ranking explanatory models as opposed to using *P*-values (Burnham and Anderson 1998, Johnson 1999). Due to my limited sample size ($n=17$ sites), I could examine only a small set of explanatory models. Therefore, I developed a set of 9 *a priori* candidate models based on the known ecology of the cerulean warblers, which are known to prefer large trees (variable = number of trees >38 cm dbh), heterogeneous canopy with gaps (variables = canopy cover and numbers of snags), high amounts of forest cover (variables = percent forest cover within 1 km and distance to nearest harvest) (Table 1).

I used PROC GENMOD (SAS Institute, Inc. 2001) to test candidate models using a binary distribution with a logit link. From the log-likelihood values generated by the GENMOD procedure, I calculated a bias-corrected version of Akaike's information criteria (AIC_c), differences among models (ΔAIC_c), and Akaike weights (ω) to rank and select the model (s) best supported by the data (Burnham and Anderson 1998). The model with lowest AIC_c value and $\Delta AIC_c = 0.00$ was considered the best explanatory model (Burnham and Anderson 1998). Confidence intervals also were considered when judging the relative importance of variables.

Results

Cerulean warblers were detected on 6 of the 17 sites. A model containing distance to nearest harvest best explained variation in the occurrence of cerulean warblers, and the weight of evidence for this model was 1.5x greater than the next model (Table 1). Occurrence of cerulean warblers was negatively related to distance to harvest, meaning that birds were more likely to occur closer to, rather than farther from, harvests (parameter estimate = -0.0021 ± 0.0017). However, 4 univariate models were closely ranked to the top model ($\Delta AIC_c < 2.0$), which means that they are also plausible given the data. These models included the following variables: forest cover (0.038 ± 0.043), canopy cover (0.036 ± 0.067), snags (-0.10 ± 0.25), and large trees (-0.46 ± 0.38) (Table 1). However, Wald 95% confidence intervals for each of these variables included zero, suggesting that even the best models were not strongly related to occurrence of cerulean warblers.

Discussion

Occurrence of cerulean warblers in the managed forests of Pennsylvania was not strongly related to the local and landscape variables examined here, which may reflect the relatively small sample size used in this study. Nevertheless, the patterns identified can prove useful in filling some of the knowledge gaps regarding the ecology of the species and its response to forest management.

At the landscape scale, presence of cerulean warblers was negatively related to distance to nearest harvest and positively related to forest cover within 1 km. The positive relationship with forest cover is consistent with the species' reported preference for forested landscapes. The affinity of cerulean warblers for forests near recent harvests is less readily explained, but it may result from increased canopy gaps adjacent to harvests and the unimproved forest roads that access them. Indeed, anecdotal observations indicate that, in many parts of their range, cerulean warblers commonly sing adjacent to clearcuts and along forest roads. Other accounts report that the species can tolerate disturbances, such as hurricanes or ice storms (Hamel 2000). For example, Jones et al. (2001) reported that, after initial declines, populations of cerulean warblers were relatively resilient to a large-scale natural disturbance (ice storm) in southern Ontario.

Table 1.—Results from the information-theoretic approach applied to 9 *a priori* models to explain variation in the occurrence of cerulean warblers at 17 forest sites in central Pennsylvania, 1997-1999. Log-likelihood ratios were calculated by Proc GENMOD (SAS Institute Inc., 2001). The number of parameters (k) includes estimates for main factors listed and an intercept value.

Candidate Models	Log-likelihood	K	AICc	$\Delta AICc$	w_i
Distance to harvest	-10.18	2	25.06	0.00	0.282
Forest cover	-10.61	2	25.92	0.86	0.183
Canopy cover	-10.89	2	26.49	1.43	0.138
Snags	-10.95	2	26.61	1.55	0.130
Large trees	-11.03	2	26.77	1.71	0.120
Forest cover, distance to harvest	-10.13	3	27.76	2.70	0.073
Snags, canopy cover	-10.84	3	29.18	4.12	0.036
Snags, large trees	-10.94	3	29.38	4.32	0.033
Full model (including all variables)	-7.32	6	33.10	8.04	0.005

Table 2.—Correlations among local and landscape variables for 17 forest stands in central Pennsylvania, 1997-1999. Pearson correlation coefficient (r) followed by P-values are listed; n = 17 in all cases.

	Distance to harvest (m)	Percent forest cover
Number of snags (≥ 15 cm dbh)	-0.515, 0.035	0.480, 0.051
Number of small trees (8-24 cm dbh)	-0.344, 0.176	0.887, <0.001
Number of medium trees (24.1-38 cm dbh)	-0.467, 0.059	0.765, 0.003
Number of large trees (>38 cm dbh)	-0.420, 0.093	0.289, 0.260
Number of understory stems (< 8 cm)	-0.414, 0.098	0.230, 0.375
Number of logs	-0.443, 0.075	0.351, 0.168
Percent grass cover	0.197, 0.448	-0.504, 0.039
Percent bare ground cover	-0.177, 0.496	0.654, 0.004
Percent canopy cover	0.036, 0.892	0.303, 0.238

Despite the controversy that sometimes accompanies timber operations, ecologists have often suggested that forest management and maintenance of multiple seral stages within forests may benefit bird communities. For example, Thompson et al. (1992) showed that compared to unmanaged mature forests (i.e., without timber harvests), managed-forest landscapes containing multiple seral stages as the result of clear-cutting (i.e., 10 percent regeneration, 10 percent sapling, 80 percent pole-timber within 200 ha) contained equal or greater densities of most forest interior migrants, greater densities of shrubland birds, and no differences in avian nest predators and brood parasites.

At local scales, occurrence of cerulean warblers was positively associated with stands containing greater canopy cover, fewer snags, and fewer large trees than other stands. As a whole, these associations are less intuitive than ones at the landscape scale, and the patterns seem to contradict what is known of habitat selection by cerulean warblers (Hamel 2000). The apparent mismatch between findings at local and landscape scales may be, in part, explained by correlations among variables (Table 2). For example, distance to harvest was weakly correlated with numbers of snags and medium-sized trees, while percent forest cover was correlated with numbers of snags, small and medium trees, grass and bare ground cover. Thus, it is possible that cerulean warblers were selecting for landscape features that were simply correlated with local ones (or vice versa).

One important caveat of my study is that occurrence in a stand does not necessarily indicate that quality habitat for that species. For example, cerulean warblers may prefer to nest near harvests because

they provide a more open canopy. However, if harvest edges attract nest predators or brown-headed cowbirds (*Molothrus ater*, a brood parasite), then stands could act as ecological traps (Gates and Gysel 1978). In general, though, silvicultural edges within forested landscapes are not thought to negatively impact forest birds to the same extent that agricultural edges do (Bayne and Hobson 1997). In fact, most studies of edge-related nest predation in forested landscapes have found no significant edge effects (Andrén 1995, Hartley and Hunter 1998, but see King et al. 1998, Manolis et al. 2000). Within the same Pennsylvania study system, Rodewald (2002) found no evidence of edge-related nest predation for common forest birds.

Another consideration is that the forested landscape context of my study system may have dampened negative effects of silviculture, and relationships could dramatically change if the extent of forest cover was reduced. Regional forest cover is known to mediate the effects of edge and area, and can profoundly affect the influence of landscape-scale factors on forest birds (e.g., Robinson et al. 1995, Donovan et al. 1997). Maintenance of a mature forest matrix may be a key factor governing use of silviculture in conservation efforts for cerulean warblers.

As populations of cerulean warbler continue to decline, natural resource managers and biologists must carefully consider forest management activities in the context of conserving cerulean warblers and their mature-forest allies. Unfortunately, inadequate knowledge of species' basic habitat and landscape requirements and demographic responses to habitat alterations impede effective conservation. Hopefully, additional investigations by researchers conducted throughout the range of cerulean warblers will elucidate these relationships in the near future.

Acknowledgments

I am grateful to J. Avery and A. Lyter for field assistance. R. H. Yahner, M. C. Brittingham, W. L. Myers, P. G. Rodewald, and K. C. Steiner commented on the larger study design and survey methods. P. G. Rodewald provided helpful comments on this manuscript. As part of a larger study, research funds were provided by the Wild Resource Conservation Fund of the Pennsylvania Game Commission, SDE / Graduate Women in Science, Inc., and the Wilson Ornithological Society (Paul A. Stewart Award). I also thank the Henry Luce Foundation for the Penn State Women in Science and Engineering Clare Boothe Luce Fellowship and the Penn State School of Forest Resources for support.

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USING WOOD WARBLER FORAGING BEHAVIOR TO UNDERSTAND AVIAN RESPONSES TO FOREST MANAGEMENT

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ABSTRACT.—Although uneven-aged forest management often retains large numbers of canopy trees that can benefit mature forest birds, understory vegetation also may be removed. In a concurrent study, mature stands subjected to understory removal and canopy thinning had few forest understory birds relative to mature stands (Rodewald and Smith 1997). We hypothesized that this difference was related to foraging habitat requirements. We studied foraging behaviors of Black-and-white, Worm-eating, and Hooded Warblers in mature oak-hickory forests of northwest Arkansas during May-July 1993-94. Worm-eating and Hooded warblers were strongly associated with forest understory, with foraging heights averaging < 5 m and trees used for foraging averaging < 8 m. Black-and-white Warblers foraged at greater heights (mean = 8.5 m), and in taller trees (mean = 15.1 m), and this may explain their lower sensitivity to understory management. Foraging Worm-eating Warblers used aerial leaf litter and live foliage substrates (especially flowering dogwood) in equal amounts (39 percent), and Hooded Warblers used live foliage substrates heavily (88 percent); both substrates were heavily reduced by management. Black-and-white Warblers primarily used bark substrates (84 percent), which were less affected by management. Results suggest that foraging ecology of Worm-eating and Hooded Warblers may be incompatible with management practices that involve understory removal. Alternative practices that remove less understory vegetation could lessen negative effects on understory birds in hardwood forests.

With rising concern for conservation of forest birds over the last 25 years, forest management has become increasingly complicated as managers struggle to conduct silvicultural activities while minimizing negative impacts on mature forest birds and other wildlife. Uneven-aged management is a silvicultural approach suggested to strike a balance between these potentially conflicting goals. Certainly, retention of large numbers of overstory trees within stands will promote use by many forest bird species. However, avian ecologists have some concerns about uneven-aged management if widely used to replace more traditional even-aged methods. Use of uneven-aged management in oak-hickory forest presents a great challenge, as oak does not regenerate well in dense shady stands. Consequently, in 1992, the Southern Research Station, U.S. Forest Service, initiated research that examined effects of uneven-aged management on growth and regeneration of oak-hickory stands in the Ozark-St. Francis National Forest (Graney and Murphy 1993). Management practices used in that study included a heavy cutting of understory vegetation, either alone or in combination with both thinning and group-selection cutting of overstory trees. After treatment, forest stands had an open park-like understory with scattered small gaps in the canopy. An unintended result of these forest management practices was avoidance of treated stands by common understory nesting birds, including Worm-eating (*Helmintheros vermivorus*) and Hooded (*Wilsonia citrina*) warblers, Ovenbird (*Seiurus aurocapillus*), and Acadian Flycatcher (*Empidonax virescens*), but not for Black-and-white Warblers (*Mniotilta varia*) (Rodewald and Smith 1997). Because these species are known to frequently forage in the forest understory, foraging habits were suspected to have played a role in reduced abundances within treated stands.

We quantified foraging behavior of Worm-eating, Black-and-white, and Hooded warblers during the breeding season in mature oak-hickory forests of the Arkansas Ozarks. We discuss foraging behavior and habitat use in relation to uneven-aged forest management practices used in the Arkansas Ozarks. This information is useful in identifying habitat characteristics of importance to these species and to successfully design uneven-aged management practices that lessen their effects on forest understory birds.

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Study Area

Study sites were located in Ozark-St. Francis National Forest, Pope and Newton counties, northwest Arkansas. Forests in that area are oak-hickory with little pine and elevations ranged from 400-620 m. Common tree species included: white oak (*Quercus alba*), northern red oak (*Q. rubra*), black oak (*Q. velutina*), red maple (*Acer rubrum*), sugar maple (*A. saccharum*), black gum (*Nyssa sylvatica*), mockernut hickory (*Carya tomentosa*), black hickory (*C. texana*), black cherry (*Prunus serotina*), and flowering dogwood (*Cornus florida*).

Methods

From May-July 1993-94 we collected foraging data in mature, mesic secondary forests that were 80-100 years old and occurred on north- and east-facing slopes. Approximately 70 percent of foraging data were collected within 250 m of vegetation sampling plots (see below); the remaining 30 percent were more widely collected, but still within 10 km of study sites. Data were collected throughout the day, but mostly between 0700-1200 hours. After locating a foraging warbler, we waited at least ten seconds before recording data on the first foraging maneuver observed. We collected observations on as many individual birds as possible to minimize potential influence of individual differences in behavior. We alternated sites of data collection on a daily basis to try to avoid collecting data from the same individual on different days within a given year. Up to three observations were recorded from some individuals, but for most individuals, only a single observation was recorded. To increase statistical independence of observations, multiple observations from an individual bird were included in analyses only after at least two foraging maneuvers had occurred since the previous observation.

We collected data on type of foraging maneuver, foraging substrate, foraging height, tree height, and tree species in which maneuver took place. Foraging maneuvers were recorded as either glean (near-perch attacks directed towards prey on a substrate) or sallies (wing-powered attacks used to capture aerial prey). We defined foraging substrates as the substrate towards which a foraging maneuver was directed and these included bark surface, foliage, foliage curls (leaf shelters of leaf-rolling caterpillars), aerial leaf litter (dead leaves suspended in vegetation), air, spider webs, and ground leaf litter. Foraging height and tree height were visually estimated.

In June-July 1993, woody vegetation was characterized at 12 point-count locations in mature secondary forest on north- and east-facing slopes (Rodewald and Smith 1997) using a modified James and Shugart (1970) approach. Four vegetation plots were completed at each point: one at the point-count center, and three located 35 m from the center in directions of 120°, 240°, and 360°. All shrubs and saplings (0-8 cm in diameter at 10 cm above the ground) were counted by species within a 5 m radius. Trees greater than 8 cm DBH were counted by species within an 11.3 m radius.

Chi-square tests were used to compare frequencies of substrate use, tree species use, and foraging maneuvers across bird species. To minimize numbers of cells with low values in contingency tables, only the six most commonly used tree taxa were included, black gum, dogwood, elms (*Ulmus* spp.), hickories, oaks, and maples. Likewise, rarely-used substrates (e.g. spider web) were not included in contingency tables. Chi-square was also used to test whether birds foraged in tree species in proportions different than the relative availability of tree species. This test used the same tree taxa mentioned above, but to minimize numbers of cells with low values, did not include elms. Low sample sizes for Hooded Warbler prevented an analysis of selectivity. Foraging height and tree height data were not normally distributed, so comparisons among species were made using Kruskal-Wallis test (SAS Institute 1990).

Results

Sample sizes for foraging observations were: Hooded Warbler 51, Black-and-white Warbler 89, and Worm-eating Warbler 274. The three warbler species differed in their foraging behavior in several respects. Warblers differed in use of the six most frequently used tree taxa ($\chi^2 = 69.3$, $df = 10$, $P < 0.001$; Fig. 1). Worm-eating Warblers foraged most often in dogwood (40.7 percent of observations),

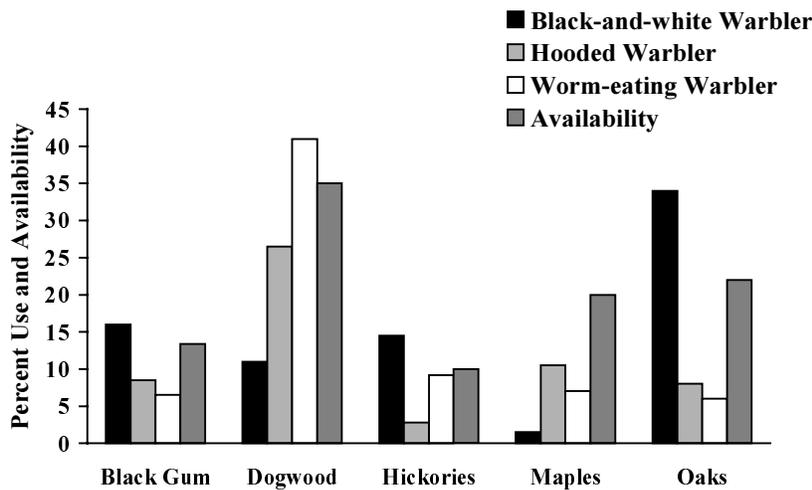


Figure 1.—Percent use of tree taxa by three wood warbler species in relation to percent availability in the Arkansas Ozarks, May-July 1993-94.

whereas other trees were used in much lower frequencies (hickories 9.7 percent, black gum 6.6 percent, and oaks 6.2 percent). Hooded Warblers also foraged in dogwood most frequently (26.1 percent), and used other plant taxa less frequently (black gum 8.7 percent and oaks 8.7 percent). Alternately, Black-and-white Warblers were most frequently observed foraging in oaks (34.2 percent), and less frequently in black gum, hickory, and dogwood (15.9 percent, 14.6 percent, and 11.0 percent, respectively). Hooded, Black-and-white, and Worm-eating warblers were observed foraging in 39 different species of trees and shrubs (19, 17, and 30 species, respectively).

Both Worm-eating and Black-and-white warblers utilized tree taxa in proportions different from relative availabilities (Fig. 1). Worm-eating Warblers used black gum ($\chi^2 = 9.3$, $df = 1$, $P = 0.002$), oaks ($\chi^2 = 34.2$, $df = 1$, $P < 0.001$), and maples ($\chi^2 = 25.7$, $df = 1$, $P < 0.001$) less frequently than relative availabilities, whereas hickories ($\chi^2 = 0.1$, $df = 1$, $P = 0.738$) were used in proportions similar to availability. For Worm-eating Warblers there was a trend towards greater use of dogwood relative to availability, but the difference was not significant ($\chi^2 = 3.3$, $df = 1$, $P = 0.068$). Black-and-white Warblers used oaks ($\chi^2 = 7.4$, $df = 1$, $P = 0.006$) more frequently than availability, whereas maples ($\chi^2 = 17.8$, $df = 1$, $P < 0.001$) and dogwood ($\chi^2 = 20.3$, $df = 1$, $P < 0.001$) were used less frequently. Black-and-white Warblers used black gum ($\chi^2 = 0.5$, $df = 1$, $P = 0.492$) and hickories ($\chi^2 = 1.5$, $df = 1$, $P = 0.214$) in proportions similar to availability.

Mean foraging heights were 8.5 m (± 4.9 SE) for Black-and-white, 4.6 m (± 3.9 SE) for Hooded, and 4.2 m (± 3.8 SE) for Worm-eating warblers. Black-and-white Warblers foraged at greater heights than either Hooded (Kruskal-Wallis $\chi^2 = 22.4$, $df = 1$, $P < 0.001$) or Worm-eating warblers (Kruskal-Wallis $\chi^2 = 57.3$, $df = 1$, $P < 0.001$). Hooded and Worm-eating warblers foraged at similar heights (Kruskal-Wallis $\chi^2 = 0.7$, $df = 1$, $P = 0.415$).

Mean tree heights used by foraging warblers were: 15.1 m (± 7.4 SE) for Black-and-white, 7.7 m (± 6.7 SE) for Hooded, and 7.1 m (± 6.0 SE) for Worm-eating warblers. Black-and-white Warblers foraged in taller trees than either Hooded (Kruskal-Wallis $\chi^2 = 30.6$, $df = 1$, $P < 0.001$) or Worm-eating warblers (Kruskal-Wallis $\chi^2 = 71.6$, $df = 1$, $P < 0.001$). Hooded and Worm-eating warblers foraged in trees of similar heights (Kruskal-Wallis $\chi^2 = 0.2$, $df = 1$, $P = 0.731$).

The three warbler species directed prey attacks toward different substrates ($\chi^2 = 245.5$, $df = 6$, $P < 0.001$). Substrates commonly used by Worm-eating Warblers included foliage (39 percent), aerial leaf

Table 1.—Mean (\pm SE) foraging height and tree height of three wood warbler species in Oak-hickory forests of the Arkansas Ozarks, May-July 1993-94.

Warbler species	Black-and-white Warbler	Hooded Warbler	Worm-eating Warbler
Foraging height ^a	8.5 (4.9) A	4.6 (3.9) B	4.2 (3.8) B
Tree height	15.1 (7.4) A	7.7 (6.7) B	7.1 (6.0) B

^aHabitat means within a row that do not share a letter were significantly different (Kruskall-Wallis test; $\pm = 0.05$).

litter (39 percent), and foliage curls (13 percent). Foraging maneuvers made by Worm-eating Warblers were 84 percent gleans, with nearly half of these being probes of aerial leaf litter and foliage curls; sallies accounted for 16 percent of maneuvers. Black-and-white Warblers primarily used bark (tree trunks, limbs, and twigs; 84 percent), and foliage (13 percent) substrates, and gleaned prey from these substrates for 74 percent of observations; sallies (18 percent) typically involved a downward chase of falling prey. Hooded Warblers used foliage substrates (88 percent) heavily while foraging, and frequently employed sally maneuvers to obtain prey from the underside of leaves; gleans were used less often (26 percent).

Discussion

Our results demonstrate heavy use of mature forest understory habitat by Worm-eating and Hooded warblers, and indirectly suggest that this habitat is a required component of their foraging ecology. In nearby areas, forest understory removal, whether or not it was done in combination with overstory cutting, was associated with low abundance of several forest understory bird species (Rodewald and Smith 1997). We believe that removal of understory vegetation likely precluded foraging of these and other understory species on managed plots. Included in the understory layer were many different species of trees and shrubs from which these birds would normally obtain their prey. Bird species may alter their foraging behavior in response to changes in vegetation structure associated with forest cutting (Szaro and Balda 1979, Maurer and Whitmore 1981, Franzreb 1983), or in response to differences in vegetation structure associated with particular tree species or forest strata (Robinson and Holmes 1982, Robinson and Holmes 1984). However, in our study the nearly complete removal of understory vegetation likely made behavioral modification impossible.

While the near absence of understory birds from plots receiving understory cutting could have been caused by loss of nesting habitat, the inability to forage in the understory of managed plots likely played an important role. For example, if understory birds avoided plots due solely to changes in nesting habitat, we expect that these birds would have foraged in midstory and canopy vegetation, but this happened very infrequently. Overstory tree removal could have been responsible for some of the differences in understory bird abundance. However, the three warblers and other understory birds (e.g. Acadian Flycatcher and Ovenbird) were common in nearby forest stands where the forest canopy had been selectively logged, but the understory was still largely intact (Rodewald and Smith 1997).

Tree Species Use and Availability

The three warblers foraged in a wide variety of plant species, but the greater diversity used by Worm-eating Warblers may have been a function of sample size. Warblers foraged at different frequencies in the different tree taxa, using some taxa at frequencies different from their relative availabilities. Most striking was the heavy use of dogwood by Worm-eating and Hooded warblers. Dogwood not only provided prey items on foliage and bark surfaces, but also in accumulated aerial leaf litter, which had fallen from overstory trees. However, nearly all dogwood trees were cut within stands receiving understory removal, making it the most heavily cut understory tree (pers. obs.). This alone may have been responsible for the near absence of Worm-eating and Hooded warblers from managed plots. Overall, 73 percent of Worm-eating Warbler foraging, 80 percent of Hooded Warbler foraging, and 38 percent of Black-and-white Warbler foraging within mature forests occurred in tree and shrub species that were extensively removed during forest management in nearby areas.

Black gum was regularly used by Hooded and Worm-eating Warblers and was the second most frequently utilized tree species by Black-and-white warblers. However, black gum was also heavily cut during understory removal (pers. obs.). Black-and-white Warblers used oaks and hickories more frequently than the other warblers, and used oaks more frequently than their availability. Because many oaks and hickories were retained during forest management, removal of those tree species from managed plots was less likely to have had a strong effect on Black-and-white Warbler abundance. The low use of maples by Black-and-white Warblers suggests a possible aversion to maples. This may have been because young and mid-aged sugar and red maples have smooth bark, providing little foraging substrate for a primarily bark-foraging species. Additionally, the long petiole of maple leaves should make foliage arthropods less accessible (Robinson and Holmes 1984) to a near-perch gleaning species. Although Black-and-white Warblers occasionally used sally maneuvers, which could allow access to more distant prey, nearly all sallying involved chasing prey that had flown or fallen from bark substrates. Interestingly, Hooded Warbler, a species that frequently used sallying to glean prey from foliage, used maples more than the other two warblers. This is consistent with Holmes and Schultz (1988) who suggested that bird species that use sally techniques to obtain prey might have greater foraging success in sugar maple.

Foraging and Tree Heights

Although all three species foraged in forest understory, Black-and-white Warblers also regularly foraged on tree trunks and limbs in the midstory and subcanopy within mature forests. This habit may have made that species less sensitive to understory removal on managed stands because many larger trees were not cut during forest treatment. Survey data are consistent with this idea because Black-and-white Warbler was the only understory-nesting species with similar abundance on both managed and mature plots (Rodewald and Smith 1997). Individual trees used by foraging Worm-eating and Hooded warblers averaged shorter in height than trees used by Black-and-white Warblers, indicating that Hooded and Worm-eating warblers are not simply foraging at low heights on tall trees. Thus, understory removal effectively eliminated foraging habitat for Hooded and Worm-eating warblers, and other understory species.

Substrate Use and Foraging Methods

Foraging substrates and maneuvers were used in different proportions by the three warbler species in mature forests. Bark substrates used heavily by Black-and-white Warblers were primarily those of larger trees, which may explain why this species was occasionally found on managed forest plots where other understory species were very infrequently recorded (Rodewald and Smith 1997). Hooded and Worm-eating warblers frequently used live foliage substrates while foraging in the understory. Worm-eating Warblers utilized aerial leaf litter heavily for foraging (39 percent) in mature forests. Aerial leaf litter provides shelter for a variety of different arthropods and accounted for 58 percent-84 percent of foraging maneuvers by Worm-eating Warblers in non-breeding areas in Central America and the Caribbean (Greenberg 1987). However, aerial leaf litter accounted for only 11 percent of foraging during breeding periods in Maryland (Greenberg 1987), suggesting a higher importance of this foraging substrate in the Ozark mountains. Understory foliage and aerial leaf litter were removed from managed plots and this significantly altered substrate availability for understory birds.

Management Implications

Forests in this area are primarily managed for oak-hickory, and understory removal is believed to be important in lowering competition for regenerating oak and hickory. In addition to strongly affecting nesting habitat, understory removal has, at least in the short-term, excluded these and other forest understory birds from foraging in managed plots. As a result, forest managers should consider reducing, eliminating, or altering the pattern of removal of understory vegetation (e.g. retaining patches of understory) in forest management plans as this could lessen negative effects on mature forest birds. Experiments that examine effects of removing varying amounts of understory on both birds and forest regeneration would be useful. Acceptable regeneration of oak and hickory might be obtained under uneven-aged systems by cutting smaller percentages of understory tree and shrub species important for forest birds. If this is possible, some understory bird species may then be able to utilize

these managed forests for foraging and possibly nesting. However, even if adequate regeneration of marketable tree species were achieved, additional research on the long-term effects of this uneven-aged management technique on bird abundance and reproductive success would be necessary.

Acknowledgments

Financial support was provided by the Southern Research Station of the U.S. Forest Service. We thank Forest Service personnel J.B. Andre, D.L. Graney, P.A. Murphy, J.S. Osborne, and J.S. Self for project assistance. A.D. Rodewald, G. Gale, D.A. James, and several anonymous reviewers made useful comments on the manuscript. J.A. Baker and R.W. McNew provided statistical advice. J. Bider, R. King, K.M. Lohraff, and L. Pomara assisted with vegetation analysis and/or data entry.

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THE MISSOURI OZARK FOREST ECOSYSTEM PROJECT: FINDINGS FROM TEN YEARS OF EVALUATING MANAGEMENT EFFECTS ON FOREST SYSTEMS

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ABSTRACT.—In 1989, the Missouri Department of Conservation initiated the Missouri Ozark Forest Ecosystem Project, 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. Here we report the ten-year findings. Pre-treatment data were collected from 1991-1995 and the first harvest entry occurred from 1996 to 1997. Post-treatment results presented here are from 1997-2000. Relative to the no-harvest 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, except American toad abundance declined on all treatments with the steepest declines observed on no-harvest 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. Overall, harvest treatments have changed the faunal communities at landscape scales and even-aged treatments had the greatest effect. However, harvesting was not necessarily detrimental to plant and animal communities. Overall, forest management objectives, including regeneration, do not appear to conflict with other management objectives such as sustaining diverse forest overstories, ground flora, and wildlife communities.

Forest management is becoming more controversial and is increasingly under scrutiny. Often, the general public, environmental groups, and land managers disagree about forest management methods that are acceptable or suitable for managing public forest land. In particular, considerable controversy surrounds forest harvesting. Some advocate selection harvesting because they believe it is less detrimental to wildlife habitat and ecosystem function than clearcutting, while others believe that any kind of harvesting is harmful to forest habitats. Many of these beliefs are rooted in perception rather than grounded in scientific evidence. State and Federal forest management agencies have a responsibility to base forest management decision making on the best scientific evidence available.

The Missouri Ozark Forest Ecosystem Project (MOFEP) was initiated in 1989 by the Missouri Department of Conservation (MDC) to generate the best scientific evidence about the effects of even-aged, uneven-aged, and no-harvest management systems on the flora and fauna of upland oak ecosystems (Brookshire and others 1997). It is a long-term study, designed to extend for a least one 100-year rotation from the first harvest entry in 1996. It is also landscape scale with forest management compartments, each approximately 1000 acres in size, as observational units. It is also a fully replicated and designed experiment. It was initiated because the impacts of forest management on

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songbirds and other non-commodity forest attributes such as diversity of native plant and animal species have been poorly quantified in the Missouri Ozarks. The purpose of this paper is to summarize what has been learned through the first 10 years of study at MOFEP.

The MOFEP Study Sites and Treatments

MOFEP was designed to experimentally study the effects of forest management practices on the entire forest over entire rotations rather than simply in research plots immediately following harvesting. Therefore, the management practices selected for comparison, and the scale and timeframe of the study reflect those commonly used to operationally manage forests in the Ozark Highlands. MOFEP consists of nine forest sites ranging in size from 772 to 1, 271 acres, primarily within the Current River Oak Forest Breaks and the Current River Oak-Pine Woodland Hills landtype associations of the Ozark Highlands (fig. 1). These sites were selected because they contained mature, second-growth forests largely free of manipulation for > 40 years, were the size of administrative compartments commonly used by the Missouri Department of Conservation for managing forests, are owned by the Missouri Department of Conservation, and are in close proximity to each other. More detailed descriptions of the MOFEP sites were presented by Meinert and others (1997) and Kabrick and others (2000).

Each site was divided into areas having common slope and aspect and then divided into stands which averaged 12 acres in size (Brookshire and others 1997). A detailed landscape-scale soil mapping and ecological classification typing project was conducted on MOFEP in 1994-1995 (Kabrick and others 2000, Meinert and others 1997). This included characterizing important physical and vegetation characteristics of each study site.

Groups of three sites were allocated into “blocks” by spatial proximity (fig. 1). The three sites within each block were randomly assigned one of three treatments: (1) even-aged management with harvesting by clearcutting and intermediate thinning, (2) uneven-aged management with harvesting by single-tree selection and group selection, and (3) no-harvest management. Sheriff and He (1997) provided a more detailed description of blocking and treatment allocation to sites. Even-aged management has been practiced by MDC managers for at least three decades and uses clearcutting as the principal means of stand regeneration. With this method approximately 10 percent of the acreage in a forest compartment (i.e., a MOFEP site) was designated as “old growth” and will be excluded from future harvesting. About 10 to 15 percent of the remaining area is clearcut during each re-entry for forest regeneration. Thinnings (intermediate cuttings) are conducted periodically within stands to improve quality and increase growing space for residual trees. Rotation lengths are approximately 100 years with a 15-year re-entry.

Uneven-aged management, as practiced by MDC managers, is relatively new to upland oak ecosystems in the Ozarks. Uneven-aged management is commonly practiced in bottomland forests in Missouri and in mixed hardwood forests elsewhere where shade-tolerant species are prevalent, competitive and desirable. Forest management on the Pioneer Forest, a privately-owned forest in the Ozarks, suggests that uneven-aged management may be a viable silvicultural alternative in Missouri’s upland forests where soils and climate favor oak species and limit competition by undesirable species (Loewenstein 1996). Following the guidelines developed by Law and Lorimer (1989), uneven-aged management in MOFEP included single-tree selection and group selection for timber harvest and forest regeneration. Just like with even-aged management, approximately 10 percent of the forest area was designated as “old growth.” The remaining area was grouped into management units of 20 to 80 acres. Within each management unit, harvest objectives are set for the largest diameter tree (LDT), residual basal area (RBA), and q-value. The overall RBA is equivalent to B-level stocking with adjustments made for logging damage (Roach and Gingrich, 1968). Target q-value objectives average 1.5 but can range from 1.3 to 1.7 (Law and Lorimer 1989). Within harvested areas, group selection openings are also created to regenerate shade-intolerant species. These groups are approximately one to two tree heights in diameter, depending on aspect. At MOFEP, they are 70 feet (0.09 acres) on south-facing slopes, 105 feet (0.20 acres) on ridge tops, and 140 feet (0.35 acres) on north-facing slopes. These openings are to sum to about 5 percent of the total harvested area. For all harvesting, re-entries coincide with those of

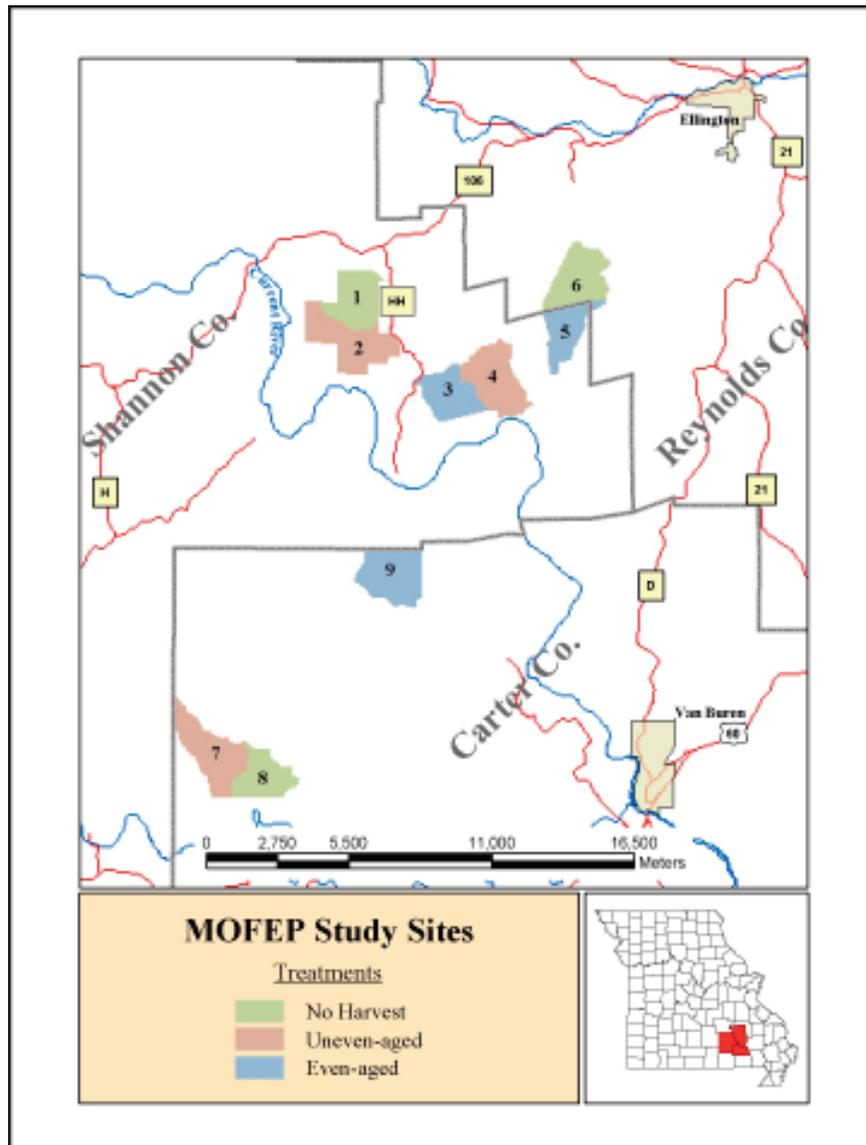


Figure 1.—Location of the nine MOFEP experimental sites (compartments) located in southeastern Missouri. Sites were allocated into blocks for statistical analyses by spatial proximity (sites 1 through 3 are in block 1, sites 4 through 6 are in block 2, and sites 7 through 9 are in block 3). Treatments (even-aged management, uneven-aged management, and no-harvest management) were randomly assigned within each block.

even-aged treatments. On uneven-aged sites, the LDT, RBA, and q-values were selected to create similar diameter distributions on a forest-wide basis as under even-aged management (see Brookshire and others, 1997).

The no-harvest management treatment will not be harvested. Wildfires are suppressed and natural events such as tornadoes, fires, insect and disease outbreaks are treated the same as on any other forest land owned by the MDC, except that salvage harvests will not occur. This treatment serves two purposes. It demonstrates patterns of forest development that result from natural disturbances and successional processes, and it also serves as an experimental “control” to compare with the two other management practices.

Methods

MOFEP comprises more than 28 different studies to more completely quantify the effects of forest management on the flora and fauna of Ozark forest ecosystems (Brookshire and Shifley 1997, Shifley and Brookshire 2000, Shifley and Kabrick 2002). Studies included soil characteristics and distribution, below- and above-ground 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, herpetofaunal communities and distribution, small mammals abundances, leaf litter arthropod communities, and abundances of leaf-chewing insects. All projects on MOFEP share a common, randomized complete block design as the basis for statistical analyses; by necessity, each study has its own sampling and analysis protocols to adequately characterize the wide array of flora and fauna found at the study sites (Sheriff and He 1997, Sheriff 2002). MOFEP is a long-term project designed to extend beyond a single rotation. As such, flora and fauna are inventoried on a regular basis before and after harvest entries. In this paper we report on pre- and post-treatment differences as a consequence of the first harvest entry implemented from May 1996 to May 1997. Here we focus on the response of forest vegetation, songbirds, reptiles and amphibians, and small mammals. Specific data collection and analysis methods used in this paper are described below.

Vegetation

Vegetation was sampled in 648 permanent 0.5-acre plots distributed approximately equally among the nine MOFEP sites. At least one plot was established in each stand on all sites. This paper includes overstory data collected during inventories conducted in 1994-1995 (pre-treatment) and 1997-1998 (post-treatment). Within permanent plots, live and dead trees ≥ 4.5 inches DBH were sampled in 0.5-acre circular plots; trees between 1.5 and 4.5 inches DBH were sampled in four 0.05-acre circular subplots; trees at least 3.3 feet tall and less than 1.5 inch DBH were sampled in four, 0.01-acre circular subplots nested within the 0.05-acre subplots. Characteristics recorded for each tree included species, DBH, or size class for trees < 1.5 inches DBH and > 3.3 feet tall, status (e.g., live, dead, den, cut, blow-down), and crown class (e.g., dominant, codominant, intermediate, suppressed) (Jensen 2000). Plot and subplot data were combined to obtain plot averages by DBH or size class and all values are converted to an acre basis. Ground flora data (e.g., species, foliar coverage, and number of stems for woody seedlings ≤ 3.3 feet tall) were sampled in sixteen, permanently-marked, 3.3-foot by 3.3-foot quadrats located within 0.5-acre vegetation plots (Grabner 2000). Ground flora data included in this paper were collected from June 1 through August 25 in 1994 and 1995 (pre-treatment) and again in 1997 and 1998 (post-treatment).

Birds

Five bird species associated with mature forest—Acadian Flycatcher (*Empidonax virescens*), Wood Thrush (*Hylocichla mustelina*), Ovenbird (*Seiurus aurocapillus*), Worm-eating Warbler (*Helminthos vermivorus*), and Kentucky Warbler (*Oporornis formosus*)—and six early successional species—Indigo Bunting (*Passerina cyanea*), Yellow-breasted Chat (*Icteria virens*), Hooded Warbler (*Wilsonia citrina*), Prairie Warbler (*Dendroica discolor*), Blue-winged Warbler (*Dendroica pinus*), and White-eyed Vireo (*Vireo griseus*)—were focal species of the study. Species densities are collected using spot-mapping and reproductive data are collected by locating and monitoring nests. Spot mapping data were collected in seven, 110-acre spot-mapping plots per site and was done eight to ten times at 2- to 3- working day intervals from mid May through the end of June each year, except for the treatment year, since 1991. Each day, a single map was produced for each spot-map plot. Data from these spot maps were used to develop estimates of songbird territory densities by overlaying spot maps and creating composite territory maps for each species. To locate territories, we identified clusters of three or more observations of the same individual. Daily survival rates for nests located within spot mapping plots were estimated using methods of Mayfield (1961 and 1975). Nests were monitored every three to five days until nest fate was determined (Clawson and others 1997, Clawson and others 2002). In this paper, we include data collected from 1991 to 2000.

Amphibians and Reptiles

Amphibians and reptiles were sampled in twelve, randomly-located drift fence arrays (modified from Jones, 1981) on north- and east-facing, and south- and west-facing slopes within each site (Renken and Fantz 2002). Amphibians and reptiles were trapped in March – June and September – October in 1992-1995 (pre-treatment) and 1998-2000, and September – October 1997-2000 and March-June 1998-2000 (post-treatment). North- and east-facing and south- and west-facing slopes were sampled because they comprised 73 percent of the landscape on MOFEP.

Small Mammals

Small mammals were trapped using baited Sherman live traps on 2, 18.7-acre randomly-located grids on north- and east-facing slopes on each site (Fantz and Renken 2002). Each grid had 144 traps evenly spaced 25 meters apart. Animals were trapped for six consecutive nights on each site during April – May in 1994-1995 (pre-treatment) and 1998-2000 (post-treatment).

Analyses

To analyze harvesting effects on forest vegetation, we evaluated pre- and post-harvest differences using ANOVA. Treatment (even-aged, uneven-aged, and no-harvest management) and block were the main effects with an alpha = 0.05 (n = 9 sites). For trees, we evaluated changes in richness (per plot), trees per acre, quadratic mean diameter, basal area, and percent canopy cover. Where differences were found, we used a least significant difference test to identify attributes that differed from those of no-harvest sites. For ground flora, we evaluated pre- and post-harvest differences in percent cover of species groups (annuals/biennials, forbs, graminoids, legumes, shrubs, woody vines) as well as overall species richness (per plot) and percent ground cover (Kabrick and others 2002, Grabner and Zenner 2002).

To analyze songbird density during pre-treatment years (1991-1995), we used multivariate repeated-measures ANOVA. Year effects were not significant, so we used pre-treatment mean density as a covariate in the analysis of post-treatment data. To evaluate harvest treatment effects, we used a multivariate repeated-measures analysis of covariance with treatment and block as main effects and pre-treatment density as the covariate with an alpha = 0.1. We used this alpha level because power was low (n = 9 sites) in the experiment (Sheriff and He 1997). Contrasts were used to compare even-aged and uneven-aged treatments to the no-harvest (control) treatment (Gram and others 2003).

To analyze the first-entry harvest effects on amphibians, reptiles, and small mammal abundance, annual post-treatment (years 1998-2000) abundance estimates for overall small mammal abundance, overall amphibian and reptile abundance, and for thirteen focal species of amphibians and reptiles [Spotted Salamander (*Ambystoma maculatum*), American Toad (*Bufo americanus*), Common Five-lined Skink (*Eumeces fasciatus*), Broad-headed Skink (*Eumeces laticeps*), Central Newt (*Notophthalmus viridescens*), Western Slimy Salamander (*Plethodon albagula*), Southern Red-backed Salamander (*Plethodon serratus*), Northern Spring Peeper (*Pseudacris crucifer*), Green Frog (*Rana clamitans*), Little Brown Skink (*Scincella lateralis*), Northern Fence Lizard (*Sceloporus undulatus*), Northern Red-bellied Snake (*Storeria occipitomaculata*), Smooth Earthsnake (*Virginia valerae*)] for 1998, 1999, and 2000 were subtracted from the mean pre-treatment abundance estimates for each site (Fantz and Renken 2002, Renken and Fantz 2002, Renken and others in press). These difference scores were the dependent variables in randomized complete block ANOVAs (Fantz and Renken 2002, Renken and Fantz 2002) and split-plot repeated measures ANOVA (Renken and others in press) models used to detect treatment effects. Treatment and block were main effects and the treatment x block interaction was the error term used to test main effects. An alpha of 0.10 was used for tests because power was low (n=9 sites) in the experiment (Sheriff and He 1997). In Renken and others (in press), tests of main effects were followed with contrasts (with an alpha of 0.03) to test differences among treatments (Renken and others in press). Qualitative comparisons were also made to examine the effect of treatment upon the species composition of the amphibian, reptile, and small mammal communities.

Results and Discussion

During the first harvest entry, 2.4 million board feet of timber were harvested on the three even-aged sites (3,360 board feet per harvested acre or 876 board feet per site acre) and 3.4 million board feet were harvested on the three uneven-aged sites (1,620 board feet per harvested acre or 932 board feet per site acre). On even-aged sites, 11 percent of the area was clearcut and 15 percent was thinned. On uneven-aged sites, 57 percent of the area was harvested with selection and group methods. Regardless of harvest method, black oak (*Quercus velutina* Lam.) and scarlet oak (*Q. coccinea* Muenchh.) in combination comprised 60 percent of the harvested basal area; white oak (*Q. alba* L.) and post oak (*Q. stellata* Wangenh.) accounted for an additional 20 to 30 percent. On a percentage basis, harvested trees included more scarlet and black oak basal area and less white oak and shortleaf pine (*Pinus echinata* Mill.) basal area than the sites had prior to harvest. The treatments significantly reduced the mean number of trees > 1.5 inches DBH per acre ($P=0.05$), basal area per acre ($P=0.01$), and percent canopy cover ($P=0.01$) on the harvested sites, but mean diameter was unchanged. Following treatment, the relative size distribution of trees by diameter class was virtually identical for each of the three treatments. On no-harvest sites, the total basal area increased an average of 1 ft²/ac between 1995 and 1998. Following treatment, there was virtually no change in the density of trees in the reproduction size classes (taller than 3.3 feet and smaller than 1.5 inches DBH) except for the number of stump sprouts. More than 700 stump sprouts per acre occurred in clearcuts; fewer than 120/ac occurred in areas harvested by a combination of single tree and group selection. Stands that were not harvested averaged fewer than 7 sprouts per acre (Kabrick and others 2002).

Harvesting affected ground flora species composition at the site scale and most of the effects occurred directly within treated stands. The increased light levels caused by harvesting increased the mean species richness on harvested sites. However, richness unexpectedly decreased on no-harvest sites ($P<0.01$). We do not know if this decrease in no-harvest sites was due to sampling error or to some other phenomenon such as the drought that occurred throughout the late 1990's. We did find that total percent ground cover increased on all sites and increased most on harvested sites ($P<0.01$). In both even-aged and uneven-aged sites, annual and biennial species, which were essentially absent prior to harvesting, increased in mean relative cover after treatment ($P=0.02$), particularly in clearcuts and group selection openings. Woody vines such as summer grape (*Vitis aestivalis*) and early-successional shrubs such as blackberries (*Rubus pensilvanicus*) also increased ($P=0.01$), but primarily in clearcuts and group selection openings. Legumes such as common tick trefoil (*Desmodium nudiflorum*) and hog peanut (*Amphicarpa bracteata*) decreased ($P<0.01$) in harvested sites, most likely because the increased light favored other ground flora species (Grabner and Zenner 2002).

The mature forest bird species present during both the pre-treatment and post-treatment years of the study declined following treatment, even on no-harvest sites (figs. 2 and 3). Some early-successional bird species did not appear until after tree harvest. In post treatment years 1997-2000, treatment effects were found. They were: Ovenbird densities were lower ($P=0.03$) on even-aged sites than on no-harvest sites; Wood Thrush ($P=0.07$), Prairie Warbler ($P<0.01$), and White-eyed Vireo ($P=0.04$) densities were higher on even-aged sites than no-harvest sites; Kentucky Warbler ($P<0.1$), Indigo Bunting ($P<0.01$), and Yellow-breasted Chat ($P<0.08$) densities were higher on both even-aged and uneven-aged sites than no-harvest sites. No significant treatment effects were found for reproductive success: daily nest survival rates did not change significantly from pre- to post-treatment; brood parasitism rates were low (Gram and others, 2003), averaging 3.2 percent in both the pre- and post-treatment periods. In general, the forest management treatments affected bird species densities and each species had species-specific responses to even-aged and uneven-aged management. Although early successional bird species increased on the MOFEP sites, some used both small openings (e.g., group selection openings and clearcuts < 10 acres) and large openings (e.g., clearcuts > 10 acres) for nesting and some used only the large openings. Yellow-breasted Chats and Prairie Warblers rarely used openings that were smaller than 10 acres. Other early succession species such as Indigo Bunting, Hooded Warbler, White-eyed Vireo, and Blue-winged Warbler used both small and large openings. The bird community is dynamic and likely will continue to change through time, in composition and density, in response to harvest and re-growth of the forest (Clawson and others 2002, Gram and others 2003).

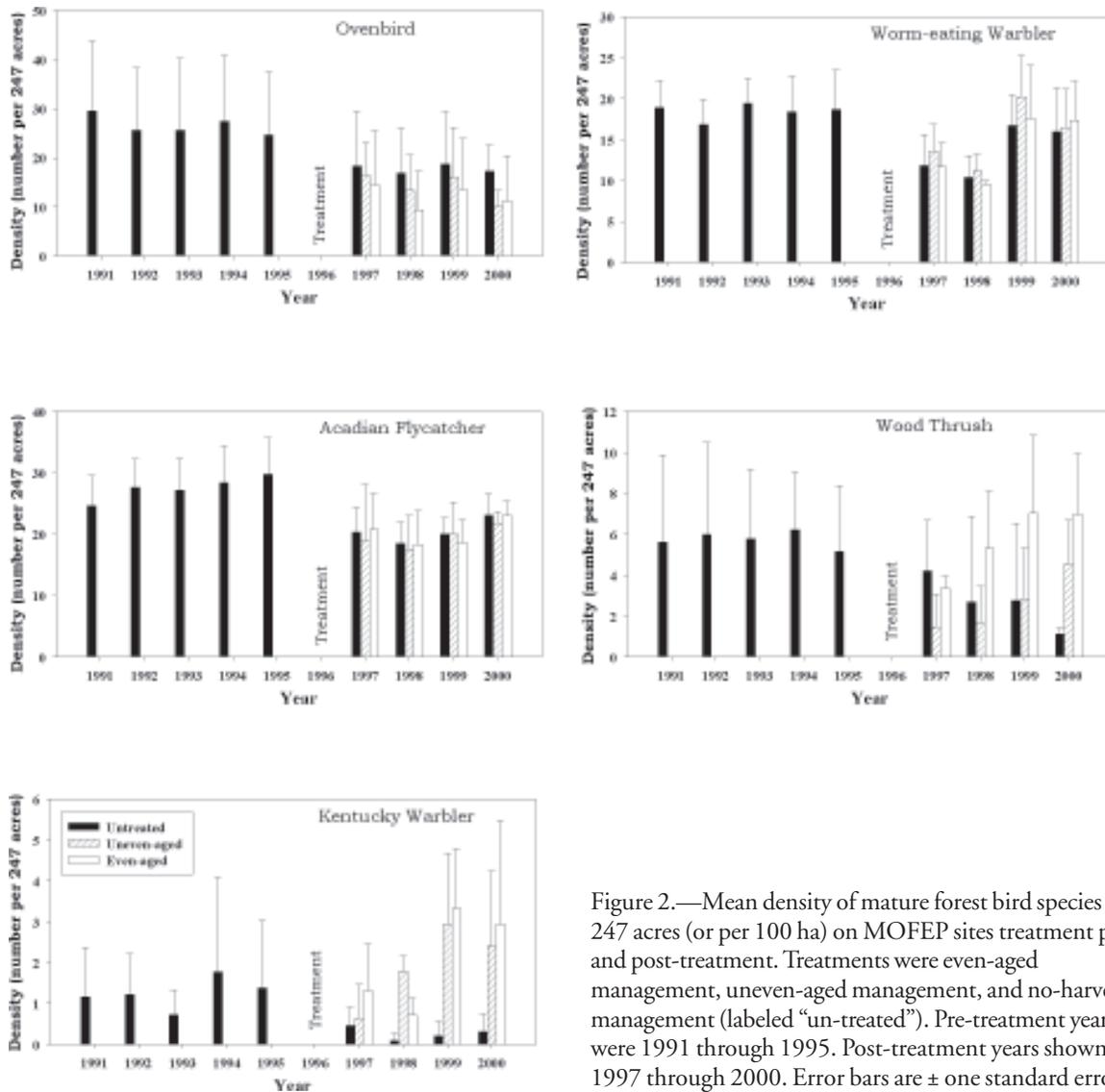


Figure 2.—Mean density of mature forest bird species per 247 acres (or per 100 ha) on MOFEP sites treatment pre- and post-treatment. Treatments were even-aged management, uneven-aged management, and no-harvest management (labeled “un-treated”). Pre-treatment years were 1991 through 1995. Post-treatment years shown were 1997 through 2000. Error bars are \pm one standard error.

Prior to harvesting, eight species of small mammals and 43 species of amphibians and reptiles were captured on MOFEP sites. Following harvests, no species disappeared and no new species appeared. Harvest treatments did not affect overall amphibian and reptile abundance and the abundances of twelve of thirteen focal amphibians and reptiles (Fantz and Renken 2002, Renken and Fantz 2002, Renken and others in press, fig. 4). After treatment, small mammal abundance on even-aged sites remained the same but declined slightly on uneven-aged sites and declined substantially on no-harvest sites (fig. 5). American toad abundance followed a similar pattern (fig. 6). The decline of small mammals and American toads on no-harvest sites suggests there was a natural decline in some animal populations within the region, perhaps associated with a regional drought during the years immediately following the first entry harvest (Renken and others in press). Even though some animal populations declined on no-harvest sites, conditions on even-aged, and to a certain extent on uneven-aged sites, buffered or dampened the decline the populations would have experienced. Even-aged sites may have had more invertebrate and seed food resources (Harper and Guynn 1999, Hooven 1973, Perry and others 1999), and more cover from predators than existed on no-harvest sites during the immediate post-treatment period.

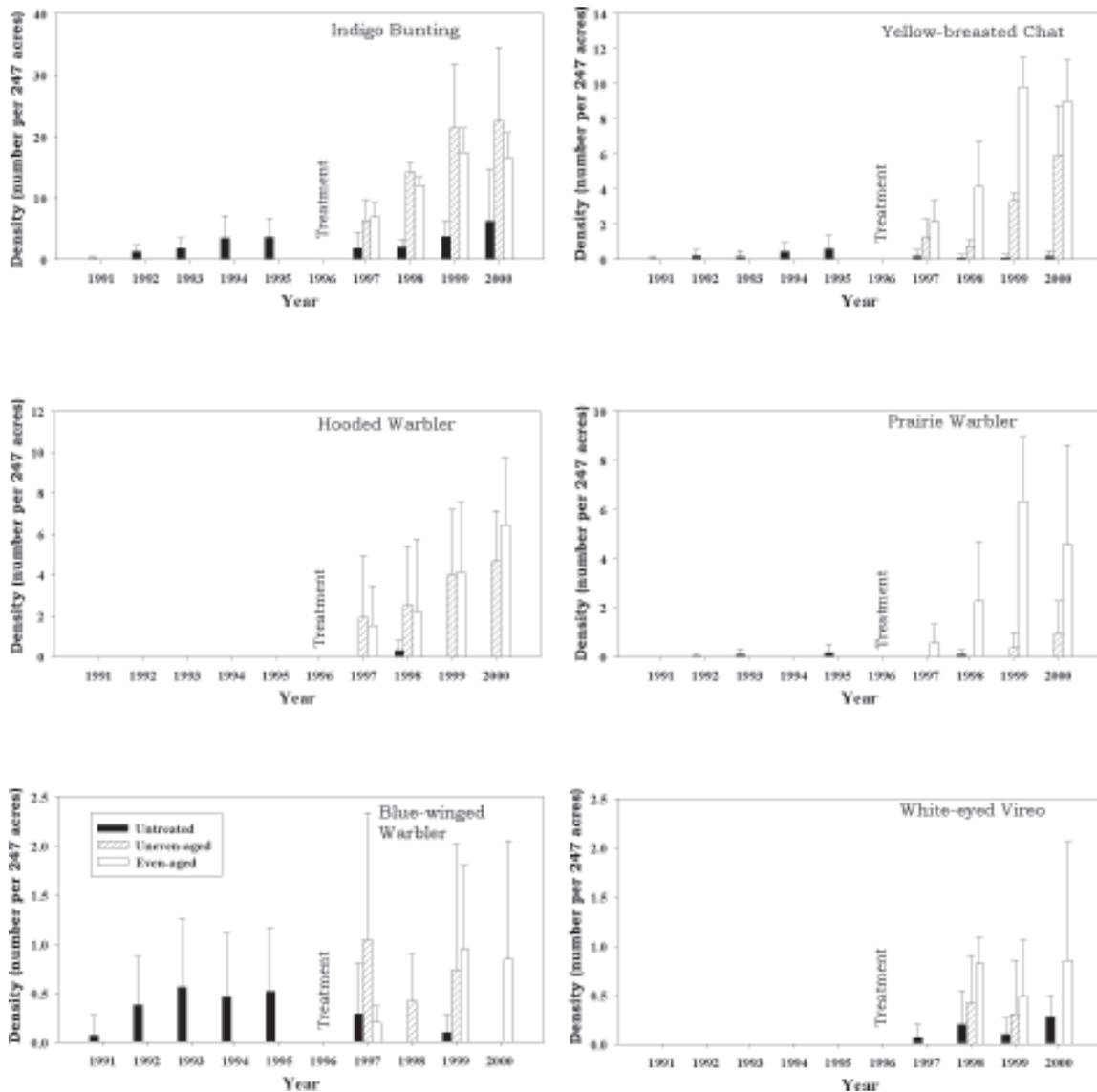


Figure 3.—Mean density of early-successional bird species per 247 acres (or per 100 ha) on MOFEP sites pre- and post-treatment. Treatments were even-aged management, uneven-aged management, and no-harvest management (labeled “un-treated”). Pre-treatment years were 1991 through 1995. Post-treatment years shown were 1997 through 2000. Error bars are \pm one standard error.

Implications for Forest Management

Forest harvesting clearly affected the floral and faunal communities and, in general, the more intensive the harvesting, the greater the observed effects (e.g., effects of clearcut harvest > selection and group harvest). However, the effects of harvesting were not necessarily detrimental to the flora and fauna and in many cases were favorable. For example, relative to no-harvest sites, ground flora richness increased after harvesting largely because of the increased light reaching the forest floor and/or perhaps because of soil disturbance caused by harvesting favored the establishment of early-successional species. The numbers of many early-successional songbird species such as Indigo Buntings, Yellow-breasted Chats, and Prairie Warblers, have also responded to habitat created by the harvest treatments and have increased. Mature forest bird species such as Ovenbirds and Worm-eating Warblers were of considerable concern when MOFEP was initiated (Clawson and others 1997, Clawson and others 2002). However, mature forest bird species remain dominant and an important component of the species composition at MOFEP. For songbirds in particular, one of the greatest concerns was that

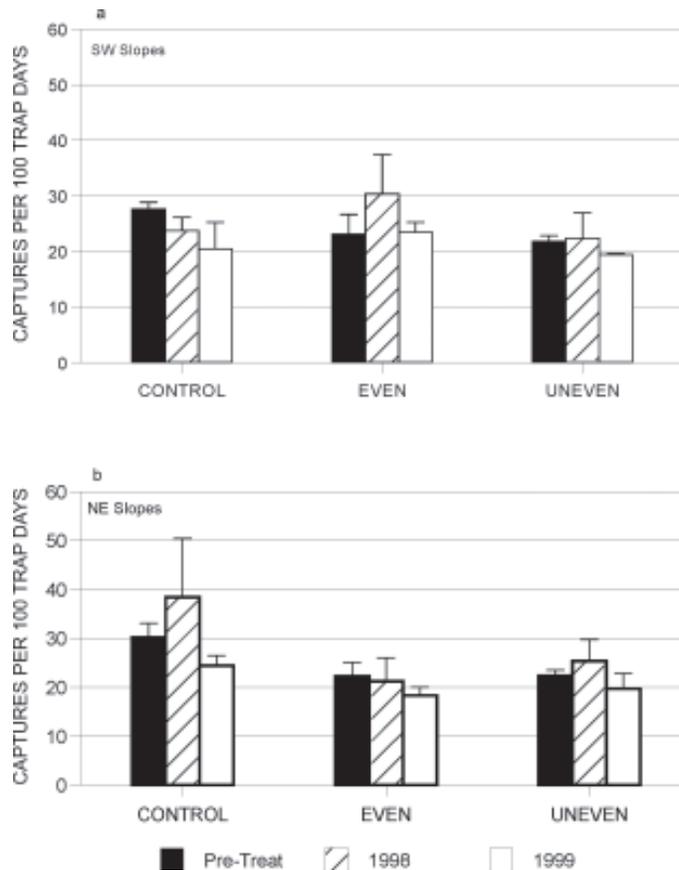


Figure 4.—Mean relative abundances of amphibian and reptiles by treatment type on southwest-facing (a) and northeast-facing (b) slopes on MOFEP sites pre- and post-treatment. Treatments were even-aged management, uneven-aged management, and no-harvest management (labeled “control”). Pre-treatment years were 1992 through 1995; data were combined. Post-treatment years shown were 1998 and 1999. A trap day is defined as a day-long period during which traps were operated. Error bars are \pm one standard error.

harvesting would cause an increase nest parasitism, which reportedly increases with increasing forest fragmentation (Gibbs and Faaborg 1990, O’Conner and Faaborg 1993, Donovan and others 1995). However, we found neither nest predation nor nest parasitism increased following harvest treatments.

One surprising finding from MOFEP is that on no-harvest sites, plant species richness and the abundances of small mammals, reptiles, and amphibians have decreased while these same populations remained at or above pre-treatment levels on harvested sites (figs. 2 through 6). We do not fully understand the reasons for these declines on no-harvest sites and cannot rule out the possibility that treatments are affecting some of the animal populations on nearby no-harvest sites. Sampling error is another possible but unlikely explanation because the declines occurred with different animal and plant species. Gram and others (2003) and Renken and others (in press) speculated that wide-spread drought was partially to blame for some of these declines. For now, we cannot explain these declines and can only acknowledge the cyclical nature of many plant and animal populations.

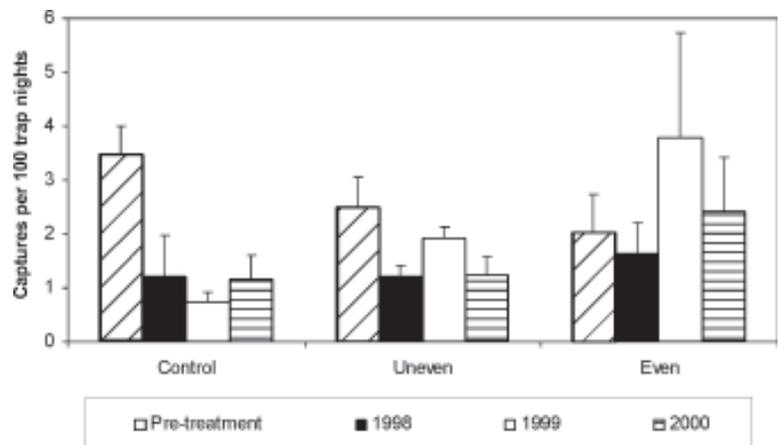


Figure 5.—Mean relative abundances of small mammals by treatment type on northeast-facing slopes on MOFEP sites pre- and post-harvest. Treatments were even-aged management, uneven-aged management, and no-harvest management (labeled “control”). Pre-treatment data were collected in 1994 and 1995; data were combined. Post-treatment years were 1998 through 2000. One trap night is defined as a night-long period during which traps were operated. Error bars are \pm one standard error.

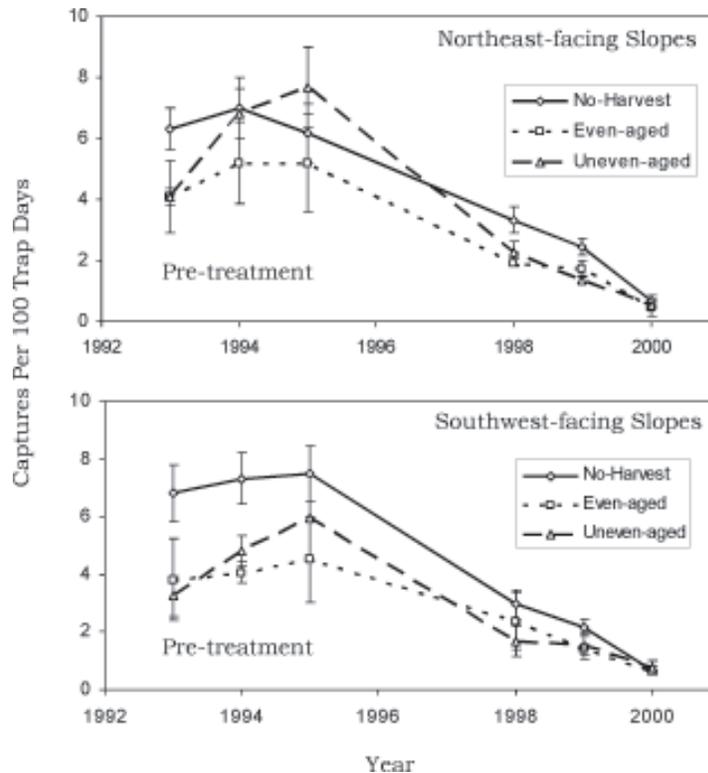


Figure 6.—Mean relative abundances of American Toads (*Bufo americanus*) on southwest-facing and northeast-facing slopes on MOFEP sites by treatment pre- and post-harvest. Treatments were even-aged management, uneven-aged management, and no-harvest management. Pre-treatment data shown were collected in 1993 through 1995. Post-treatment data shown were collected in 1998 through 2000. Error bars are \pm one standard error.

Continued research through MOFEP will likely reveal explanations for these observed trends. It demonstrates that a larger-scale phenomenon may be affecting plant and animal populations in this ecosystem and shows the value of having control sites (in our case, no-harvest sites) for evaluating management effects on forest ecosystems.

Overall, it appears that forest management systems commonly used throughout the Central Hardwood Region are not negatively affecting most of the plant and animal populations discussed in this paper at landscape scales in the Missouri Ozarks. However, it is important to point out that these are only the ten-year findings from a study designed to extend for at least 300 years and it will require at least 100 years (i.e., one full rotation) before even-aged sites are fully regulated. It remains unknown if the trends observed to date will continue with repeated harvest entries. Moreover, as MOFEP becomes more fully integrated, other system components such as long term productivity, below-ground biodiversity, and long-term forest health issues will be included in analyses to develop a more comprehensive understanding of the effects of management on forest ecosystems.

Acknowledgments

We thank the hundreds of interns, technicians, and research scientists who have help gather and analyzed data for MOFEP during the past ten years. We also thank the Missouri Department of Conservation, the University of Missouri-Columbia, the University of Missouri-St. Louis, the National Fish and Wildlife Foundation, the USGS Biological Resources Division, and the USDA Forest Service North Central Research Station for partial support for MOFEP. This research was also funded in part by the Federal Aid in Wildlife Restoration Act, under Pittman-Robertson Project W-13-R.

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DIRECT SEEDING OF HARDWOODS IN NORTHEAST IOWA AND SOUTHEASTERN MINNESOTA

Gary Beyer, Bruce Blair, Jim Edgar, Lance Sorensen, and John Olds[†]

Direct seeding of hardwoods has been done in northeast Iowa since 1883 and in Southeast Minnesota since 1999. This method of reforestation has been completed on Conservation Reserve Program acres and on other non-forested lands including both abandoned pasture and cropland acres. Direct seeding is an alternative to traditional seedling planting for establishment of tree cover on open ground. There are several advantages over seedling planting when an area is direct seeded. These include: 1. Rapid conversion of open sites to forest cover. 2. Straight, dense, naturally pruned trees. 3. Decreased length of time that is needed to control competing weeds, noxious weeds and rodent pests, such as pocket gophers. 4. Reduced impact from white-tailed deer. The benefits of direct seeding are not always achieved. Some of the problems that have been identified are 1. Weed and competing vegetation control 2. Poor seed germination and survival due to any of several factors that include; poor cultural practice, herbicide deficiency, poor seed quality, rodent, bird and squirrel predation. The poster shows photos of direct seeding projects at various stages of growth from open fields to successfully seeded young hardwood forest. Regeneration surveys show the wide range of successes and attempts to identify reasons for good and bad results. Heavy weed growth can nearly eliminate some species such as oak. Other species such as walnut and ash have a greater ability to grow through the competition and become established. Test plots have evaluated cultural practices, herbicide effectiveness for weed control and decreased seed germination. Cover crops have been evaluated for their ability to reduce soil erosion and competing vegetative growth. As in most private land (and public land) projects, the ability of the land owner to do the weed control and other follow up work is critical to the success of the project. A landowner who is determined to stay committed to the project for the 3-5 years needed for establishment, will likely be rewarded with a dense, diverse young stand of hardwoods.

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PATTERNS OF POST-GLACIAL RECOLONIZATION OF NORTHERN RED OAK (*QUERCUS RUBRA* L.)

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Quercus rubra is a dominant species in the forests of east central North America and is a source of valuable hardwood. The purpose of our research is to detect the impact of postglacial recolonization patterns on contemporary population substructure. We hypothesize that the species has undergone two reductions of effective population size: a natural reduction about 18,000 years ago, during the Wisconsinian glaciation and an anthropogenic reduction during European colonization. After the glacier retreated, trees surviving in southern and western refugia recolonized the northeastern United States and Canada. We hypothesize that during this recolonization process a loss of organelle genetic diversity occurred due to genetic drift. Thus, we expect to see genetic diversity diminish as a function of distance north from the southernmost advance of the Wisconsinian ice sheet. An alternative hypothesis is that genetic diversity will peak in the middle of the contemporary range as a result of mixing two refugial source populations: a population west of the Mississippi River and a population south of the Ohio River. We are using PCR-RFLP of three intergenic regions in oak chloroplast DNA sequences to test this hypothesis. The maternally inherited chloroplast genome will retain the signal of postglacial migrations even after considerable contemporary disturbance. For our preliminary survey we chose sites with histories of minimal human disturbance within the last one hundred years. We have detected only three haplotypes north of the glaciation line (fig. 1). Most Northern sites had either haplotype I or haplotype II. Haplotype V was locally abundant at one Indiana site and occurred at four other sites north of the glaciation line. We detected only one haplotype in each of the two northernmost sites in Wisconsin (15 trees/site). These data support the first hypothesis.

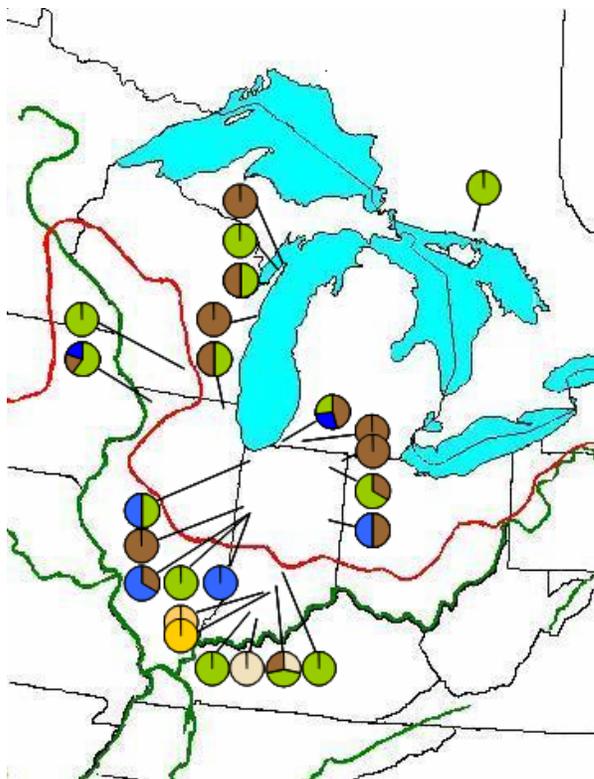


Figure 1.—Geographic distribution and frequencies of haplotypes at each site. Haplotypes I, II, III, IV and V are represented as brown, green, beige, yellow and blue, respectively. The maximum advance of Wisconsinian ice sheet indicated by the red line.

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ECOSYSTEM RESTORATION AND WILDFIRE MANAGEMENT TREATMENTS AFFECT SOIL ORGANIC MATTER AND MICROBIAL ACTIVITY IN FOUR CONTRASTING FORESTS

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Jennifer A. Brinkman, and Annemarie Smith†*

As part of a national-scale evaluation of the consequences of restoration and wildfire fuel reduction treatments in ecosystems that historically had frequent fire (www.ffs.fs.fed.us), we determined the effects of reintroduction of dormant season fire (functional restoration) and thinning from below (structural restoration) on soil organic matter characteristics and microbial activity in two mixed oak forests in the central hardwoods region, and contrasted them with the effects of the same treatments in two conifer forests. The hardwood forests were located on the Allegheny Plateau of southern Ohio (Ohio Hills/OH) and in the southern Appalachian Mountains of North Carolina (Green River/NC). The conifer forests were a pine-oak site in the piedmont of South Carolina (Clemson Forest/SC) and a pine-fir site in the southern Cascades of northern California (Goosenest Adaptive Management Area/CA).

At OH and SC fire resulted in reduced soil organic C and total soil N, both alone and in combination with thinning. In contrast, thinning alone did not affect either C or N at OH, SC, or CA. Soil C:N ratio decreased in the plots that were burned at OH and SC, but not in those that were thinned or thinned+burned; at CA soil C:N ratio decreased in response to thinning. Fire, thinning, and the combination of the two resulted in reductions in acid phosphatase activity in soils at OH and SC, whereas at CA only the combination of fire and thinning reduced this metric of overall microbial activity. Chitinase activity was reduced by all three treatments at OH and SC. At CA, chitinase activity decreased in the order Control>Thin>Thin+Burn. Phenol oxidase activity changed little, and did not differ among treatment types at OH, SC, or CA.

Multiple regression models predict that posttreatment enzyme activity in these forests is more closely linked to changes in organic matter quality (soil C storage) and organic matter quality (N content, C:N ratio, phosphatase:phenol oxidase ratio) produced by the experimental treatments than to macroclimate, landscape-scale microclimate, geomorphological, or soil textural variations among treatment units within a site. These results suggest that the short term (1-2 yr) consequences of restoration treatments on soil microbial activity vary within and among ecosystem types in relation to their effects on the quality and quantity of soil organic matter, more so than in relation to vegetation, macroclimate, or geological setting.

This is contribution number 41 of the National Fire and Fire Surrogate Research Project. This research was funded through the National Fire Plan of the U.S.D.A. Forest Service. Although the authors received no direct funding from the U.S. Joint Fire Science Program (JFSP), this research could not have been accomplished without JFSP support of the existing FFS research project.

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UNDERSTANDING EARLY HEIGHT GROWTH OF OAK REGENERATION FOLLOWING SEASONAL PRESCRIBED FIRES

Patrick Brose[†]

The 4.5-ft stem height and accompanying 0.50-inch basal diameter are widely regarded as the minimum size thresholds when evaluating oak regeneration for final harvest of the overstory. These two stem characteristics were compared to root collar diameter by regression analysis as to their explanatory ability of early height growth of oak regeneration following seasonal prescribed fires in oak-dominated shelterwood stands. In the unburned control, there was no difference in the coefficients of determination among the three variables. Conversely, when the stems were top-killed by spring, summer, and winter fires, root collar diameter explained much more of the subsequent height growth than either basal diameter or pre-burn stem height because many small oak stems produced tall, vigorous sprouts. Examination of these sprouts revealed that they had large root systems and shared certain pre-burn characteristics such as number of stems, number of full-sized leaves, and stem height. These findings indicate that foresters need to consider root collar diameter when assessing the size adequacy of oak regeneration in previously disturbed stands and provide guidance of what stem characteristics indicate small oak stems with large root systems.

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INTEGRATING PRESCRIBED FIRE INTO MANAGEMENT OF MIXED-OAK FORESTS OF THE MID-ATLANTIC REGION: DEVELOPING BASIC FIRE BEHAVIOR AND FUELS INFORMATION FOR THE SILVAH SYSTEM

Patrick Brose, Thomas Schuler, and Jeffrey Ward[†]

In Pennsylvania and other mid-Atlantic states, the SILVAH system is an expert system for managing cherry-maple and northern hardwood forests. Developed by the US Forest Service - Northeastern Research Station, SILVAH is now being updated to develop prescriptions to sustain mixed-oak forests based on understory conditions. This latest version will recommend prescribed fire at critical times in stand development to (1) promote oak seedling establishment and (2) release existing oak reproduction from competing vegetation. However, forest managers will be hindered in using fire, in part, because basic fuels and fire behavior research is lacking.

To help fill these knowledge gaps, funding was obtained from the Joint Fire Sciences Program to evaluate the appropriateness of the existing hardwood fuel models (FM 6, 8, and 9) to common fuel conditions found in mixed-oak forests through a series of computer simulations and intensely monitored prescribed fires. Research sites are in Connecticut, New Jersey, Pennsylvania, Virginia, and West Virginia. In each state, mixed-oak forests slated to be prescribed burned are scouted for areas of uniform fuels consisting of ericaceous shrubs, leaf litter, or hardwood slash. In these areas, rectangular fire behavior plots measuring 30 to 50 feet on a side are delineated and inventoried for fuel loadings and other forest-floor conditions. In these plots, five data loggers and thermocouplers are placed, one at each corner and the center. These devices measure and record the heat and speed of the passing flame front, allowing us to compare rate-of-spread to outputs produced by the BEHAVE fire prediction system. The prescribed fires are also visually monitored to estimate flame length.

The excessively wet spring and summer of 2003 made prescribed burning last year a difficult operation. Consequently, only a few fires were monitored so results are preliminary and incomplete. At this point in time, it looks as if Fuel Model 8 is a good representation of mixed mesophytic or northern hardwood leaf litter and Fuel Model 9 accurately describes mixed-oak leaf litter. No fuel model appears to depict ericaceous shrubs, especially mountain laurel, and hardwood slash fuels have not yet been tested.

This study will continue for the foreseeable future until enough data are accumulated to definitively determine the applicability of the hardwood fuel models to forest-floor conditions commonly encountered in mixed-oak forests.

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EFFECTS OF ALTERNATIVE THINNING TREATMENTS ON TREE GRADES AT THREE UPLAND HARDWOOD SITES IN KENTUCKY AND OHIO: 30 YEAR RESULTS

John Brown, Gary W. Miller, and Kurt W. Gottschalk[†]

ABSTRACT.—Results of this study examine the effects of a wide variety of thinning treatments with respect to potential tree grade distribution of white oak (*Quercus alba* L.). Data for this analysis was collected 30 years after study establishment from two sites in Kentucky and one site in southeastern Ohio. Potential tree grade distributions were analyzed using generalized logistic regression and where appropriate, cumulative logistic regression. Thinning was found to have a statistically significant effect at the two Kentucky sites—Mckee and Baldrock—but found to have no effect at the Ohio site—Mead. Statistical contrasts of thinning treatments at the Mckee site indicated that the moderate thinning had decreased odds for poorer quality trees as compared to the odds of poorer quality trees in the severe thinning. Results for the Baldrock site were an even stronger indicator that less intensive thinning produced higher quality white oak trees. Contrasts here demonstrated that the Very Light, Moderate/Light 2nd and Moderate/Moderate 2nd treatments all had increased odds for better quality trees than did the severe treatment. No statistical differences were found among thinning treatments for potential grade at the Mead site, which had a narrower range of reduction in initial basal area among treatments.

Acknowledgments

The authors would like to thank the staff at the USDA Forest Service RWU-4153 for long term monitoring and measuring of these stands. Additional funding for data analysis was provided by the National Hardwood Lumber Association.

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LONG-TERM CHANGES IN SPECIES COMPOSITION AND FOREST STRUCTURE IN AN ARKANSAS OAK FOREST

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In 1934, a detailed forest inventory was conducted of the Sylamore Experimental Forest (SEF), a 4,000-acre oak-dominated forest in northern Arkansas. We recently discovered the field data sheets of this inventory. For approximately 2,000 acres of the SEF, we compared the results of the 1934 inventory with those from our 2002 inventory of the SEF. Among overstory trees, there was a significantly higher density of red oak, white oak, miscellaneous species, and total trees in 2002 than in 1934. For midstory trees, there were significantly more white oak, hickory, miscellaneous species, and total stems in 2002. In the understory, white oak, hickory, miscellaneous species, and total tree density were significantly greater in 2002. Fire scar data indicated the study area was frequently burned between the 1840s and the early 1900s. We attribute the relatively high current tree densities to fire suppression activities over the past 70 years.

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1ST YEAR PERFORMANCE OF NORTHERN RED OAK (*QUERCUS RUBRA* L.) ON RECLAIMED MINED LANDS IN INDIANA

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Northern red oak (*Quercus rubra* L.) seedlings were planted April 2003 on two reclaimed surface coal mines in Indiana, USA. Poor soil physical properties, low nutrient availability, and severe compaction characterize these sites. These characteristics can result in low seedling survival and poor performance, which may lead to conversion of the land to other uses. For reclamation to forestland to be effective, seedling establishment success must be improved. The objectives of this research are to compare the effectiveness of four stocktypes and assess the contribution of controlled-release fertilizer (CRF) and mycorrhizal inoculation (MI) to survival and performance of northern red oak on reclaimed mined lands. Northern red oak is known to survive on a variety of sites and has a high commercial value and was therefore selected as the trial species. The four stocktypes consisted of June-sown (Ju) and January-sown (Ja) containerized seedlings, and standard-density (Sd) (75 seedlings/m²) and low-density (Ld) (21 seedlings/m²) 1+0 bareroot seedlings. Three treatments were applied to each stocktype: MI, addition of CRF, and both MI and CRF. A control, with neither MI nor CRF, was established for each stocktype. Initial height and root-collar diameter (RCD) were recorded immediately after outplanting. Competing vegetation was controlled by herbicide application and a 2.3 m fence was erected to minimize animal damage. In October 2003, survival was assessed and height and RCD were measured for all surviving seedlings. Survival for Ld (68%) and Sd (69%) seedlings was greater than that of Ju seedlings (50%), which also had higher survival than Ja seedlings (30%). There was a significant stocktype × CRF interaction for Ju containerized seedlings, whereby survival was lower for seedlings that received CRF (64%) than for those that did not (35%). Height growth was greater for Ld seedlings than both Ju and Ja. Sd seedling height growth was also greater than that of Ja. Differences in RCD growth were not significant between any treatments. Leaf water potential (LWP) was measured to evaluate plant moisture stress. Ja seedlings were less stressed than Ld and Sd seedlings, and Ju seedlings were less moisture stressed than Ld seedlings. These differences could be partially attributed to the entire root system of containerized seedlings as compared to the loss of fine roots of bareroot seedlings. CRF increased seedling moisture stress. These results affirm that future research efforts need to focus on producing a stocktype that is able to excel on former surface coal mines.

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ECOLOGICAL CLASSIFICATION IN THE CENTRAL TILL PLAIN REGION OF INDIANA

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Research has begun to create an ecological classification system for forests of the Bluffton Till Plain Section of the Central Till Plain Region of Indiana. This system will follow the structure for ecological classification systems described by the US Forest Service. Research sites in this highly fragmented region have been selected from Indiana's Classified Forest Program, state nature preserves, and recreational areas. Data collected include tree species composition and size, herbaceous species cover, soil profile characteristics such as pH and drainage, and physiographic variables relating to topography. A variety of multivariate techniques have been used to explore the species-environment relationships, including Mantel tests, cluster analysis, factor analysis, DCA, NMDS, and MRPP. Analysis of data from one field season indicates that herbaceous species abundance may be more important as a site condition indicator than presence-absence of any particular species. Plant species abundance is dependent largely on moisture regimes created through the swell and swale topography found throughout the Central Till Plain. Current data collection is focused on the relative importance of the change in species composition and the role of minor topographic changes in the vegetation-soil-physiography relationship. A first approximation of ecological land types will be presented.

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SEDIMENT AND NUTRIENT FILTERING EFFICIENCY OF NEWLY ESTABLISHED MULTI-SPECIES RIPARIAN BUFFERS IN DIFFERENT AGRICULTURAL CROPPING SYSTEMS

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The following research project is conducting simulations to determine the effectiveness of newly established multi-species riparian buffers to filter the three most common elements associated with agricultural production land uses (sediment, nitrogen, & phosphorus). Agricultural tillage patterns are linked to the aquatic ecosystems they border through the riparian buffer. Conservation practices have been indicated as the key to reducing the impact that land use has on the aquatic ecosystems. In efforts to address the issue conservation agencies have been promoting and utilizing riparian buffers as the preferred best management practice (BMP). The removal of land from production and the investment to install and maintain a riparian buffer system is a great sacrifice placed upon the landowner whom repeatedly asks how effective is the BMP at addressing the issue. Better baseline knowledge is needed as to the effectiveness of these buffer systems to filter out sediments, and nutrients (N, P) derived from agricultural production land uses. As important we don't fully understand the physiological process the buffers travel through to become filters. We don't know when they start to filter, when they reach their peak filtering ability, and when they reach their carrying capacity. Through this research we have established 12 blocks (each containing 4 separate plots) of buffers representing two slope percentages (<6% and >6%) that are integrated with four agricultural land use tillage practices (no till, mulch till, clean till, and organic tillage practices). The buffers were designed and constructed to meet NRCS conservation standards for herbaceous cover buffers (practice code 390) and for riparian forest buffers (practice code 391) using real life dimensions in that our buffers are 50 to 55 feet in width. The herbaceous cover utilized is 4C warm season grasses, and the woody species were chosen as representative species within the watershed research area. This knowledge gained from this research will aid in assigning areas of concern or areas in need of buffers to reduce the land use impacts on the aquatic ecosystems. The data will also serve as a powerful tool to promote the BMP to landowners as a vital component within their comprehensive management system.

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THE EFFECTS OF PERIODIC PRESCRIBED FIRE ON THE GROWTH AND SURVIVAL OF RED MAPLE AND OAK SEEDLINGS ON XERIC RIDGE TOPS IN EASTERN KENTUCKY

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Declines in oak (*Quercus* spp.) advanced regeneration have been accompanied by increases in more shade tolerant species such as red maple (*Acer rubrum*), which successfully outcompete oak seedlings and saplings in lower light levels. It has been proposed that a lack of fire has contributed to these changes. A long term seedling population study was established on the Stanton District of the Daniel Boone National Forest in 1998 to quantify effects of repeated prescribed fires on survival and growth of oak (including *Q. alba*, *Q. coccinea*, *Q. prinus*, and *Q. velutina*) and red maple seedlings. Burn treatments were conducted in the early spring in different years on two non-contiguous ridges. Treatments included fire excluded, twice burned, and 3 times burned. On Klaber Ridge, prescribed fires were conducted in 1995, 1999, and 2000 for the treatment with three burns, and in 1996 and 2000 for the treatment with two burns. On Whittleton Ridge, fires were prescribed in 1995, 1999, and 2002 for the 3x burned, and in 1997 and 2002 for the 2x burned. Both ridges had 'fire excluded' treatments as well. Seedling height, annual growth, and survival have been measured since 1999. Basal diameter has been measured since 2001. Light measurements using hemispherical photography were taken over seedling clusters in 2001, 2002, and 2003.

Height growth was greatest in the first season after fire, as surviving seedlings resprouted. In 2002, red maple seedlings were tallest on 3x burned treatments (30.1 cm) and shortest on fire-excluded treatments (18.2 cm), whereas oak seedlings were tallest on the fire-excluded sites (18.1 cm) and shortest on 2x burned treatments (12.0 cm). Basal diameters were similar for red maple and oak species on all treatments (red maple: 3.5 mm; oak spp: 3.2 mm). Oak seedling survival was higher on burned sites (80 percent) compared to maples (73 percent), whereas seedling survival was similar for oaks and maple on fire-excluded treatments (87-90 percent).

Our data suggest little beneficial effect of prescribed burning on oak seedling growth, despite a slightly higher survival rate of oak versus maple seedlings with burning. Burned sites were characterized by prolific sprouting by fire-damaged or killed trees, resulting in considerable occlusion of light. Due to moderate intensity and severity, prescribed fires have not effectively killed rootstocks, which continue to sprout after successive fires. We plan to continue this study to determine long term survival and growth of oak and red maple seedlings in response to fire.

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GROUND TRUTH ASSESSMENTS OF RED OAK BORER INFESTATION IN THE INTERIOR HIGHLANDS OF ARKANSAS, OKLAHOMA AND MISSOURI

James M. Guldin, Eric Heitzman, John Kabrick, Rose-Marie Muzika, and Edward A. Poole[†]

ABSTRACT.—Forests of the Interior Highlands of Arkansas, Oklahoma and Missouri are being affected by an outbreak of a beetle named the red oak borer, *Enaphalodes rufulus* (Coleoptera: Cerambycidae). Roughly 33 percent of the area in the Interior Highlands, in stands dominated by oaks (*Quercus* spp.) that are 70 years old or older, are at risk; the dollar value at risk exceeds \$1 billion. A qualitative expert systems approach was used to define four risk strata for sampling—high, moderate, low, and no risk. From that, a stratified random sample of plots across the region was generated using a risk-based polygon approach. We sampled 191 plots over the two field seasons; of these, 108 were in Arkansas, 15 in Oklahoma, and 68 in Missouri. Tree condition varied from more than 90 percent healthy in conifers to less than 40 percent healthy in the red oak group. More than 30 percent of red oaks were in major decline or at the point of mortality. The data also suggest that white oaks are in relatively good condition across the study area relative to the red oak group; nearly 70 percent of white oaks were evaluated as healthy, and fewer than 10 percent were in major decline or at the point of mortality. Poor red oak health was observed in all three states; the percentage of healthy red oaks in the sample ranged between 35-40% in each state. However, with greater than 40 percent of sampled red oaks in the poorer two condition classes, Oklahoma may be in slightly greater risk of immediate likelihood decline in the red oak group than either Arkansas or Missouri.

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IMPACT OF BEDROCK GEOLOGY ON STREAM NITROGEN CONCENTRATIONS IN SHAWNEE NATIONAL FOREST WATERSHEDS

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Forested watersheds exhibit significant regional variability in stream nitrogen export. Bedrock geology has been identified as an important factor that can help determine stream nitrate concentrations through its control on soil fertility in unglaciated forested watersheds in the mid-Appalachian region of the United States. The objective of our study was to examine whether stream nitrogen variability was related to differences in bedrock geology in the unglaciated watersheds of the Shawnee National Forest. Twenty-four watersheds with 100% forest land and no recent disturbances were included in the study. Twelve of the watersheds contain limestone bedrock and the other twelve watersheds have sandstone bedrock. We hypothesize that the limestone watersheds will have greater stream nitrogen concentrations than the sandstone watersheds due to greater soil fertility and enhanced soil nitrogen cycling. Atmospheric nitrogen deposition rates are expected to be similar among the study watersheds given the relatively small geographical area of the Shawnee National Forest. Beginning in May 2003 and continuing for one year, stream samples will be collected monthly in each watershed at baseflow conditions. The samples will be analyzed for dissolved nitrate, ammonium, organic N, pH, and specific conductivity in the Forestry Laboratory of Watershed research (FLOW) at Southern Illinois University Carbondale. Overstory vegetation will be assessed also on each watershed.

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ASSESSMENT OF OAK DECLINE IN MISSOURI, ARKANSAS, AND OKLAHOMA

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Since 1999, widespread and locally severe oak decline and mortality have occurred throughout the Ozark Highlands of Missouri and Arkansas and the Ouachita Mountains of Arkansas and Oklahoma. Aerial reconnaissance indicates that over 400,000 acres throughout the region have been severely damaged. A unique contributing factor in the decline is an outbreak of the red oak borer (*Enaphalodes rufulus*) (Haldeman) (Coleoptera:Cerambycidae). Despite the magnitude of the decline, estimates of the problem that are based on field surveys have been lacking. In 2002 and 2003, approximately 250 field plots were established throughout Missouri, Arkansas, and Oklahoma to quantify the distribution and severity of oak decline. The project is a collaborative effort among the U.S. Forest Service North Central Research Station, the Southern Research Station, the University of Missouri, and the University of Arkansas-Monticello. Data were collected from overstory, understory, and regeneration plots in high risk and low risk forests. High risk areas were defined as having a high basal area of species in the red oak group, while low risk forests had a low basal area of red oak. Preliminary results indicate that red oak had the highest importance values on high risk plots while shortleaf pine (*Pinus echinata* L.) was the most important species on low risk plots. Fifty percent of red oak density and 53 percent of red oak basal area were dead/dying on high risk plots. In contrast, 20 percent of red oak density and 20 percent of red oak basal area were dead/dying on low risk plots. Red oak mortality was not related to tree size on either high risk or low risk plots. Sapling and seedling regeneration on high risk plots was dominated by shade tolerant species such as red maple (*Acer rubrum* L.), blackgum (*Nyssa sylvatica* Marsh.), and flowering dogwood (*Cornus florida* L.). In areas with high levels of mortality, decline-accelerated changes in species composition away from red oak-dominated forests appear to be occurring.

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A COMPARISON OF PRE-EUROPEAN SETTLEMENT AND PRESENT-DAY FORESTS IN STONE COUNTY, ARKANSAS

Eric Heitzman, Don C. Bragg, and Adrian Grell†

General Land Office (GLO) surveyors' notes from 1829-30 were compared with data collected in 1995 from Forest Inventory and Analysis (FIA) plots to quantify changes in forest species composition and tree diameters in Stone County, Arkansas. White oaks (mostly white oak (*Quercus alba* L.) and post oak (*Q. stellata* Wangenh.)) and red oaks (mostly northern red oak (*Q. rubra* L.) and black oak (*Q. velutina* Lam.)) dominated the pre-European settlement forests and accounted for 74 percent of the trees recorded in the GLO survey. In 1995, white oaks and red oaks comprised only 40 percent of the trees tallied in the FIA plots. The decrease in oak importance was accompanied by increases in the relative frequency of eastern redcedar (*Juniperus virginiana* L.), shortleaf pine (*Pinus echinata* L.), hickories (*Carya* spp.), and flowering dogwood (*Cornus florida* L.). White and red oak diameters were generally larger in 1829-30 than in 1995. Possible explanations for these changes include the increased frequency of human-caused disturbances such as timber harvesting, a decrease in wildfires since the early to mid-1900s, and surveyor bias.

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IMPACTS OF OAK DECLINE ON SPECIES COMPOSITION AND STAND STRUCTURE IN NORTHERN ARKANSAS: FOUR CASE STUDIES

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Oak decline has caused moderate to severe damage to red oaks throughout the Ozark Mountains of Arkansas and Missouri. In northern Arkansas, four mature oak forests severely damaged by oak decline were selected as case studies to describe changes in species composition and stand structure and to assess regeneration potential of oaks and non-oak species. In summer 2003, 36 plots are being established in each of four stands located on the Ozark-Saint Francis National Forest. Plots consist of one BAF 10 overstory/midstory plot in which all living, dying, and dead trees > 5.0 in. dbh are being tallied by species, tree condition, and dbh. A series of smaller plots are nested within each prism plot. These include: three 0.005-ac sapling plots in which all living trees 1.0-5.0 in. dbh are being recorded by species and dbh; and three 0.001-ac regeneration plots in which we are measuring tree seedlings less than 1.0 in. dbh and taller than 1.5 ft by species and 1-ft height class. Portions of the four study areas will be prescribed burned in fall 2003 and spring 2004 to assess the role of fire in restoring oak forests severely impacted by oak decline in the Ozark Mountains.

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PLANTING OAKS IN ARKANSAS: THREE CASE STUDIES

Eric Heitzman and Adrian Grell[†]

In 2001-2003, we used one-person power augers to plant 1-0 northern red oak (*Quercus rubra* L.) and white oak (*Q. alba* L.) seedlings in 1/2-1 ac group selection openings in three upland oak stands in Arkansas. Site preparation in each stand consisted of chainsaw felling, or girdling and chemically injecting, all stems > 1 in. dbh. This paper reports 3-year seedling survival and height growth results from one stand (site index=65-70), 2-year seedling survival and height growth results from a second stand (site index=90), and 1-year seedling survival and height growth results from a third stand (site index=65-70). Oak seedling survival was high, but height growth was slow. The relatively rapid height growth of competing non-oak species, particularly on better sites, indicates that a release may be needed to ensure continued seedling survival.

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ABUNDANT ESTABLISHMENT OF *AILANTHUS ALTISSIMA* (TREE-OF-HEAVEN) AFTER RESTORATION TREATMENTS IN AN UPLAND OAK FOREST

Todd Hutchinson, Joanne Rebbeck, and Robert Long†

In 2001, three treatments (thin only = T, burn only = B, thin+burn = TB) were applied to a 67 ha site in Tar Hollow State Forest, southern Ohio. The primary objective of the study was to evaluate these treatments as tools for improving oak forest sustainability. After the treatments, the exotic tree *Ailanthus altissima* became established in high densities in some areas. The objectives of our research here were to 1) map the pre-treatment distribution of *Ailanthus* trees 2) quantify post-treatment *Ailanthus* seedling/sapling abundance, and 3) to better understand the establishment of *Ailanthus* in relation to its pre-treatment distribution, treatment type, and light availability.

In 2003 *Ailanthus* abundance (stems ≥ 0.5 m height) was quantified in 5 m radius plots ($n = 280$). Prior to treatments, *Ailanthus* trees (≥ 10 cm DBH) were present but not abundant; in 2003 we located and mapped (with a GPS) only 32 trees or stumps (Fig. 1a). Of the 32 trees, 28 were in the TB unit; only 3 and 1 were located in the T and B units, respectively. Additional data indicates that *Ailanthus* seedlings were also sparse prior to treatments. By 2003, *Ailanthus* stems (0.5 to ~3 m height) were widely distributed and abundant in the TB unit, present in 96 percent of plots (mean density = 17.1 stems/100 m² (2179/ha)). *Ailanthus* was present in 39 percent of plots (density = 5.8 stems/100 m²) in the T unit and only 13 percent of plots (density = 0.6 stems/100 m²) in the B unit (Fig. 1b). Most seedlings presumably established from seed dispersal though aggressive clonal spread is also common for this species. Open sky (%) and *Ailanthus* abundance were not significantly correlated.

It is possible that the TB treatment created better conditions for *Ailanthus* germination and establishment than the T treatment, by both opening the canopy and causing greater disturbance to the forest floor. However, we conclude that the pre-treatment distribution of *Ailanthus* trees was likely the dominant factor determining post-treatment establishment. Our study shows that even when present at low densities, *Ailanthus* can disperse widely and establish in high densities after forest management activities, which may in turn inhibit the regeneration of native tree species.

Acknowledgments

We thank William Borovicka, Will Rohrig, Zach Traylor, Brad Tucker, and Kristy Tucker for field assistance. Although this research received no direct funding from the U.S. Joint Fire Science Program (JFSP), it could not have been accomplished without JFSP support of the existing National Fire and Fire Surrogate (FFS) research project sites.

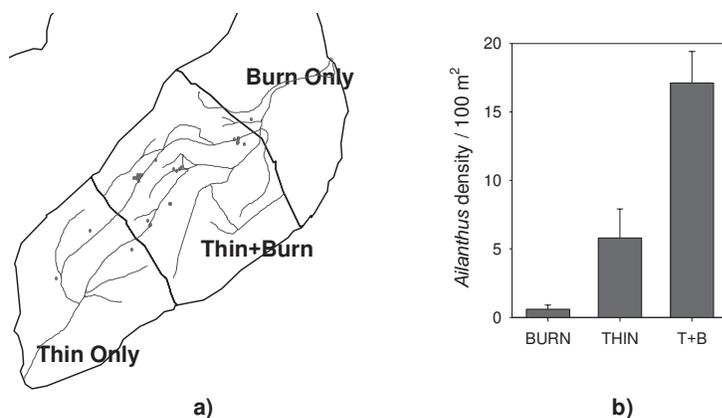


Figure 1.—A) Pre-treatment distribution of 32 *Ailanthus* trees (red dots) among the treatment units. Skid roads are indicated in blue. B) Mean density (± 1 S.E.) of *Ailanthus* stems (≥ 0.5 m height) three years after treatments.

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A COMPARISON OF THE INTEGRATED MOISTURE INDEX AND THE TOPOGRAPHIC WETNESS INDEX AS RELATED TO TWO YEARS OF SOIL MOISTURE MONITORING IN ZALESKI STATE FOREST, OHIO

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We collected soil moisture data during the growing seasons of 2001 and 2002 on over 100 points at the Zaleski State Forest in Vinton County, southern Ohio as part of a prescribed fire and thinning study (Iverson and others in press). These data were collected with a portable time domain reflectometry unit (TRIME) which measured volumetric soil water through PVC tubes via an electromagnetic field measurement of the dielectric constant of soil at 0-18, 18-36, and 36-54 cm. Measurements were taken 8 times during the growing season in 2001 and 10 times in 2002. The intention was to sample every two weeks but equipment failure plagued this effort. The PVC tubes were located on a 50 m grid throughout the study area (control and thin + burn treatments), and were accurately located via differentially corrected global positioning.

We developed the Integrated Moisture Index (IMI) some years ago to represent the long-term moisture condition of forested habitats in irregular terrain (Iverson and others 1997). Based on a conceptual model of topographic hillshading, flow accumulation downslope, curvature, and soil water holding capacity, it has been related successfully to vegetation composition (Iverson and others 1996, Hutchinson and others 1999), productivity (Iverson and others 1997), soil characteristics (Boerner and others 2000), and bird distributions (Dettmers and Bart 1999), but hitherto not measured soil moisture. IMI values traditionally have been calculated using ArcInfo Grid from 30 m digital elevation models (DEMs) that were smoothed by bilinearly resampling the 30 m pixels to 7.5 m or even smaller subcells. We modified and improved the traditional IMI algorithm by using the GRASS software function "r.flowmd", which computes flow under an infinite number of aspects, rather than the flowaccumulation function in GRID, which computes flow with only 8 possible aspects. We also acquired LIDAR data which allowed the production of 1, 2, and 4 m DEMs. Through multiple trials, we determined that the 4 m DEM best captured the flowlines downslope, so it was used for all landscape metrics. These data provide a much more detailed DEM, and consequently IMI map, where smaller drainage patterns can be captured as compared to the 30 m data.

Another index of long-term moisture is the topographic wetness index (TWI), or topographic convergence index (TCI) (Bevin and Kirkby 1979), which uses the upslope contributing areas and slope to determine an index of moisture for each cell. It has been used in several hydrological and landscape studies (e.g., Urban et al. 2000). This index was also computed with the 4 m LIDAR DEM for our study area and related to IMI and actual soil moisture values.

Extraction of landscape variables for a 6 m radius around each PVC sampling tube was accomplished via ArcInfo (ESRI 2001). This radius was selected because it roughly matches a 3x3 cell area for the 4 m cells. Average values for the circles were calculated for each of the landscape variables and joined with the moisture data for statistical treatment. Pearson correlations were calculated between landscape variables and soil moisture variables. For this analysis, grand means of soil moisture were calculated for the surface, middle, and deeper horizons (plus all horizons together) for 2001, 2002, and both years together. A subset of seven dates was also selected that had average percentages below 16 percent moisture; these represented 'dry' conditions. Multiple linear regression and stepwise regression were also run to assess the relationship of combinations of landscape variables to moisture.

The relationship between the landscape variables and recorded moisture conditions was explored by date and depth over the two years. Though expected trends of increasing soil water with increasing

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Table 1.—Correlation coefficients of landscape variables with measured soil moisture variables. First column is moisture variables for 2001 (top, middle, and bottom horizons), 2002, average of both years, average for 7 of the 18 dates with average moisture <16 percent, and a single (dry) date in September 2002. Remaining columns pertain to IMI calculated with 30 m DEM and 4 m DEM (with all four variables), the topographic wetness index, and IMI with total available water capacity removed), hillshade, and combinations of hillshade and TWI with 60:40 and 40:60 ratios, respectively. N=98, all values > 0.203 are significant at P<0.05 and values > 0.327 at P<0.001.

Moisture	IMI_30m	IMI_4var	TWI	IMI_3var	HILL	HILL60TWI	HILL40TWI
G01.top	0.211	0.256	0.174	0.263	0.260	0.278	0.279
G01.mid	0.085	0.132	0.262	0.167	0.241	0.241	0.273
G01.bot	0.123	0.186	0.314	0.205	0.238	0.251	0.279
G01.all	0.141	0.199	0.292	0.223	0.267	0.278	0.304
G02.top	0.269	0.285	0.173	0.306	0.302	0.310	0.319
G02.mid	0.085	0.107	0.223	0.156	0.229	0.221	0.256
G02.bot	0.051	0.248	0.216	0.241	0.225	0.270	0.251
G02.all	0.150	0.205	0.247	0.234	0.271	0.277	0.300
2yr.top	0.249	0.280	0.179	0.294	0.290	0.304	0.309
2yr.mid	0.086	0.121	0.245	0.164	0.238	0.233	0.268
2yr.bot	0.090	0.223	0.274	0.230	0.239	0.269	0.274
2yr.all	0.146	0.204	0.272	0.230	0.271	0.279	0.304
dry2y.top	0.276	0.292	0.205	0.330	0.312	0.327	0.334
dry2y.all	0.174	0.202	0.233	0.245	0.290	0.296	0.317
Sep02.top	0.300	0.330	0.170	0.360	0.360	0.380	0.380
Average	0.162	0.218	0.232	0.243	0.269	0.281	0.297

TWI or IMI were apparent, the correlations were low (table 1). Regression analysis revealed that a combination of hillshade and TWI provided slightly more explanatory power than any single index, so it too was included here. The table lists the variables in increasing overall relationship with the 15 soil moisture variables. Even with the high resolution DEM and the precise spatial location of the moisture tubes, the relationships account for little variance, though the correlations are mostly significant (P<0.05). All IMI and hillshade values tend to have relatively higher correlations with the surface moisture, whereas TWI correlates relatively higher with deeper soil moistures. As expected, the 4 m DEM is better than the 30 m DEM for this purpose, although the coarser data also do allow general mapping of stress zones over broader areas during drought periods (as evidenced by a relatively high correlation on the dry September 2002 sampling date). Curvature, total water holding capacity, and flow accumulation, three components of the 4-variable IMI, were not independently correlated to soil moisture. Hillshade, the largest component of IMI (40 percent for the 4-variable IMI and 50 percent for the 3-variable IMI), overall rated higher than IMI, but IMI_3var correlated slightly higher at the surface horizon. The combination of hillshade and TWI had the best correlations, regardless of moisture variables. TWI weighted 60 percent and hillshade at 40 percent was slightly better than the reverse weightings (table 1). TWI is not correlated with hillshade or IMI.

We believe that micro-scale phenomena near the PVC tubes and poor contact between tube and soil may be responsible for a large part of the unaccounted for variance in the relationships. The TRIME technology only measures soil moisture in a ~25 cm radius of the tube, and needs good contact (no air space) between tube and soil. Any air space would curtail the dielectric constant and underestimate the water contact. Tree roots near the tube can quickly dry out the soil moisture in the vicinity regardless of landscape position. In any case, the TRIME is not a fully dependable technology at this point. On the other hand, small depressions or rills, even if only several cm in size near the tubes, can collect and hold moisture even if the overall landscape is prone to dryness. Variability in the amount of litter, duff, and soil organic matter can also account for variation in either direction, as can variation in soil texture or coarse fragments. Finally, there may still a spatial mismatch between tube and the landscape variables being tested.

We have shown that some landscape variables do have weak, but significant, relationships to actual measured soil moisture for this one site in southern Ohio. The IMI, previously shown to relate well to many soil and vegetation characteristics, relates weakly to surface soil moistures. We intend to determine if these relationships hold on another site, and will explore whether a combination metric of hillshade and TWI relates well to vegetation and soil patterns.

Acknowledgments

Although this research received no direct funding from the U.S. Joint Fire Science Program (JFSP), it could not have been accomplished without JFSP support of existing Fire and Fire Surrogate (FFS) project sites. The authors are indebted to the field crews who diligently collected soil moisture over two growing seasons: David Hosack, Kristy Tucker, Brad Tucker, Bill Borovicka, Lisa Pesich, Justin Wells, and Jeff Mathews, and to Marty Jones and Pete Knopp for reviews.

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BORER DAMAGE TO OAKS IN THE SOUTHEAST MISSOURI OZARKS

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Oak mortality has increased recently following several years of drought conditions in much of the Ozarks. Among the sources of mortality has been an unusually large infestation of the native red oak borer. We studied the incidence and severity of red oak borers and other large wood-boring insects on even-aged, uneven-aged, and no-harvest study sites in the Missouri Ozark Forest Ecosystem Project. Half-acre vegetation plots (648) were randomly allocated within the sites proportional to the acreage of ecological land types present. Data on oak borer damage and crown decline were collected on live trees in the red oak group (3,385) and white oak group (3,528) from a pool of 45,900 permanently tagged trees in the winter of 2002-2003.

Overall, 77 percent of the trees in the red oak group and 33 percent of trees in the white oak group had evidence of borer sign (exit holes, bark scars, or sap stains) on the 8-foot butt log. Almost all of the trees damaged in the white oak group were in the least severe class of 1 to 10 borer signs; whereas, 28 percent of the trees in the red oak group had more severe damage. This severe damage was more frequent on trees of the red oak group on SW slopes (32 percent) and ridges (24 percent) than on NE slopes and upland waterways (20 percent each). Within the red oak group, borer damage on non-harvested stands was not different from damage on even-age stands that were thinned or uneven-age stands that were harvested by selection methods in 1996. Borers severely damaged 28 percent of the red oak group trees on overstocked stands compared to 26 percent of the trees on stands with 60 to 80 percent stocking and 23 percent of the trees on stands with 81 to 100 percent stocking. In the red oak group, the number of damaged trees and the severity of damage increased with decreasing diameter and crown class. More than 10 borer signs were present in 11 percent of the sawtimber and 9 percent of the trees with dominant crowns, but were present in 50 percent of the saplings and small pole timber and 54 percent of the trees with intermediate or suppressed crowns. Further study is needed to determine whether this high rate of infestation in small trees will adversely affect stand dynamics, or whether cultural treatments might reduce the scope of the problem.

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NED SOFTWARE FOR SUSTAINABLE FOREST MANAGEMENT: NEW DEVELOPMENTS

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The term NED describes a set of computer programs designed to help resource managers and landowners develop goals, assess current and potential conditions, and produce sustainable management plans for forested properties. A number of new programs have been released recently. Stewplan, published in 2003, is software that helps create standard forest stewardship plans to facilitate enrollment in the Stewardship Program sponsored by the USDA Forest Service and state agencies throughout the United States. NEDLite allows use of handheld computers using the Palm OS for field data collection, with automatic transfer of data to NED-1 and NED-2. NED-1 is a comprehensive analysis program to evaluate goals and current conditions on a forested property from the perspective of various forest resources for management areas up to several thousand acres. NED-2, presently in beta testing, expands on previous versions of NED software by integrating treatment prescriptions, growth simulation, and alternative comparisons of management scenarios across a management unit. NED-2 implements a goal-driven decision process that ensures that all relevant goals are considered; the character and current conditions of forestland are known; alternatives to manage the land are designed and tested; the future forest under each alternative is simulated; and the alternative selected will achieve the owner's goals. The resources NED addresses include visual quality, ecology, forest health, timber, water, and wildlife, allowing a user to evaluate the degree to which individual stands or a management unit as a whole may provide the conditions required to accomplish specific goals. Programs such as the Forest Stewardship Planning Guide and NEWILD allow people with an interest in managing their forests to improve their understanding of various management activities and their effects on the forest, even if they lack detailed data. The software is distributed without charge and is available for downloading from the internet at <http://www.fs.fed.us/ne/burlington/ned>. NED development is led by the USDA Forest Service's Northeastern and Southern Research Stations in cooperation with many other organizations and individuals.

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EVALUATION OF PASSIVE FLAME HEIGHT SENSORS FOR THE CENTRAL HARDWOODS REGION

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We conducted a pilot study to determine the best material for measuring flame height in the oak-hickory forest litter fuels present in the Central Hardwood Region. Fire-retardant-soaked cotton string and four different compositions of lead/tin solder (63/37, 60/40, 50/50/, and 40/60) were compared. We evaluated three replicates containing four strands of each material suspended vertically between two guy wires. At the end of each linear replicate two height poles were positioned within view of a video camera. Backing and head fires were set so that the flaming front advanced perpendicular to each replicate. The string was measured to the highest point where it was uniformly blackened and the solder sensors were measured where they melted off. Averages for each sensor type were computed for each replicate and regressed on actual average flame height determined from a video taken during each test. Analysis indicated that string soaked in fire retardant performed the best having the highest R², 0.99 [Flame Height = (.61)*(Uniformly Blackened String Height)]. All compositions of solder had R² > 0.96, with 50/50 solder having a R² of 0.98.

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IMPACTS OF PINE BLUESTEM RESTORATION ON NUTRIENT REGIMES OF SHORTLEAF PINE-HARDWOOD STANDS IN THE OUACHITA MOUNTAINS OF ARKANSAS

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The current forest and vegetative communities of the Ouachita Mountains reflects fire suppression that followed removal of the virgin forests during the late 19th and early 20th century (Bukenhofer and Hedrick 1997). Shortleaf pine (*Pinus echinata* Mill) still dominates the overstory of these second growth forests but current forests contain a much higher density of hardwoods than did the frequently burned, virgin shortleaf pine forests (Foti and Glenn 1991). Woody understory vegetation, rather than the forbs and grasses that occurred in the more open grown virgin forest, dominate the understories of these second growth forests. In an effort to provide habitat for red-cockaded woodpeckers (*Picoides borealis*) and other biota, the Ouachita National Forest is currently restoring shortleaf pine-bluestem grass (SPBG) ecosystems to a portion of this region (Bukenhofer and Hedrick 1997). Restoration includes harvesting and competition control to reduce pine and hardwood basal areas to approximately 13-14 and 2-3 m²/ha respectively. In addition prescribe fires are performed on a 3-5 year interval during the dormant season to reduce understory woody vegetation (Masters et al. 1996). To better quantify the effects of long-term shortleaf pine bluestem restoration activities on nutrient availability, we compared mineral surface soil chemistry and foliar nutrient concentrations in stands that had experienced SPBG restoration activities for 17-21 years to those in unrestored stands. We feel that results from this study may be applicable to land managers who are converting dense mixed hardwood stands to more open oak woodlands, since both practices significantly reduces stand density, initially adds large amounts of hardwood woody debris to the forest floor, and utilizes repeated prescribed fires to control unwanted vegetation and alter understory composition.

Six shortleaf pine-hardwood stands were used for the study. Three stands had not received any restoration or silvicultural activities for a period of 40 years prior to the study and were typical closed canopy shortleaf pine-hardwood stands (control). The other three stands were restored SPBG stands. Initial overstory and midstory harvesting and competition control activities occurred from 1978-1980 in the SPBG stands. Midstory and overstory hardwoods that were felled were typically left on the ground due to the lack of suitable hardwood markets. Prescribed fires were generally applied on a 2-4 year interval starting between 1978-1980. The last prescribed fire at all three SPBG stands occurred during March of 1997 prior the initiation of the study. All stands occurred on Carnasaw or Sherless soils series (NRCS 1998) and have loamy surface textures. These soils have similar surface soil (35-50 percent) and subsurface soil (35-40 percent) rock contents. Stands chosen for the study were located on 10-20 percent slopes, on southern to southwestern aspects at elevations between 237 and 317 m above MSL, and within 2-3 km or 34° 47' N Latitude and 94° 10' W Longitude.

Mineral soil to a depth of 15 cm from five locations and current year foliage from five dominate shortleaf trees was collected from each of four 25-35 m diameter plots in each stand. Soils and foliage were collected in the fall for three consecutive years following the March 1997 fires. Soils were air dried and sieved through a 2 mm screen. Soil pH (1:2 soil:water ratio), total nitrogen and carbon (Leco CN analyzer), mineralizable N (anaerobic incubation), and P, K, Ca, and Mg (Mehlich III extraction) were determined. Foliage was collected from the top 1/3 of the crowns. Foliage was ground to pass a 1 mm sieve and then N, P, K, Ca, Mg, and S concentrations were determined following either a micro-Kjeldhal or a perchloric digestion.

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Table 1.—Mean surface soil (0-15 cm) characteristics collected during the fall of 1997-1999 within three shortleaf pine bluegrass stands (SPBG) and three control stands in the Ouachita Mountains of Arkansas.

	SPGB	Control	P-Value
pH	5.3	4.9	0.010
C (g/kg) ¹	25.6	19.8	0.003
C:N ¹	20.9	17.9	0.016
N (g/kg) ¹	1.2	1.1	0.041
Min. N (mg/kg)	59.8	50.4	0.011
P (mg/kg)	6.2	7.0	0.379
K (mg/kg)	76.0	66.1	0.142
Ca (mg/kg)	533.1	332.2	0.031
Mg (mg/kg)	117.2	134.7	0.525

¹Measurements only from the fall 1999 sample collection

In 1998 shortleaf pine and hardwood basal areas averaged respectively 24.3 and 8.4 m²/ha at the control stands and 15.8 and 3.7 m²/ha at the SPBG stands. Surface mineral soil pH, mineralizable N, C:N ratios, as well as Ca, N, and C concentrations were consistently higher in the SPBG stands than the controls (table 1). Increases of pH and Ca frequently occur following fire (Fisher and Binkley 2000). However, increases of soil Ca concentrations have also been reported following harvesting in mixed oak stands (Johnson and Todd 1998) and attributed to decomposition of Ca rich slash and woody debris. Initial harvesting and competition control in shortleaf pine-hardwood stands within the Ouachita Mountains frequently adds large amounts of hardwood woody debris to the forest floor because there are poor markets for hardwood pulp and sawtimber. Increases of C, N, and mineralizable N have also been reported following harvesting and application of fire. However, we believe that the large increases in C, N, mineralizable N as well as Ca observed in this study were due to the combination of these two silvicultural treatments. Masters et al. (1993) reported that increases in nutrient concentrations in soils of shortleaf pine-hardwood stands were much greater when these two treatments were combined than they were applied individually. Our results support this conclusion since increases in pH and Ca were generally greater than those reported for McKee Jr. (1991) following application of dormant season fires in southern pine stands on similar intervals and time spans as those utilized in this study.

Alterations of foliar nutrient concentrations of the dominate and codominate shortleaf pine were also evident (table 2). During the fall immediately following the March 1997 prescribed fire, foliar concentrations of N, P, and K were higher in the SPBG stands than the control stands. Only foliar concentrations of K remained elevated in the SPBG stands during the entire study period. It seems likely that the short-term increases in foliar N and P were directly related to an increase in available N and P following the fire. The elevated concentrations of foliar K for a longer period of time reflect a more long-term impact of fire and/or the reductions in stand densities within the restored stands.

These results indicate that harvesting and prescribed fire associated with pine bluestem grass ecosystem restoration has altered nutrient regimes in these stands. However, rather than having a negative impact and a reduction in nutrient availability, these activities have appeared to increase nutrient availability. Availability of N and Ca has increased in the soils of the restored stands. In addition concentrations of nutrients such as K have increased in the foliage of the shortleaf pine. We hypothesize that the use of similar harvesting and prescribed fire scenarios in at least poor quality dense mixed oak stands may have similar impacts. First, conversion of dense mixed oak stands to open woodland condition entails large reductions in overstory and midstory densities. Second, conversion of dense mixed oak stands like initial shortleaf pine restoration activities dramatically increases the amount of hardwood slash and downed woody debris in the stand. Finally, conversion of these mixed hardwood stands will most likely require recurring use of prescribed fire to maintain an oak dominated overstory.

Table 2.—Mean nutrient concentrations collected during 1997-1999 from shortleaf pine dominate and codominate trees in three shortleaf pine bluestem grass (SPBG) and three control stands in the Ouachita Mountains of Arkansas.

Nutrient (g/kg)	1997		1998		1999	
	SPBG	Control	SPBG	Control	SPBG	Control
N	12.5a ¹	11.4b	13.4b	14.2a	13.3a	13.8a
P	0.9a	0.8b	1.1a	1.1a	1.0a	1.0a
K	4.0a	3.6b	4.5a	4.2b	4.4a	4.0b
Ca	2.5a	2.2a	2.0a	2.1a	2.3a	2.4a
Mg	1.2a	1.2a	1.1a	1.3a	1.3a	1.4a

¹Treatments for a given year and concentration with the same letter are not different at the p=0.05

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EVALUATING BOLE DAMAGE AND CROWN DECLINE AFTER PRESCRIBED FIRE IN AN APPALACHIAN HARDWOOD FOREST ON THE CUMBERLAND PLATEAU, KY

Elizabeth Loucks and Mary A. Arthur[†]

Wounding of valuable timber trees during prescribed fires is a primary concern of forest managers, as fire scars often serve as the entry pathway for decay fungi and insects. Bark-char height has been found to be a good predictor of mortality and wounding, and thus future wood quality. However bark-char height and bole damage have not been well studied on a landscape scale in closed canopy forests after prescribed fires for which fuel loading and fire behavior were also measured. Sixty 10 x 40 m plots were established during the summer of 2002 on three separate study sites. Slope and aspect were recorded for each plot. Within each plot, trees were tagged and measured, crowns were evaluated for dieback, boles were evaluated for damage below dbh, the presence of woody litter greater than 1 inch in diameter within 30 cm of each tree was recorded, and litter depth around each tree was categorized. Plot fuel load was measured in January and February 2002 using Brown's planar intercept transects method and by collecting 30 X 30 cm sections of the forest floor. Management prescribed fires were ignited either by hand or by helicopter, in late March and mid-April, resulting in considerable variability in fire intensity and severity among the three study sites. Temperatures were recorded using temperature sensitive paints on aluminum tags. Immediately after the fires, fuel loads and stem-bark char heights and widths were recorded. During the summer of 2003, trees will be re-evaluated for crown decline and bole damage. We will use multiple regression to analyze the factors that influence char height, including dbh, species, plot fuel loading, litter depth, and presence of woody fuels. Bark-char heights ranged from zero to seven meters in height. Char heights tended to be lower on smooth, thin barked species, such as American beech (*Fagus grandifolia*) and sugar maple (*Acer saccharum*), and higher on rough barked species such as sourwood (*Oxydendrum arboreum*) and black oak (*Quercus velutina*). While accurate estimates of bole damage will not be available for one to two growing seasons after the fires, immediate mortality, bole damage, and crown dieback appear to be greatest in areas of high fire intensity where bark-char heights also tended to be higher.

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SPECIES AND SITE DIFFERENCES IN STEM-BARK CHAR AFTER PRESCRIBED FIRE IN A SOUTHERN APPALACHIAN HARDWOOD FOREST, KENTUCKY

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Wounding of valuable timber trees during prescribed fires is a primary concern of forest managers, as fire scars often serve as an entryway for decay fungi and insects. Bark-char height and width have been found to be good predictors of mortality and wounding, and thus future wood quality. However stem-bark char and bole damage have not been studied on a landscape scale in closed canopy forests for which fuel loading and fire behavior were also measured. Sixty 10 x 40 m plots were established during the summer of 2002 on three separate study sites on the Cumberland Plateau in the Morehead District of the Daniel Boone National Forest. Within each plot, trees (>2 cm DBH) were tagged and measured, boles and crowns were evaluated for dieback, and litter and fine woody fuels were measured around each tree. Brown's (1974) planar intercept transects and 30 X 30 cm forest floor blocks were used to quantify fuel loads in each plot in January and February 2002. Temperature sensitive paints on aluminum tags were placed at three heights (0, 20, and 40 cm) in each plot to record fire temperatures. Prescribed fires were ignited either by hand or by helicopter, in late March and mid-April, resulting in considerable variability in fire intensity and severity among the three study sites. While temperatures on the Buck Creek and Chestnut Cliffs sites did not differ, the Wolf Pen site had higher temperatures than Buck Creek at 20 cm (230° and 313° C) and 40 cm (165° to 225° C), and higher temperatures than Chestnut Cliffs at 0 cm (574° to 500° C). Immediately after the fires, fuel loads and stem-bark char heights were recorded. Bark-char heights ranged from zero to seven meters in height. A regression model with class variables was used to analyze the bark-char data. Bark-char heights on smooth barked species, such as red maple (*Acer rubrum* L), were lower than on rough barked species, such as yellow poplar (*Liriodendron tulipifera* L.). As anticipated, mean bark-char heights increased with larger DBH size classes. Landscape position was a significant predictor of bark-char height, with higher char in the xeric locations. While accurate estimates of bole damage will not be available until two or three growing seasons after the fire, mortality, bole damage, and crown die-back appear to be greatest in areas of high fire intensity where bark-char heights also tended to be higher, as expected.

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RESPONSE OF GARLIC MUSTARD (*ALLIARIA PETIOLATA*) AND FOREST UNDERSTORY TO HERBICIDE AND PRESCRIBED BURNING

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Biological invasions of exotics are potentially disrupting and costly to native ecosystems. Invasive and exotic species may negatively affect a community by leading to decreases in population numbers, increases in species extinctions, or altering ecosystem function (Mooney and Drake 1986; Vitousek et al. 1996). The effects may be brought about by several factors, including increased competition pressures, predation, disease, or amensalism (Williamson 1996). One such invasive, garlic mustard (*Alliaria petiolata*), has been increasing in numbers in many midwestern habitats. Due to high propagule pressure and ability to thrive in a wide variety of sites, garlic mustard can degrade sites quickly if left unchecked. Methods to eradicate garlic mustard have produced varied results, with herbicide and prescribed burning the most effective. The study areas utilized for this project are located at Martell Forest, a Purdue University owned property in Tippecanoe County. Two study areas are used, and plots are established prior to treatments. Plots are 20 X 20 meters in outside diameter, and contain an inner 10 X 10 meter sampling and treatment area. For the herbicide and burn treatment area, the plots are divided in half, and treatments of herbicide or burn/no burn will be randomly assigned to either half. Arsenal® and Plateau® will be applied in several concentrations to the herbaceous layer, and separate burn events will take place in the study plots. Data collected on the mortality and regeneration of the garlic mustard and native vegetation after treatments will be integral to developing a long term management goal.

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HARDWOOD SEED PRODUCTION IN AN OLD-GROWTH MIXED MESOPHYTIC FOREST IN SOUTHEASTERN OHIO

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Old-growth forests provide an invaluable ecological benchmark in an otherwise widely managed forest landscape. These forests permit insight into important processes that might not otherwise be observable in the landscape; i.e., processes that only manifest themselves at the latest stages of forest succession or in the absence of anthropogenic disturbance. Dysart Woods is one of a small number of remnant old-growth stands in Ohio and is believed to be the only example of mixed mesophytic forest vegetation in the state.

Previous studies at Dysart Woods (McCarthy and others 2001) surveyed two separate stands and found that the overstory (stems > 10 cm DBH) is dominated by white oak (*Quercus alba*; DEN = 20 stems ha⁻¹, BA = 9.6 m² ha⁻¹), sugar maple (*Acer saccharum*; DEN = 68.6 stems ha⁻¹, BA = 3.3 m² ha⁻¹), and beech (*Fagus grandifolia*; DEN = 72.86 stems ha⁻¹, BA = 11.0 m² ha⁻¹). There are a relatively large number of white oak stems in the 85-125 cm dbh classes. Red oak (*Q. rubra*), tuliptree (*Liriodendron tulipifera*), blackgum (*Nyssa sylvatica*), and cherry (*Prunus serotina*) are also locally important. The midstory (DBH ≥ 2.5 and < 10 cm dbh) is dominated by Sugar maple (DEN = 349 stems ha⁻¹) and beech (DEN = 201 stems ha⁻¹), as is the seedling layer (stems < 2.5 cm DBH; DEN = 14,507 and 1,536 stems ha⁻¹, respectively). Interestingly, the canopy dominant white oak is virtually absent from the midstory and understory. Based on dendrochronological studies (Rubino and McCarthy 2000) we estimate that there has been almost no white oak recruitment in these woods for upwards of 200 years.

Our composition and structural results present a conundrum. Recent studies suggest that fire may be an important silvicultural tool in regenerating oaks. A limited disturbance chronology (373 years) was constructed from one large (1.0 m dbh) basal slab of white oak which indicated that the woods burned repeatedly during the major period of regional settlement (1810-1860)(McCarthy and others 2001). No fire scars were detected post 1880. Thus, we would expect a much greater number of mid-diameter size classes of white oak had it recruited during that time frame. The paucity of smaller diameter classes is still consistent with an absence of fire hypothesis. The role of fire is quite unclear, and may be more so under relatively mesic conditions.

Given the lack of regeneration, apparently for a long period of time, we decided to examine seed production in the two stands (ca. 11 ha each). Forty-eight conical seed traps, 0.25 m² in size, were arrayed in a 6 × 8 grid in each stand, with each trap spaced 10 m apart (96 traps total, 24 m² total sample area). Traps were deployed in mid-August each year and sampled again in mid- September, October, and November from 1996-2000. Seeds were destructively sampled and scored as sound, aborted (not fully developed), or depredated (by fungal pathogen or insect predator).

There was an order of magnitude difference in the number of sound seed produced within and among species (fig. 1). Sugar maple produced from 34 to 6473 seeds (1999 & 1998, respectively), beech produced 349 to 3071 seeds (1999, 2000), tuliptree produced 730 to 3481 seeds (1997, 1998), but white oak produced only 1 to 473 seeds (2000, 1998) per 24 m².

There was a strong periodicity to the data for all species (fig. 1). Virtually all species (including cherry, blackgum, and red oak; data not shown) alternated between years of increased and decreased seed production. These data lend support to a resource limitation hypothesis regarding seed production; however, the fact that all species respond on the same schedule also lends considerable support to the

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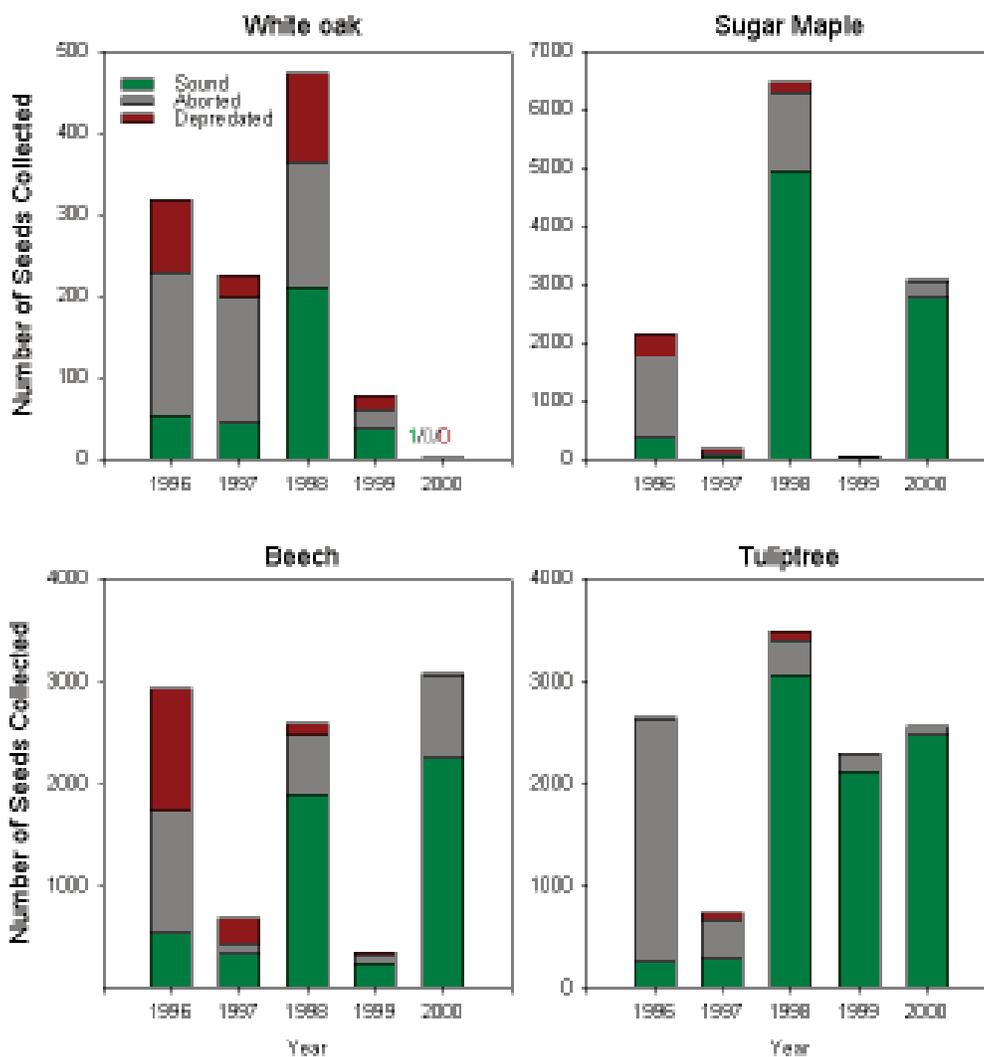


Figure 1.—Number of seeds collected per 24m² of sample area for the period 1996-2000. Data shown are for the four major overstory species present at Dysart Woods (Belmont Co., OH) broken down by seed quality (sound, aborted, depredated). Note the difference in scales among the species.

influence of the environment in setting that schedule. Similar patterns have been observed in Missouri (Christisen and Kearby 1984) and elsewhere.

White oak was the poorest seed producer of all the species and its fecundity was generally only 10 percent of its co-dominants (fig. 1). It always produced the most seed in the more xeric of the two stands (data not shown). All species displayed a modest degree of seed abortion (inability to fully mature the fruit) and/or pre-dispersal seed predation (by insects or fungi; largely the former) depending upon the year. However, the proportion of unsound seed for white oak was considerably greater for (range 52 to 83 percent) than it was for most other species in most years of this study. Thus, the absolute number of white oak propagules available for germination is incredibly small. Of the small number of viable seeds reaching the forest floor, most of these likely suffer post-dispersal predation by small rodents. A post-dispersal predation rate of 99 percent is not uncommon among many nut-bearing trees (McCarthy 1994).

With or without fire, white oak will be unable to maintain dominance in these woods—there is no longer a sufficient source of propagules to maintain the population and the woods is relatively isolated in an otherwise agricultural landscape. Most of the white oaks in this stand are of senescent age and past the period of major reproductive output (Johnson 1994). Regeneration of white oak is virtually absent and it is unclear that there has been any recruitment for almost 100-200 years. White oak may hold its dominance a bit longer on the drier of the two stands, but it too will likely give way to beech, sugar maple, and tuliptree populations as they generally support a healthier reverse-J diameter distribution.

The authors wish to thank the following Miami University Middletown students for their field assistance with this project: George Bakonyi, Cybil Franz, Jessica Hawk, and Peggy Parks. We would also like to thank the Miami University Middletown Research and Grants committee and the Ohio University Honors Tutorial College for providing partial funding for this project.

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ACCURACY OF TREE GRADE PREDICTIONS FOR FIVE APPALACHIAN HARDWOOD SPECIES

Gary W. Miller, Aaron T. Graves, Kurt W. Gottschalk, and John E. Baumgras[†]

Tree quality is an important factor that influences cut-or-leave decisions in thinning prescriptions implemented by landowners, forest managers, and wood-using industries. Tree grade is a useful estimate of tree quality, and predicted tree grade can be used to assess changes in tree quality and value over time. This study measured the accuracy of predicted tree grades for five Appalachian hardwood species over a period of 12 to 15 years.

Two West Virginia sites were used in this study. The first study area is located on the Monongahela National Forest (MNF) in northern Pocahontas County (Miller 1997), where the initial overstory was predominantly black cherry (*Prunus serotina* Ehrh.), with some white ash (*Fraxinus americana* L.) and red maple (*Acer rubrum* L.). The second study area is located on the West Virginia University Forest (WVUF) in Monongalia and Preston Counties (Graves and others 2000), where the initial overstory consisted of yellow-poplar (*Liriodendron tulipifera* L.), northern red oak (*Quercus rubra* L.), black cherry, sweet birch (*Betula lenta* L.), white oak (*Quercus alba* L.), chestnut oak (*Q. prinus* L.), and scarlet oak (*Q. coccinea* Muenchh.), with smaller amounts of red maple.

Thinning treatments were applied in each study area, and permanent plots were used to monitor stand and individual tree development. All residual trees were graded according to Hanks (1976) in thinned and control plots after the thinning treatments. Initial tree grade data were collected in 1986 on the MNF site and in 1989 on the WVUF site. At that time, a future tree grade was predicted assuming that DBH would increase to ≥ 15.6 inches and the tree would retain the surface and cull defects apparent at the beginning of the study.

In 2000, each tree was graded a second time using the same rules, thus providing a means of assessing the accuracy of tree grade predictions in both thinned and control stands. The data set consisted of 588 black cherry, 404 northern red oak, 167 red maple, 191 white and chestnut oaks, and 450 yellow-poplar, for a grand total of 1,800 trees. Pearson chi-square tests were performed on tree counts by three grade prediction categories (higher, correct, and lower) to detect differences between thinned and control plots. When chi-square tests were significant ($p < 0.05$), asterisks were placed on values that contributed most to the chi-square statistic (table 1), based on an examination of standardized deviates (SYSTAT 2000).

When all 1,800 trees were considered together, 9 percent were higher than the predicted grade, 80 percent were predicted correctly, and 11 percent were lower than the predicted grade (table 1). Grade predictions for black cherry and northern red oak appeared to be less accurate in thinned stands compared to controls, but such differences were not apparent for other species. Grade predictions for larger diameter and higher grade trees were generally less accurate than those for smaller or lower grade trees. Trees that had lower grades than predicted usually exhibited unforeseen increases in cull deduction. While grade factors such as DBH and percent sweep are readily apparent and can be measured uniformly, estimation of internal rot and prediction of clear cuttings are less exact and may be an important source of error. Still, the conservative approach of assuming that current surface defects and cull deductions would persist, as used in this study, provided remarkably reliable predictions of future tree grade.

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Table 1.—Accuracy of predicted tree grades in 2000 by species and treatment.

Species	Treatment	Actual Grade vs. Predicted ¹			n	Chi ²	p-value ²
		Higher	Correct	Lower			
Black cherry	Thinned	11	72	17*	445	11.3	<0.01
	Control	7	87	6*	143		
Northern red oak	Thinned	10	79	11*	174	7.5	<0.01
	Control	10	86	4*	230		
Red maple	Thinned	12	83	5	82	1.9	0.17
	Control	11	78	11	85		
White oak group	Thinned	10	80	10	81	0.2	0.67
	Control	4	84	12	110		
Yellow-poplar	Thinned	9	80	11	203	0.3	0.57
	Control	5	82	13	247		

¹Tree grades were predicted in 1986 for black cherry; 1989 for other species.

²Pearson chi-square tests were performed on tree counts by species, treatment, and grade prediction.

Acknowledgments

This study was partially funded by National Hardwood Lumber Association, 6830 Raleigh-LaGrange Road, Memphis, TN 38184.

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TEMPORAL TRENDS IN BRANCH MORPHOLOGY IN YOUNG *POPULUS* CLONES

Stephen G. Pallardy, Daniel E. Gibbins and Ryan C. Dowell†

Temporal trends in branch morphology (living, first-order proleptic and sylleptic shoots) in the second through fourth years of growth were assessed for young *Populus* clones grown for biomass at 1 x 1 m spacing on a floodplain site in central Missouri. Angle of branch origin tended to increase with age in a *Populus deltoides* x *P. nigra* hybrid clone (I45/51) and in two of three *P. deltoides* clones derived from Midwest region collections (clones 2059 and 1112, but not 26C6R51). Total branch length, number of branches (table 1) and number of branches per meter of tree height declined over the period in the hybrid *Populus* clone while it increased somewhat or remained stable in *P. deltoides* clones. However, the hybrid clone still had a significantly larger number of branches than did *P. deltoides* clones after four years. Early in the experiment, the hybrid clone had a substantially greater number of branches placed in the lower half of the stem than did *P. deltoides* clones, but with time this tendency was lost. After the fourth year of growth all clones had few or no branches below the stem midpoint. Length-weighted vector averages of branch azimuth (fig. 1) indicated that there was a significant trend toward greater branch growth on the south side of trees, but little apparent clonal variation in this attribute. Interestingly, this tendency in orientation of branches was maintained in every year. Overall, initial differences in branch morphology between hybrid clones and *P. deltoides* clones shown in year two were much reduced or absent by the fourth year of growth. Similar trends were seen in biomass growth in these clones.

Table 1.—Number of branches per tree in a *Populus* plantation in the second through fourth years of growth. Means within a column not followed by the same letter are significantly different ($p < 0.05$).

# of branches	2 y-old	3 y-old	4 y-old
I45/51 (<i>Populus deltoides</i> x <i>P. nigra</i>)	30.0a	24.0a	26.5a
26C6R51 (<i>Populus deltoides</i>)	17.4b	16.2b	19.2b
2059 (<i>Populus deltoides</i>)	11.8b	19.3ab	18.3b
1112 (<i>Populus deltoides</i>)	10.0b	17.0b	18.2b

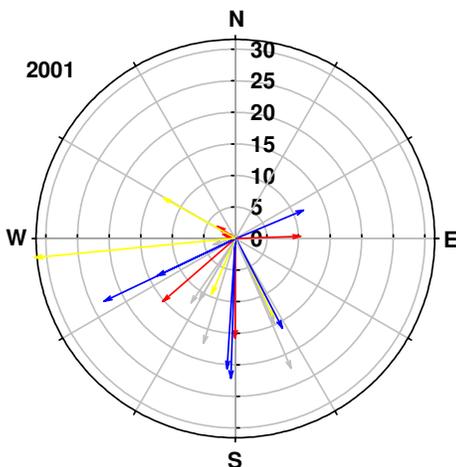


Figure 1.—Length-weighted vector averages of branch azimuth for *Populus* clones in the third (2001) growing season of growth. Different clones are indicated by color (blue=I45/51, n=6; red=26C6R51, n=5; gray=2059, n=6; yellow=1112, n=5).

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BENCHMARKING STUDY BASED ON CRITICAL SUCCESS FACTORS FOR HOUSEHOLD, OFFICE AND KITCHEN CABINET WOOD FURNITURE INDUSTRIES IN THE UNITED STATES

Henry Quesada and Rado Gazo[†]

This research focuses on defining a benchmarking methodology based on critical success factors and key business processes for household, office and kitchen cabinet wood manufacturers in the US. The results of this research aim to help those that need to refocus and determine new strategic goals to remain competitive. The first part of this study consisted of researching performance of wood furniture manufacturers by looking at companies' publicly available financial reports, stock reports and other sources of information. The objective of this first part was to identify critical success factors for this industry sector and compare US furniture manufacturers' performance against world top-class manufacturers. A statistical test with a 95% confidence level was defined to either reject or accept that US furniture wood manufacturers have the same level of performance than world top-class manufacturers. The second part consisted of visiting selected furniture manufacturing plants in Indiana with the objective of identify critical success factors, key performance metrics used to measure these factors' performance, and best manufacturing practices. Only manufacturing plants ranking 300th or better according to the January 2003's FDM 300 publication were invited to participate. Twenty Indiana manufacturing plants were contacted and four accepted to participate in the study. Results of the first part showed that customer satisfaction, price, innovation and shareholder satisfaction are among the main critical success factors defined by these manufacturers. When US furniture manufacturers' performance was compared with world class manufacturer's performance, the statistical test revealed that US wood furniture manufacturers' performance underachieve in the majority of the key selected performance measures. Results of the second part will not be released until next year.

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ESTABLISHMENT SUCCESS OF CONSERVATION TREE PLANTATIONS IN INDIANA

Amy L. Ross-Davis, Douglass F. Jacobs, Shorna R. Broussard, and Anthony S. Davis[†]

In the Central Hardwood Forest region, variable success of privately-owned hardwood plantations has been attributed to competing vegetation and damage due to animal browse. Although specific silvicultural practices are known to mitigate these pressures, few studies have addressed how the management behavior of private landowners translates into establishment success. This research (i) examined landowner motivations for establishing hardwood plantations in Indiana between 1997 and 2001, (ii) characterized landowners according to the values they hold for their land, (iii) described the silvicultural practices employed within these plantations, (iv) quantified plantation establishment success, as defined by seedling survival and vigor, and (v) related these motivations, ownership characteristics, and silvicultural practices to overall plantation establishment success. Motivations to plant and ownership characteristics were assessed through a statewide telephone survey, through which each landowner provided detailed silvicultural histories. Field data were collected from 87 of these plantations to assess seedling survival, vigor, and abundance of surrounding vegetation. Landowners valued their land for the privacy it provides, as a place of residence, and as a legacy for future generations. They plant trees primarily to provide for future generations, provide food and habitat for wildlife, and to conserve the natural environment. Plantation establishment success was lowest on sites owned by citizens who did not value their land as a legacy for future generations. Many NIPF owners engaged in requisite behaviors to ensure plantation establishment success. Survival was highest on sites that were treated with herbicide prior to planting and had been mechanically planted. The percentage of trees with evidence of dieback was highest on sites at which browse protection measures had been used. Sites planted by a professional forester and those with herbicide applied subsequent to planting had a higher percentage of trees deemed free-to-grow.

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CERULEAN WARBLER HABITAT USE IN AN OAK-NORTHERN HARDWOODS TRANSITION ZONE: IMPLICATIONS FOR MANAGEMENT

Scott H. Stoleson[†]

The cerulean warbler, *Dendroica cerulea*, is a small songbird of deciduous forest canopy found locally through much of eastern North America. Since 1966, populations have declined range-wide by over 4% per yr, resulting in heightened conservation concern and a petition to list the species as “threatened” under the Endangered Species Act. Conservation planning for regional populations of cerulean warblers is hampered by a lack of knowledge of population status, habitat affinities, area requirements, and threats. Studies have indicated the species requires large tracts of intact mature forest, leading some to implicate even-aged timber management as a threat to the species’ survival. In 2003 we initiated a study to identify and quantify habitat use by cerulean warblers in northwestern Pennsylvania in a zone of transition between oak-dominated forests to the south and northern (Allegheny) hardwoods to the north; and to assess the impacts on the birds of a shelterwood-prescribed burn treatment increasingly used in the region to regenerate oak forests. We used tape-playback methods to survey for ceruleans at 408 sampling points across a range of forest types and topographic positions. We found no ceruleans in Allegheny hardwoods (cherry-maple), the dominant forest type in the region; all ceruleans occurred in mixed oak stands (n = 48) or on sycamore-dominated river islands (n = 2). Although ceruleans were detected disproportionately on lower slopes and bottoms, this likely reflects the non-random distribution of oaks in the region. Eight of the 65 stands surveyed had been shelterwood-cut; these stands had both a significantly greater probability of having ceruleans present (P = 0.028), and a higher density of ceruleans when present (0.42 vs 0.71 birds/point; P = 0.05). A possible explanation is that shelterwood cuts simulate the heterogeneous canopy structure of the uneven-aged mature forests preferred by the warbler. While the results of this pilot study are suggestive, our sample sizes for shelterwood stands are small. Future work will examine in greater depth the impacts of shelterwoods on cerulean abundance and nesting success.

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FIRST YEAR RESULTS FOLLOWING THINNING OF 12-YEAR-OLD WHITE OAK PLANTING IN SOUTHERN INDIANA

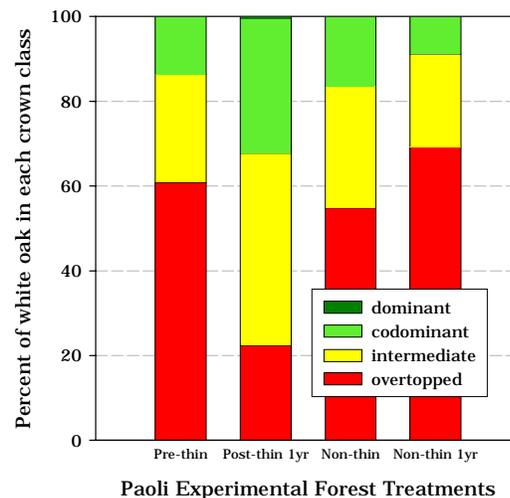
Dale R. Weigel and Daniel C. Dey[†]

Numerous recommendations for increasing the presence of oak in regenerating stands have been proposed. One such method is to plant oak seedlings beneath a shelterwood that is removed 3 years after planting (Weigel and Johnson 1998). The shelterwood creates an environment that promotes the development of a competitive oak root system while controlling competing vegetation growing under its partial shade. Despite early successes in regenerating oak in southern Indiana, maintaining oak dominance in developing young sapling stands continues to be a problem.

A study was initiated in 1991 to regenerate white oak (*Quercus alba* L.) by underplanting oak seedlings under a shelterwood in a stand at the Paoli Experimental Forest (PEF) and one at the Martin State Forest in southern Indiana. Initially, survival and growth of white oak seedlings was good but 12 years after planting and 9 years after final removal of the shelterwood the white oak are being relegated to the lower crown classes in the developing forest canopy by more competitive yellow-poplar (*Liriodendron tulipifera* L.), white ash (*Fraxinus americana* L.), and black cherry (*Prunus serotina* Ehrh.).

Twelve years following planting, white oak survival ranged from 42-59 percent and tree height varied from 2.9-3.6 meters among the different study stands. However, the height of competition was greater than 5.0 meters. Therefore, following the 12th growing season half of each planting area was thinned to favor white oak and the other half remained unthinned. Overtopping trees around each white oak were removed by chainsaw felling so that each white oak crown had a “clear view of the sky”.

One year after thinning at the PEF site, survival decreased slightly for the thinned white oak trees, but there was no difference in survival between thinned and unthinned trees. Although, height growth of white oak was unaffected by the thinning, the proportion of white oak in codominant and dominant crown classes did increase at PEF. The proportion of unthinned white oaks in the upper crown classes continued to decrease. Thus, the increase of dominant and codominant white oaks is likely the most important result so far, one year following thinning. Thinning provides the surviving trees with sufficient light and growing space to remain competitive in future years. Thus thinned white oak trees should have greater competitive capacity and higher probabilities of being in the codominant and dominant crown classes in the future. In contrast, survival of unthinned trees is expected to decline as an increasing proportion of white oaks fall into the intermediate and suppressed classes.



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THIRTY-YEAR POST HARVEST SURVEY OF A RELICT CHESTNUT OAK STAND IN SOUTHERN ILLINOIS

Jessica A. Yeagle and John W. Groninger[†]

Chestnut oak (*Quercus prinus*) is common from Maine to Georgia and reaches its northwestern distribution limits in southern Illinois occurring in a few fragmented populations. In 1973 a silvicultural clearcut and post-harvest herbicide treatment was performed on one of the few chestnut oak stands located in southern Illinois, approximately 240 hectares. Prior to the harvest, a survey was taken of the stand identifying stand structure and composition. Fifty-two permanent plots were established using the posi-plot method. Post harvest surveys of the stand were performed using the same plots at 1, 3, and 18 years after the clearcut. In the summer of 2003, the 30-year post harvest survey was conducted. Using the existing plots, all trees ≥ 6.60 cm dbh were measured and identified by species within a radius of 11.35 m (0.04 ha) of plot center. Saplings ($2.54 \text{ cm} \leq X < 6.60$ dbh) within 11.35 m of plot center and seedlings (< 2.54 cm dbh) within a radius of 3.6 m (0.004 ha) of plot center were counted and tallied by species. Calculating density and basal area will summarize tree data from each plot. As a result of permanent plots, pre- and 30-year post-harvest stand composition can be related on an individual plot basis. Since little research has been done documenting the long-term effects of clearcutting from preharvested conditions this study is important for recording the successional patterns after a major disturbance. It is also valuable in showing the reproduction response of chestnut oak at the edge of its range and the regeneration success of other valuable timber species after harvest. Coarse woody debris was also inventoried to serve as a baseline for later research.

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DEVELOPMENT OF ELTP LAYER FOR THE HOOSIER NATIONAL FOREST, SOUTHERN INDIANA

Andriy V. Zhalnin, George R. Parker, and Patrick Merchant[†]

One of the critical components of decision making in natural resource management is ecological information. The US Forest Service adopted a national ecological classification hierarchy in 1993 with ELT (ecological land type) and ELTP (ecological land type phase) forming the lower levels of the hierarchy (ECOMAP 1993). This study examines the potential of computer mapping ELTPs for the Hoosier National Forest (HNF) located in southern Indiana.

Van Kley developed an ecological classification for the forest in 1993 that includes six ELTs and sixteen ELTPs in total for both subsections (Van Kley et al. 1995). The most important factors affecting classification at ELTP level were A-horizon depth and vegetation composition. An ELT layer based on physiographical conditions was mapped with GIS tools (Shao 1998).

An extensive ELTP sampling was performed in 2001 and 2002 within the four units of HNF that are situated within the Brown County Hills (BCH) and Crawford Upland (CUP) subsections according to the eastern United States classification (Keys, Jr., McNab, & Carpenter 1995). We used a 10 m resolution USGS DEM and a soil survey map as a source data layers. GIS layers were calculated with ArcView or ArcInfo tools. Each sample point received elevation, slope, aspect, curvature and soil type values from respective GIS layers.

We performed statistical analysis only for those ELTPs that occupy sites similar in physiography and differ only in vegetation and soils in classification description. Kruskal-Wallis test of landform variable means indicated a statistically significant variation ($P < 0.05$) among all ELTPs in elevation, aspect, slope and profile curvature both for BCH and CUP subsections. A pairwise Mann-Whitney test showed a significant difference ($P < 0.05$) in general and plane curvature between ELTPs 10 and 11 (BCH), and ELTPs 21 and 23 (CUP). Elevation difference was significant between ELTPs 20 and 21 (BCH), and ELTPs 50 and 51 (BCH, CUP). Chi-Square test of soil types derived from soil survey map units revealed significant difference ($P < 0.05$) in soil type constancy between ELTPs 50 and 51 (BCH, CUP); 10 and 12 (CUP); 20 and 21 (BCH, CUP); 20 and 23 (CUP); 21 and 22 (CUP); 60 and 61 (CUP).

We generated ELTP subunit map based on physiography characteristics defined in classification and used our survey points as well as information from statistical testing to map ELTPs characterized only by differences in vegetation and soils in original classification. Resulted map provides the basis for development of Land Type Associations using “bottom-up” approach.

This research was funded by the USDA Forest Service.

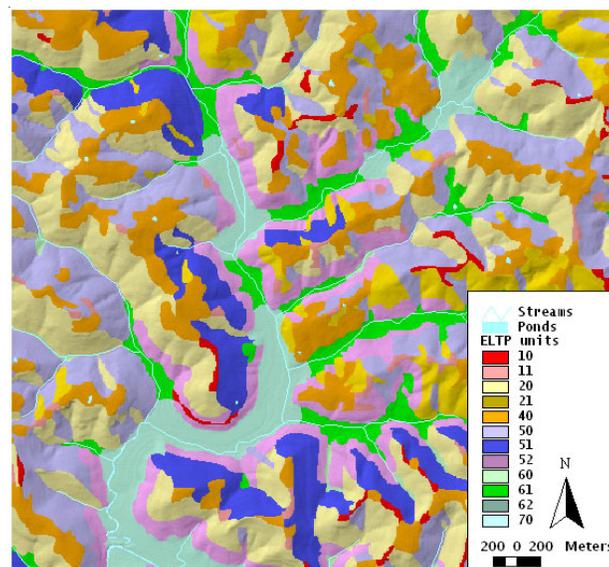


Figure 1.—An example of ELTP map, Lost River Unit, HNF.

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MODELING MOISTURE DISTRIBUTION OF A DRIED RED OAK LUMBER PACKAGE IN A HIGH HUMIDITY ENVIRONMENT

Minghui Zhang, Rado Gazo, and Daniel L. Cassens[†]

Many companies store dried lumber in an uncontrolled environment prior to use in manufacturing. They often ask - how long can lumber be stored outside and still be good to use? In this project, the relationship between the ambient environment and moisture content distribution throughout a package of dried red oak (*Quercus rubra*) lumber over time is analyzed. Thirty nine wireless moisture content transmitters were embedded in a packet of lumber 8 feet long, 42inch wide and 34 layers high. End coating was applied to both ends of each board prior to drying to 7 percent MC, but was removed from one end for this experiment to study the effect of end-coating. The environmental chamber conditions were set to EMC of 16 percent and moisture content changes were observed for several months.

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