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12th Central Hardwood Forest Conference

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FOREWORD

The Central Hardwood Forest stretches from the upper Southeast to the Great Lakes and from Arkansas to Massachusetts. It is an oak-dominated deciduous forest occurring in hilly to mountainous areas of this vast region. As such, it is the most extensive temperate deciduous forest in the world. The tree species present are well adapted to the seasonal climate changes and the moderate rainfall found in the region. The Central Hardwood Forest developed since the last ice age as forests reinvaded the region. Land-use practices impacting the region include those imposed by Native Americans as well as significant impacts from European settlers. These impacts include burning, grazing, land clearing, logging, fire control, wildlife management, and pest introductions. These practices and impacts have influenced, to a large degree, the composition and the area covered by these forests.

One-fourth of the population of the United States lives in this region and approximately 90 percent of the Central Hardwood Forest is owned by private interests comprised primarily of nonindustrial forest owners. The Central Hardwood Forest is biologically and spatially complex. The body of knowledge developed by scientists and practitioners on the biology and management of this forest is critical to the continued health and sustainability of this forest. The Conference provides a significant opportunity for scientists and practitioners to exchange information that will ultimately play an important part in the development of the Central Hardwood Forest.

History of the Central Hardwood Forest Conference

This Conference is the 12th in a series of biennial meetings that have been hosted by numerous universities and USDA Forest Service Experiment Stations in the Central Hardwood Forest Region including:

1976 Southern Illinois University
1978 Purdue University
1980 University of Missouri
1982 University of Kentucky
1985 University of Illinois
1987 University of Tennessee
1989 Southern Illinois University and the North Central Forest Experiment Station
1991 Pennsylvania State University and the Northeastern Forest Experiment Station
1993 Purdue University and the North Central Forest Experiment Station
1995 Northeastern Forest Experiment Station and West Virginia University
1997 University of Missouri and the North Central Forest Experiment Station
1999 University of Kentucky and the Southern Research Station

Conference Purpose

The purpose of this Conference has remained the same since its inception “To provide a forum for the exchange of information concerning the biology and management of central hardwoods by forest scientists from throughout the Central Hardwood Region of the Eastern United States.” As with previous conferences in this series, a wide range of subjects have been presented representing the range of research efforts underway in the region.

Central Hardwood Forest Conference—An Outlet for Peer-Reviewed Information

Since its beginning, the Central Hardwood Forest Conference has been an outlet for results of research focused on the forest itself or species that occur in the Central Hardwood Region. There were 32 oral presentations, 11abstracts, and 22 poster presentations accepted for the 12th Conference. Poster and oral presentation abstracts were accepted for publication along with full-length manuscripts. Manuscripts have undergone a peer review process by two to three anonymous reviewers. Reviewed manuscripts were returned to authors and revised electronic manuscripts were submitted for publication to the USDA Forest Service, Southern Research Station. In total, 7 percent of the manuscripts were rejected, 36 percent required major revision, 45 percent required minor revision, and 12 percent were accepted without revision. Papers were edited to a uniform format and type style; however, authors are responsible for the accuracy and content of their papers.

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Combined Session

CHANGES IN NATIONAL FOREST TIMBER SALES IN THE CENTRAL HARDWOOD REGION

William G. Luppold and John E. Baumgras[1]

Abstract—National forests contain only a small percentage of the sawtimber inventory in the Central Hardwood Region but many have proportionately more high quality timber than adjacent private lands. As a result, these forests are important to sawmill and other primary processors in this region. In this paper we analyze timber sales from 14 national forests in the Central Hardwood Region to determine the impact of changing forest polices on the forest-products industry. Since the mid-1980's, sales of hardwood and softwood sawtimber have been declining steadily, while the relative volumes of higher value species have increased. Because of this change in the mix of products sold and increasing prices of higher valued hardwood species and pine, total revenue from timber sales have declined moderately, while physical volumes sold have declined by more than 50 percent.

INTRODUCTION

The National Forest System was established to provide timber and protect watersheds for the citizenry of the United States. In recent years timber sales from these forests have declined in large part because of increasing public and political pressures. Recent policy changes with respect to the management of national forests have placed greater emphasis on watershed health, sustainable forest ecosystems, and recreation.[2] Still, the eastern national forests may be an important source of quality timber for industries proximate to these forests (Luppold and others 1998). In this paper we analyze changes in timber sales from 14 national forests in the Central Hardwood Region to determine the impact of changing forest policies on the forest-product industry in this region.

Specifically we analyze the volume and value of sawtimber sold by the national forests in the Central Hardwood Region in 1997, the degree to which timber sales have changed since 1985, and whether the changes in sales volume and revenue are consistent among these forests. Although our primary focus is hardwood sawtimber sales, sales of pulpwood and softwood sawtimber also are examined as these products are part of the total mix of timber products sold.

THE DATA

The data used in this study were developed from reports on timber cut and sold on national forests under sales and land exchanges, issued by national forests in Region 8 (southern) and Region 9 (eastern). These reports contain timber sales volumes and revenue for sawtimber, pulpwood, and other products sold by species or species group. Sawtimber as defined by the USDA Forest Service is hardwood trees larger than 11 inches in diameter at breast height (dbh) and softwood trees greater than 9 inches dbh. Hardwood pulpwood is 6 to 11 inches dbh while softwood pulpwood is 5 to 9 inches dbh. The reports were collected for the period beginning in 1985 and ending in 1997. Inventory data were obtained from the Forest Service's Eastwide Forest Inventory Data Base and the Northeastern Research Station's Forest Inventory and Analysis Unit (Hansen and others 1992).

It should be noted that there are variations in the cut and sold reports. For this study reports, were obtained for separate forests except for the states of North Carolina and Georgia. Cut-and-sold reports for North Carolina contain sales from the Pisgah, Nantahala, Croatan, and Uwharrie National Forests. The Pisgah and Nantahala are large, predominantly hardwood forests located in the western mountains of North Carolina; the Croatan and Uwharrie are small and primarily softwood forests located in the Piedmont and Coastal Plain Region. The reports for Georgia combine the Chattahoochee and the Oconee National Forest. The Chattahoochee is primarily a hardwood forest while the Oconee is primarily pine.

All volumes from cut-and-sold reports are in thousand board feet (MBF) International scale though many products are sold using other units of measurement. Sawtimber traditionally has been sold using the Scribner or International scale and pulpwood has been sold by the cord or hundred cubic feet (CCF). Since 1996, most forests in Region 8 recorded sawtimber and pulpwood sales volumes in cubic meters (J. Kirk Eichenberger, pers. commun.). The exception is the Daniel Boone, which began reporting in cubic meters in 1997. During this transition, the conversion of pulpwood from CCF to MBF for the cut-and-sold report changed from 0.77 to 0.55 MBF/CCF. Thus pulpwood sales volumes in Region 8 since 1996 must be multiplied by 1.4 to be consistent with sales volumes prior to 1996.

[1] Project Leaders, USDA Forest Service, Northeastern Reseach Station, 241 Mercer Springs Road, Princeton, WV 24740.

[2] Developed from "A gradual unfolding of a national purpose: a natural resource agenda for the 21st century," by Michael P. Dombeck, Chief of the USDA Forest Service, March 2, 1998.

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NATIONAL FORESTS OF THE CENTRAL HARDWOOD REGION

The Central Hardwood Region as defined by Hicks and others (1997), includes 14 national forests. These forests are listed geographically in Table 1 starting with the most southern forest on the eastern boundary of the region and ending with the most southern forest on the western boundary of the region. Although Ouachita National Forest in central Arkansas lies at the southwestern edge of the Central Hardwood Region, it was excluded from this study because it is predominantly southern pine. Also, the national forests in Virginia (Jefferson and George Washington) and North Carolina (Pisgah and Nantahala) were combined in Table 1 because parts of these forests overlap Forest Service survey regions.

The national forests of the Central Hardwood Region vary considerably with respect to species mix. Although the Chattahoochee has large volumes of red and white oak, eastern white pine is the most predominant species (single species or species group with the greatest volume). The Cherokee contains significant volumes of select red oak, white pine, and yellow-poplar. The combination of oak, yellow-poplar, and pine is also the composition of the North Carolina forests while the predominant species of the Virginia forests are the oaks and yellow-poplar. The Monongahela National Forest may be the most unusual national forest in the Central Hardwood Region because it includes relatively large volumes of black cherry, soft maple, and red spruce.

Although forest-survey statistics indicate that other red oaks (scarlet and southern red oaks) are the most predominant species group on the Daniel Boone, select white oak and yellow-poplar are the primary individual species. White oak and yellow-poplar also are the primary species on the Wayne while the Hoosier and Shawnee are oak/hickory forests with high proportions of select white oak.

The Mark Twain is the most fragmented forest in the Central Hardwood Region with nine noncontiguous districts. Although other red oaks account for 36 percent sawtimber volume, shortleaf pine is the most predominant species. Shortleaf pine is the most important species in the Ozark, though, select red and white oaks account for 34 percent of the sawtimber volume. The St. Francis is the only forest in the western portion of the Central Hardwood Region without a major pine component.

CHANGES IN NATIONAL FOREST TIMBER SALES

Volumes of hardwood timber sold from national forests in the Central Hardwood Region in 1997 are presented in Table 2.

Table 1—Sawtimber volume[a], hardwood component, and major species[b] of national forests in the Central Hardwood Region

National forest	State	Sawtimber volume	Hardwood component	Major species
		Million bf	*Percent*	
Chattahoochee[c]	GA	4311.1	63.7	WP[d], OWO, ORO, SRO
Cherokee	TN	3412.0	61.6	SRO, WP, YP, OP
Pisgah and Nantahala[e]	NC	7443.9	80.4	OWO, SRO, ORO, YP, SWO, WP, OP
Jefferson and G. Washington	VA	7803.0	83.9	OWO, SRO, ORO
Monongahela	WV	5181.6	83.7	SRO, BC, YP, SM, RS, OWO
Daniel Boone	KY	3437.2	85.0	ORO, SWO, YP, ORO
Wayne	OH	842.5	98.1	SWO, YP, OWO
Hoosier	IN	795.7	92.0	SWO, ORO, OWO, HK
Shawnee	IL	1020.0	95.9	ORO, SWO, HK
Mark Twain[f]	MO	3352.0	68.4	ORO, SLP
Ozark	AK	4030.2	70.9	SLP, SWO, SRO
St. Francis	AK	238.6	83.2	SWO, ORO, HK, YP

[a] Sawtimber volumes based on survey unit data extracted from the Eastwide Forest Inventory Data Base. Estimates are approximate since small portions of national forests may be in adjacent forest survey units.
[b] When combined, species equal or exceed 50 percent of the resource ranked in order of volume.
[c] Does not include the Oconee National Forest.
[d] SWO-select white oak, SRO-select red oak, OWO-other white oak, ORO-other red oak, YP-yellow-polar, HK-hickory, BC-black cherry, SM-soft maple, BE-beech, SLP-shortleaf pine, WP-eastern white pine, RS-red spruce, OP-other pine.
[e] Mountain survey region of North Carolina.
[f] Developed from Kingsley and Law 1991.

Table 2— National forest timber sales in 1997 and changes in sawtimber sales between 1985 and 1997 with respect to volume in the Central Hardwood Region

National forest	Volume of hardwood sawtimber sold in 1997	Change in volume of hardwood sawtimber sold	Volume of softwood sawtimber sold in 1997	Change in volume of softwood sawtimber sold
	MBF	*Percent*	*MBF*	*Percent*
Chattahoochee/ Oconee	1609.1	-73.5	2138.8	-92.7
Cherokee	3227.1	-75.6	7171.2	-36.7
Pisgah/Nantahala	6718.2	-74.7	7087.5	-38.5
Jefferson/ Washington	11136.5	-34.2	746.0	-88.0
Monongahela	9267.6	-50.1	211.4	-87.6
Daniel Boone	8070.0	-66.2	990.1	-87.2
Wayne/Hoosier	3802.2	-73.2	1870.3[a]	3116.3[a]
Shawnee	56.7	-96.1	8.4	-99.3
Mark Twain	25884.1	-36.9	12138.4	43.7
Ozark/St. Francis	5001.3	-30.5	34326.4	125.7

[a] Volume of softwood sold and increases in softwood sales were influenced by a salvage sale in 1997.

The Wayne and Hoosier National Forests are combined in Table 2 because they were operated as a single forest for 11 of the 13 years covered by this study. Similarly, timber sales on the Chattahoochee and Oconee and the Ozark and St. Francis National Forests are reported jointly.

Hardwood sawtimber sales on the Chattahoochee/Oconee, Cherokee and North Carolina (southern Appalachian) National Forests have declined by nearly 75 percent between 1985 and 1997. Except for the Cahattahoochee/Oconee, softwood sales have dropped less than hardwood sales. Much of the decline on the Chattahoochee/Oconee was caused by reduced sales on the Oconee. Although the southern Appalachian forests are predominantly hardwood, softwood timber was the major product sold from these forests in 1987.

Sales of hardwood sawtimber from the Jefferson/Washington, Monongahela, and Daniel Boone (mid Appalachian) also declined 34 to 66 percent. The volume of softwood sawtimber sold from these forests has declined by nearly 90 percent. In 1997, hardwood sales were 15 times greater than softwood sales in these forests.

Hardwood sawtimber sales for the Wayne/Hoosier declined by 73 percent while sales from the Shawnee have all but stopped. The large increases in softwood sawtimber from the Wayne/Hoosier resulted from salvage sales of uprooted and storm damaged trees. Prior to 1997, softwood sales were erratic but showed no downward or upward trend. Hardwood sawtimber sales also varied from year to year but have shown a decided downward trend; the decline was greatest in the Ohio (Wayne) section. Similar to the decline in the Wayne section of the Wayne/Hoosier, softwood sawtimber sales have all but ceased on the Shawnee National Forest.

The two Central Hardwood forests west of the Mississippi (Mark Twain and Ozark/St.Francis) have shown a moderate decline in hardwood sawtimber sales and large increases in softwood sawtimber sales. The increases in softwood were almost entirely those of shortleaf or other southern pines.

In the southern Appalachian forests, sales of hardwood pulpwood declined by 77 to 87 percent (Table 3). Sales of softwood pulpwood have declined by 27.3 percent on the Chattahoochee/Oconee and have increased more than 100 percent on the Cherokee. Pulpwood sales are influenced by sawtimber sales because sawtimber and pulpwood are sold together. The relative decline in hardwood pulpwood sales was greater than that for softwood pulpwood because of the greater decline in hardwood sawtimber sales. Still, the decline in relative sales of hardwood pulpwood has been greater than that of hardwood sawtimber and the decline in sales of softwood sawtimber was greater than that of softwood pulpwood.

Hardwood and softwood pulpwood sales have declined by 58 to 71 percent on the Virginia and West Virginia forests. Similar to other Mid- Appalachian forests, sawtimber and pulpwood sales from the Daniel Boone have declined by 49 to 87 percent; the decline was greatest in volume of softwood sawtimber and pulpwood.

Table 3— National forest timber sales in 1997 and changes in pulpwood sales between 1985 and 1997 with respect to volume in the Central Hardwood Region

National forest	Volume of hardwood pulpwood sold 1997	Change in volume of hardwood pulpwood sold	Volume of softwood pulpwood sold 1997	Change in volume of softwood pulpwood sold
	MBF	*Percent*	*MBF*	*Percent*
Chattahoochee/ Oconee	879.9	-87.5	8867.6	-27.3
Cherokee	2769.1	-72.2	4875.5	114.1
Pisgah/Nantahala	4932.8	-76.8	6248.1	29.5
Jefferson/ Washington	15943.2	-57.8	1640.0	-70.9
Monongahela	2702.7	-67.1	91.7	-86.0
Daniel Boone	2129.1	-49.3	646.3	-69.1
Wayne/Hoosier	1414.2	-77.4	2188.9	597.6
Shawnee	0.0	-100.0	3.2	-99.9
Mark Twain	0.0	-100.0	1320.2	192.3
Ozark/St. Francis	3779.2	-61.0	10991.3	-13.4

Similar to most other eastern national forests, hardwood pulpwood sales from the Wayne/Hoosier declined by nearly 80 percent; the 600-percent increase in softwood pulpwood sales for these forest is the result of a salvage sale. Sales of hardwood pulpwood have ceased on the Shawnee

Although sales of hardwood pulpwood have ceased on the Mark Twain, sales of softwood pulpwood have increased by more than 190 percent. Pulpwood sales volumes for both hardwood and softwood declined on the Ozark/St. Francis with the greatest decline in hardwood. Pulpwood sales from these forests vary from year to year, but the largest increase in sales of softwood pulpwood from the Mark Twain probably is abnormal because sales of this product had approached zero between 1989 and 1996.

CHANGES IN NATIONAL FOREST TIMBER SALES REVENUE

Although timber sales have declined sharply for most national forests, timber sales revenue has not declined as sharply and actually have increased on some forests (Table 4). Only sawtimber sales are shown in Table 4 because pulpwood's contribution to timber sales revenue is small for most forests in the Central Hardwood Region.

An examination of sales in the Appalachian portion of the Central Hardwood Region found that the change in revenue was related to the volumes of economically important species sold and the price growth of these species. The oaks are the most important species on the national forests in southern Appalachia. The sales volume of oak species from these forests decreased at a rate similar to that of total hardwood sales while price increased by 90 percent (North Carolina and Chattahoochee/Oconee) to 600 percent (Cherokee). These increases in price partially offset the decline in sales volume; still, revenues decreased by 30 to 54 percent between 1985 and 1997 (Table 4).

During the study period, red oak has been the most valuable species sold from national forests in Virginia. While total hardwood sawtimber sales declined by nearly 35 percent on these forests, sales of red and black oak increased by 44 percent. This increase in sales volume and a 391- percent increase in the price of red and black oak were the driving force behind the 191-percent increase in hardwood sawtimber revenues for these forests. Although red oak is the most predominant species on the Monongahela National Forest, most of the increases in sales revenue from this forest can be attributed to the steady sales volumes of black cherry and a 630-percent increase in the price of this species.

The oaks also are the most important species in the Daniel Boone National Forest, however, while the sales volume of white oak on these forests decreased by more than 50 percent, the sales volume of red oak increased by nearly 60 percent. The increase in red oak sales combined with increases of more than 300 percent in the prices of both red and white oak prices are the primary reasons why sales revenue did not decrease on the Daniel Boone.

The moderate increases in prices of red and white oaks on the Wayne/Hoosier and Shawnee forests were insufficient to offset a decrease in sales volume. By contrast, the 128-percent increase in the price of mixed oaks from the Mark Twain offset the 30-percent decline sales volume. The aggregate price of oak sold from the Ozark/St. Francis National Forests increased only slightly while prices of

Table 4—National forest sawtimber sales in 1997 and changes in sawtimber sales between 1985 and 1997 with respect to value in the Central Hardwood Region

National forest	Revenue of hardwood sawtimber sold in 1997	Change in revenue of hardwood sawtimber sold	Revenue of softwood sawtimber sold in 1997	Change in revenue of softwood sawtimber sold
	$1,000	*Percent*	*$1,000*	*Percent*
Chattahoochee/ Oconee	186.2	-38.3	254.4	-91.8
Cherokee	478.4	-30.1	973.1	8.6
Pisgah/Nantahala	658.9	-54.4	990.9	13.8
Jefferson/ Washington	2631.3	191.3	110.4	-64.6
Monongahela	3674.9	209.2	6.7	-85.1
Daniel Boone	872.1	6.3	78.1	-79.4
Wayne/Hoosier	589.1	-55.4	18.3	326.0[a]
Shawnee	6.4	-93.1	.3	-99.0
Mark Twain	3704.0	51.2	1500.3	158.5
Ozark/St. Francis	512.0	-32.6	9333.8	531.5

[a] Volume of softwood sold and increases in softwood sales were influenced by a salvage sale in 1997.

mixed hardwoods decreased by more than 50 percent. As a result, the decline in total sawtimber revenue was greater than that of sawtimber sales volume.

POTENTIAL IMPACT OF CONTINUAL REDUCTIONS IN SALES

A reduction or cessation of timber sales from national forests would seem to have little impact on the forest-products industry. In most of the states examined in this paper, the sawtimber volume on national forests is less than 10 percent of total sawtimber volume. However, national forest timber must be viewed not in aggregate but with specific users in mind.

Most of the national forests in the southern Appalachian Region are located near the North Carolina-Viginia wood-furniture industry. This industry is located in this region because of the availability of lumber. However, every southern Appalachian forest also is accessible by at least one interstate highway. Thus, the accessibility of these forests combined with their proximity to numerous urban centers may be another reason why this region has shown the greatest decline in timber sales.

Black cherry is the most valuable, commonly traded domestic hardwood species (Hardwood Market Report 1998) and rivals mahogany in importance in the manufacturing of traditional furniture. In the late 1980's the Monongahela National Forest contained about 4 percent of the nation's inventory of black cherry (Luppold and others 1998). All indications are that cherry has been harvested on private lands at a much faster rate than the national forests. As a result, the supply of cherry on these forests has increased.

The national forests in Ohio, Indiana, and Illinois contain large quantities of higher grade white oak timber (Luppold and others 1998). Although white oak is used by domestic secondary timber processors, much of the higher grade white oak from this region is exported to Europe and Asia. Reduced supplies of national forest timber from this region may not affect facilities that serve domestic customers as much as sawmills that sell to international customers.

The two national forests west of the Mississippi have substantial volumes of hardwood sawtimber. However, the market value of this timber is not as high as that of timber in other parts of the Central Hardwood Region. Still, flooring, pallet, and other industries may be dependent on sustained quantities of lumber from these forests.

SUMMARY AND CONCLUSIONS

The forests of the Central Hardwood Region range from the pine/oak forest of southern Appalachia to the primarily oak forest of the Central Plains to the pine/oak forest west of the Mississippi River. The southern Appalachian forests in Georgia, North Carolina, and Tennessee have shown greater declines in hardwood sawtimber sales than the forests of the mid-Appalachian Region of Virginia, West Virginia, and Kentucky. This difference may be a function of the proximity of the forests in southern Appalachia to large urban centers.

National forests in the Central Plains have shown large declines in hardwood sawtimber sales with the Shawnee showing the greatest decline. The only increases in timber product sales have resulted from a salvage sale of white pine on the Hoosier National Forest. The two national forests west of the Mississippi River have shown relatively moderate declines in sales of hardwood sawtimber sales and large increases in sales of softwood sawtimber.

Although timber sales have declined sharply for most national forests, timber revenues from these sales have not decreased nearly as much and have increased in some forests. These increases in revenue were greater on the Jefferson/Washington and Monongahela National Forests, the result of increased sales volume and prices of select red oak from the Jefferson/Washington and increased sales of black cherry from the Monongahela.

The reduction in timber sales on national forests has had an adverse economic impact on the industries that process and use timber products. This economic loss may be offset somewhat by gains that result from allocating timberlands to alternative uses such as watershed protection or recreation. It is important that policymakers understand economic tradeoffs associated with their decisions, and that future research focus on the impact of changing timber sales and management policies on recreational and ecological services. We believe that such analyses should be conducted on a forest-by-forest if not county-by-county basis to insure that the needs of rural communities located near national forests are considered.

REFERENCES

Hansen, M.H.; Frieswyk, T.; Glover, J.F.; Kelly, J.F. 1992. The eastwide forest inventory data base: user's manual. Gen. Tech. Rep. NC-151. U.S. Department of Agriculture, Forest Service. 48 p.

Hardwood Market Report. 1998. Memphis TN: Henry Jones Publisher. 76(7): 40 p.

Hicks, R.R. 1997. A resource at the crossroads: a history of the central hardwoods. In: Proceedings 11th central hardwood forest conference. Gen. Tech. Rep. U.S. Department of Agriculture, Forest Service: 1-22.

Kingsley, N.P.; Law, J.R. 1991. Timber resource of the Mark Twain National Forest. Res. Bull. NC-129. U.S. Department of Agriculture, Forest Service. 31 p.

Luppold, W.G.; Baumgras, J.E.; Sheffield, R.M. 1998. A comparative analysis of hardwood sawtimber quality on national forests versus adjacent lands. Forest Products Journal. 45(5): 51-56.

OAK PLANTING SUCCESS VARIES AMONG ECOREGIONS IN THE CENTRAL HARDWOOD REGION

Dale R. Weigel[1]

Abstract—This paper compares the results of planting northern red oak (*Quercus rubra* L.) and white oak (*Q. alba* L.) in the Ozark Highlands of Missouri to planting northern red oak in the Shawnee Hills and Highland Rim of southern Indiana. In both regions, seedlings were planted beneath shelterwoods thinned to 60 percent stocking. Black oak site index for Missouri planting sites was 19 meters (at an index age of 50 years) and 23 meters for Indiana sites. A total of 5,120 seedlings were planted. Three years after planting the shelterwoods were removed. Six nursery treatments were applied to growing 2+0 northern red oak stock in which seedlings were: (1) not undercut nor top-clipped, (2) not undercut but top-clipped the spring before planting, (3) undercut the first year in the nursery but not top-clipped, (4) undercut the first year and top-clipped, (5) undercut both years in the nursery but not topped clipped, and (6) undercut both years and top-clipped. Those treatments plus two additional treatments were applied to growing 3+0 white oak nursery stock in which seedlings were: (1) undercut the second year but not top-clipped, and (2) undercut the second year and top-clipped. Thirteen years after planting, survival of northern red oak ranged from 13 to 26 percent in Indiana plantings and from 50 to 77 percent in Missouri plantings, depending on treatment. Eleven years after planting, survival of white oak in Missouri ranged from 40 to 85 percent. Because survival by itself may not accurately reflect planted tree success, logistic regression was used to derive dominance probabilities for planted oaks. Accordingly, a planted tree was deemed dominant (and thus competitively successful) if it attained 80 percent of the mean height of dominant competitors a specified number of years after planting. In the Missouri plantings, estimated dominance probabilities for northern red oak increased with time since shelterwood removal and with increasing initial basal caliper of seedlings. After 13 years, probabilities for red oak seedlings with an initial caliper of 15 millimeters ranged from 0.45 to 0.60, depending on nursery treatment. Estimated probabilities for white oak did not change significantly with time. After 11 years, probabilities for white oak seedlings with an initial caliper of 15 millimeters ranged from 0.36 to 0.77. In contrast, estimated dominance probabilities for red oaks planted in Indiana declined with time. After 13 years, probabilities for seedlings 15 millimeters in initial caliper ranged from 0.02 to 0.04, depending on nursery treatment. Although effects of undercutting and top-clipping varied by species and planting region, their joint effects generally resulted in higher success probabilities than when neither treatment was applied. Low dominance probabilities for trees planted on Indiana sites resulted from high mortality and slow growth related to suppression by yellow-poplar (*Liriodendron tulipifera* L.) and aspen (*Populus* spp). In addition to nursery treatment and initial seedling size effects, the results emphasize the importance of recognizing variation in the outcome of interspecific competition among ecoregions and associated temporal changes in dominance probabilities of planted oaks.

INTRODUCTION

Many forests of the Central Hardwood region are dominated by oaks. However, regenerating oaks naturally has often been unsuccessful (Crow 1988, Lorimer 1989). Planting may be one way to overcome this problem. Numerous papers have been presented in previous Central Hardwood Forest Conferences dealing with oak planting (e.g., Bardon and Countryman 1993, Lantagne 1995, Larrick and others 1997, McNeel and others 1993, Rathfon and others 1995, Teclaw and Isebrands 1993, Weigel and Johnson 1997).

The shelterwood method can be used to create conditions favorable to regenerating oaks (Dey and Parker 1996, Hannah 1987, Johnson and others 1989, Loftis 1990). Combining oak planting with the shelterwood method offers an alternative to relying exclusively on natural regeneration (Johnson and others 1986). Shelterwoods of appropriate density can provide planted oaks with sufficient light and time for them to reestablish and expand their root systems before final shelterwood removal (Dey and Parker 1996, 1997). The planted oaks then can successfully compete with other established tree reproduction after the shelterwood is removed. However, the success of the method depends on the size and type of nursery stock that is planted (Johnson 1984, Weigel and Johnson 1998a, 1998b).

The results from two oak planting studies are reported here. They deal with northern red oak (*Quercus rubra* L.) and white oak (*Quercus alba* L.) planted beneath a shelterwood and the subsequent removal of the shelterwood. The objective of the studies was to develop methods to increase success of planted northern red oak and white oak seedlings at different locations in the central hardwood region.

METHODS

Results were obtained from two studies that included 3,840 two-year-old northern red oak seedlings (study 1) and 1,280 three-year-old white oak seedlings (study 2). Seedlings were planted under oak or mixed oak stands thinned from below to 60 percent stocking based on Gingrich's (1967) stocking equation.

Nursery Phase

Half of the northern red oak seedlings were grown in the Vallonia State Forest Nursery in Indiana and half in the George O. White State Forest Nursery in Missouri. The

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planted trees represented 10 seed sources, 5 stand collections from Missouri and 5 stand collections from Indiana. White oak seedlings were grown in the George O. White State Forest Nursery in Missouri. The planted trees represented 10 seed sources, 5 stand collections from Missouri and 5 stand collections from Mississippi. The objective of including seed source as a source of variation was primarily to ensure some genetic diversity into the study and account for that variation. The objective was not to identify superior seed sources. Nursery and seed source for northern red oak, 13 years after planting, and seed source for white oak, 11 years after planting, did not have a significant affect on survival ($p > 0.05$) and where therefore combined during analysis.

Northern red oak seedlings received one of three undercutting treatments in the nursery bed: (1) not undercut (U_0), (2) undercut during the first growing season (U_1), and (3) undercut the first and second growing seasons (U_{12}). White oak seedlings received one of four undercutting treatments in the nursery: (1) not undercut (U_0), (2) undercut during the first growing season (U_1), (3) undercut during the second growing season (U_2), and (4) undercut during the first and second growing seasons (U_{12}). Seedlings were undercut at a depth of 15 centimeters during mid to late June after the completion of one or two flushes of shoot growth. Those undercut the second year were undercut at a depth of 20 centimeters.

After spring lifting in early April 1984 and before planting, the tops of half the northern red oak seedlings in each undercutting treatment were cut off ("top-clipped") 20 centimeters above the root collar (C_1) and the other half were left intact (C_0). White oak seedlings were grown for 3 years and either fall lifted in November 1984 or spring lifted in early March 1985. Before planting, half the white oak seedlings in each undercutting treatment were top-clipped 20 centimeters above the root collar (C_1) and the other half left intact (C_0). Taproots and lateral roots of all seedlings were pruned to a common length of 25 centimeters below the root collar after lifting. The initial caliper (basal diameter measured 2.5 centimeters above the root collar) of each seedling was measured to the nearest 0.1 millimeter and recorded (table 1).

Planting Sites

Two Indiana sites were selected for planting northern red oak: Paoli Experimental Forest and the Pleasant Run Unit of the Hoosier National Forest. These sites were located in the Shawnee Hills and the Highland Rim, respectively, of the Interior Low Plateau as defined by McNab and Avers (1994). Sites were dominated by mixed oaks and yellow-poplar; black oak site index was 23 meters at an index age of 50 years based on Carmean's (1971) site index curves. In southern Missouri, two sites were selected for the northern red oak plantings and two sites for the white oak plantings on the Sinkin Experimental Forest, which lies within the Central Plateau Subsection of the Ozark Highlands as defined by McNab and Avers (1994). The sites were dominated by black oak and white oak. Black oak site index was 19 meters at an index age of 50 years based on McQuilkin's (1974) site index table.

Table 1—Mean, standard deviation, and range of initial seedling caliper for northern red oak and white oak

Species	Mean	Standard deviation	Minimum	Maximum
	--------	--------	*mm*	--------
Northern red oak	12.3	3.35	3.3	25.4
White oak	12.9	4.94	2.6	37.9

Outplanting

Northern red oak seedlings were outplanted in April 1984 at a spacing of 1- x 1-meter in a randomized block design with eight replications. Seedlings in Missouri were bar planted while those in Indiana were planted in auger holes. In April 1985 the white oak seedling were bar planted at 1- x 1-meter spacing. On all planting sites all woody stems between 2 and 4 centimeters d.b.h. and all stumps created by the shelterwood cut were treated with an herbicide (Tordon RTU) before planting. After three growing seasons (during the winter of 1986-1987 for northern red oak and during the winter of 1987-1988 for white oak), the shelterwood was completely removed. All stumps created by the final overstory removal were treated with an herbicide (Tordon RTU). Planted tree heights and survival were measured and recorded annually or biennially for 13 years for northern red oak and 11 years for white oak. For northern red oak in Indiana, the heights of dominant competitors were also measured in 1990, 1991, 1993, and 1996 (7, 8, 10, and 13 years after planting, respectively). In Missouri heights were not measured in 1993. For white oak, the heights of dominant competitors were also measured in 1990, 1992, 1993, and 1995 (6, 8, 9, and 11 years after planting, respectively). The tallest woody competitor, within a 1-meter radius of every fourth planted tree, was measured to determine mean heights of dominant competitors on each study area.

Data Analysis

Analysis of variance was used to determine influence of nursery treatments on survival and height growth. Mean separations were performed using Student-Newman-Keuls multiple range test. All treatment effects were tested at the 0.05 level. Logistic regression analysis (SAS 1989) was used to estimate the probability that a seedling of a given initial caliper would equal or exceed 80 percent of the mean height of dominant competitors. Models were developed for northern red oak in Indiana, northern red oak in Missouri, and white oak in Missouri. Analysis was done at 13 years for northern red oak and 11 years for white oak.

RESULTS

Survival after the first growing season for both species ranged from 100 percent to 87 percent (figure 1). Survival decreased for the 3 years while the oak seedlings were beneath the shelterwood. The U_0C_0 treatment, significantly different from all other treatments, showed the lowest survival for both species with white oak the lowest at 45

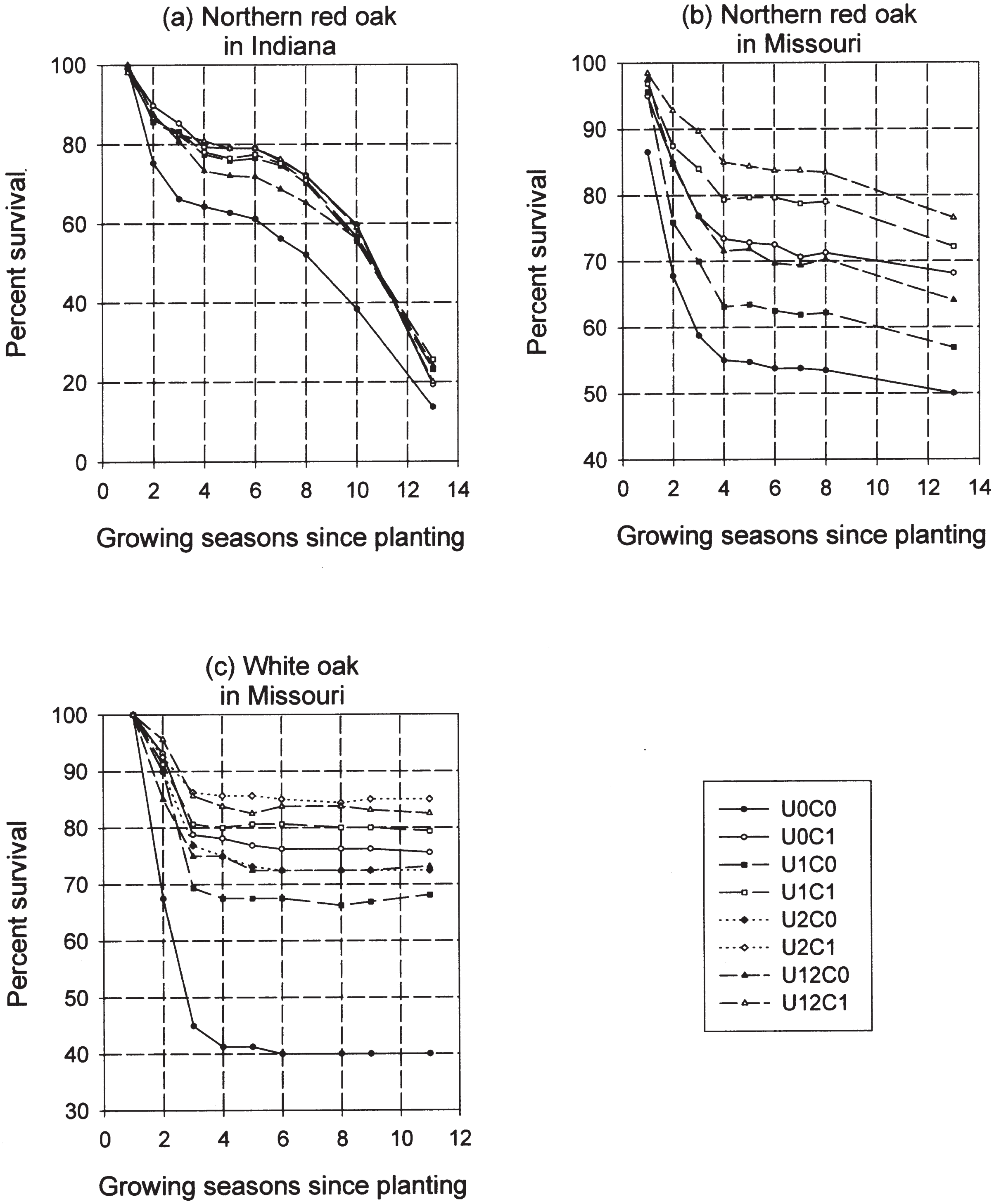

Figure 1—Survival trends for planted northern red and white oak. Seedling treatments: U_0C_0 = not undercut, not top-clipped; U_0C_1 = not undercut, top-clipped; U_1C_0 = undercut first year, not top-clipped; U_1C_1 = undercut first year, top-clipped; U_2C_0 = undercut second year, not top-clipped; U_2C_1 = undercut second year, top-clipped; $U_{12}C_0$ = undercut first and second year, not top-clipped; $U_{12}C_1$ = undercut first and second year, top-clipped.

percent. Survival for all remaining treatments for the two species was above 65 percent. Following complete overstory removal, survival remained constant for white oak ending between 40 and 85 percent 8 years after overstory removal with the U_0C_0 treatment once again significantly different from all other treatments. For northern red oak, survival remained fairly constant for the first 5 years but then decreased. In Indiana, 10 years after complete overstory removal, survival dropped to between 13 and 26 percent. The U_0C_0 treatment was significantly different for all treatments except U_0C_1 and $U_{12}C_1$. For Missouri the survival rate ranged from 50 to 77 percent with the U_0C_0 treatment significantly different for all treatments except U_1C_0.

During the early years, mean heights of survivors of white oak and northern red oak in both Missouri and Indiana differed by treatments. Those seedlings not top-clipped showed no net height growth during the first 3 to 5 years following outplanting (figure 2). However, seedlings top-clipped showed a positive height growth so that by the third to sixth growing season following outplanting there was no

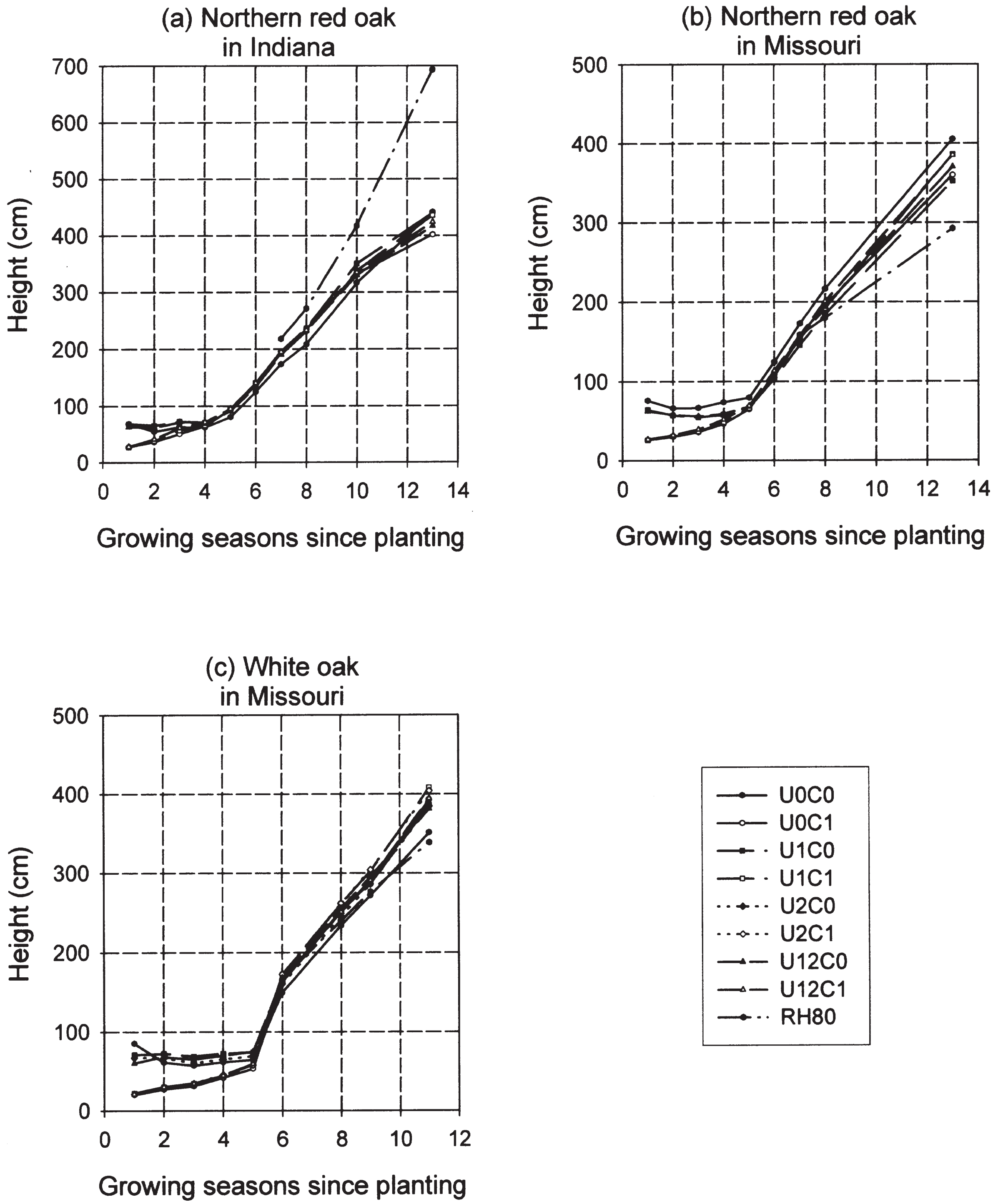

Figure 2—Mean heights of surviving planted northern red and white oak compared to the height of 80 percent of the competition height. Seedling treatments: U_0C_0 = not undercut, not top-clipped; U_0C_1 = not undercut, top-clipped; U_1C_0 = undercut first year, not top-clipped; U_1C_1 = undercut first year, top-clipped; U_2C_0 = undercut second year, not top-clipped; U_2C_1 = undercut second year, top-clipped; $U_{12}C_0$ = undercut first and second year, not top-clipped; $U_{12}C_1$ = undercut first and second year, top-clipped. RH80 (relative height 80) represents 80 percent of the mean height of dominant competitors.

significant difference ($p > 0.05$) in height between top-clipped and non-top-clipped seedlings.

Eight years after complete overstory removal (11 years after planting) white oak top-clipped treatments out grew non-top-clipped treatments ($p < 0.05$), with the U_0C_0 treatment showing the poorest performance ($p < 0.05$). For northern red oak, 10 years after complete overstory removal (13 years after planting), the U_0C_0 treatment outperformed all other treatments. Heights for northern red oak were not significantly different ($p > 0.05$). The difference between treatments for both species was less than 60 centimeters.

A planted oak was considered dominant if it attained 80 percent of the height of the competition. In Missouri, 8 years (white oak) and 10 years (northern red oak) after complete overstory removal all treatments were taller than 80 percent of the competition height. However, in Indiana all treatments for northern red oak were outgrown by woody competitors. The best treatment result was over 2.5 meters shorter than 80 percent of the woody competitor height.

Seedling survival and mean height of surviving trees do not always equate to seedling success. Moreover, such averages do not take initial seedling caliper into account. Logistic regression analysis therefore was used to estimate the probability that a seedling of a given initial caliper would equal or exceed 80 percent of the mean height of dominant competitors up to 10 years (northern red oak) or 8 years (white oak) following shelterwood removal. The resulting values were termed dominance probabilities because they express the likelihood that a planted seedling of a given treatment and initial size would be dominant or codominant (table 2).

Logistic regression results produced dominance probabilities that varied by species and ecoregion. Dominance probabilities increased with increasing initial seedling caliper for both species (figure 3). Northern red oak in Missouri showed increased dominance probabilities with time since shelterwood removal while in Indiana they showed decreased dominance probabilities with time since shelterwood removal. For white oaks in Missouri, dominance probabilities remained constant with time since shelterwood removal. Within a given undercutting treatment group, dominance probabilities for a given initial basal caliper are larger for clipped than for unclipped seedlings. The U_0C_0 treatment for both species produced the lowest dominance probabilities.

DISCUSSION

Survival rates after the first growing season were higher than that reported by Crunkilton and others (1989) for northern red oak in Missouri. Six years after outplanting, survival rates were similar to those reported by Gordon and others (1995). McGee and Loftis (1986) reported northern red oak and black oak survival, during a similar growing period, comparable to those reported here. Much of the mortality that did occur during the shelterwood period may have been attributable to suboptimal light levels, associated shoot dieback, and the death of seedlings of initially poor physiological quality.

Crunkilton and others (1989), Gordon and others (1995), McGee and Loftis (1986), and Zaczek and others (1993, 1997) reported height growth rates similar to or slightly less than those reported here. Possible logging damage following complete shelterwood removal could partially explain slow height growth.

Although survival and height growth of planted seedlings are important to their success, they do not always tell the complete story. The important end result is producing a competitively successful seedling; survival and height growth are not always indicators of a competitively

Table 2—Logistic regression models for dominance probabilities that a planted oak seedling will be dominant or codominant

	Parameter estimates[a]									
Species	b_0	b_1	b_2	b_3	b_4	b_5	b_6	b_7	p[b]	N[c]
White oak	-3.7414	1.1667	0.6984	0.9590	0.8022				0.10	5120
NRO in Missouri	-1.3070	0.5248		0.2484	0.3783			-3.2712	0.85	5760
NRO in Indiana	-1.7829	1.2593	0.4743	0.4060	0.2699	0.6659	-0.0825		0.21	7680

[a] Regression models are of the form $P = \{1+\exp[-(b_0 + b_1X_1 + b_2X_2 + b_3X_3 + b_4X_4 + b_5X_5 + b_6X_6 + b_7X_7)]\}^{-1}$, where P is the estimated probability that a planted oak seedling will produce a successful seedling; X_1 is the natural log of initial seedling caliper in millimeters; X_2 = 0 if not undercut, undercut second year, or both years, = 1 if undercut first year; X_3 = 0 if not undercut or undercut first year, = 1 if undercut second year or both years; X_4 = 0 if not top-clipped, = 1 if top-clipped; X_5 = years since shelterwood removal; X_6 = (years since shelterwood removal)^2; X_7 = 1/years since shelterwood removal. All parameter estimates differ significantly from zero at $\propto = 0.05$.
[b] The probability (p) that estimated dominance probabilities differ from the observed based on the Hosmer-Lemeshow chi-square test (Homer and Lemeshow 1989).
[c] N = number of seedlings planted • number of years competition measured (4 times for white oak and northern red oak in Indiana and 3 times for northern red oak in Missouri).

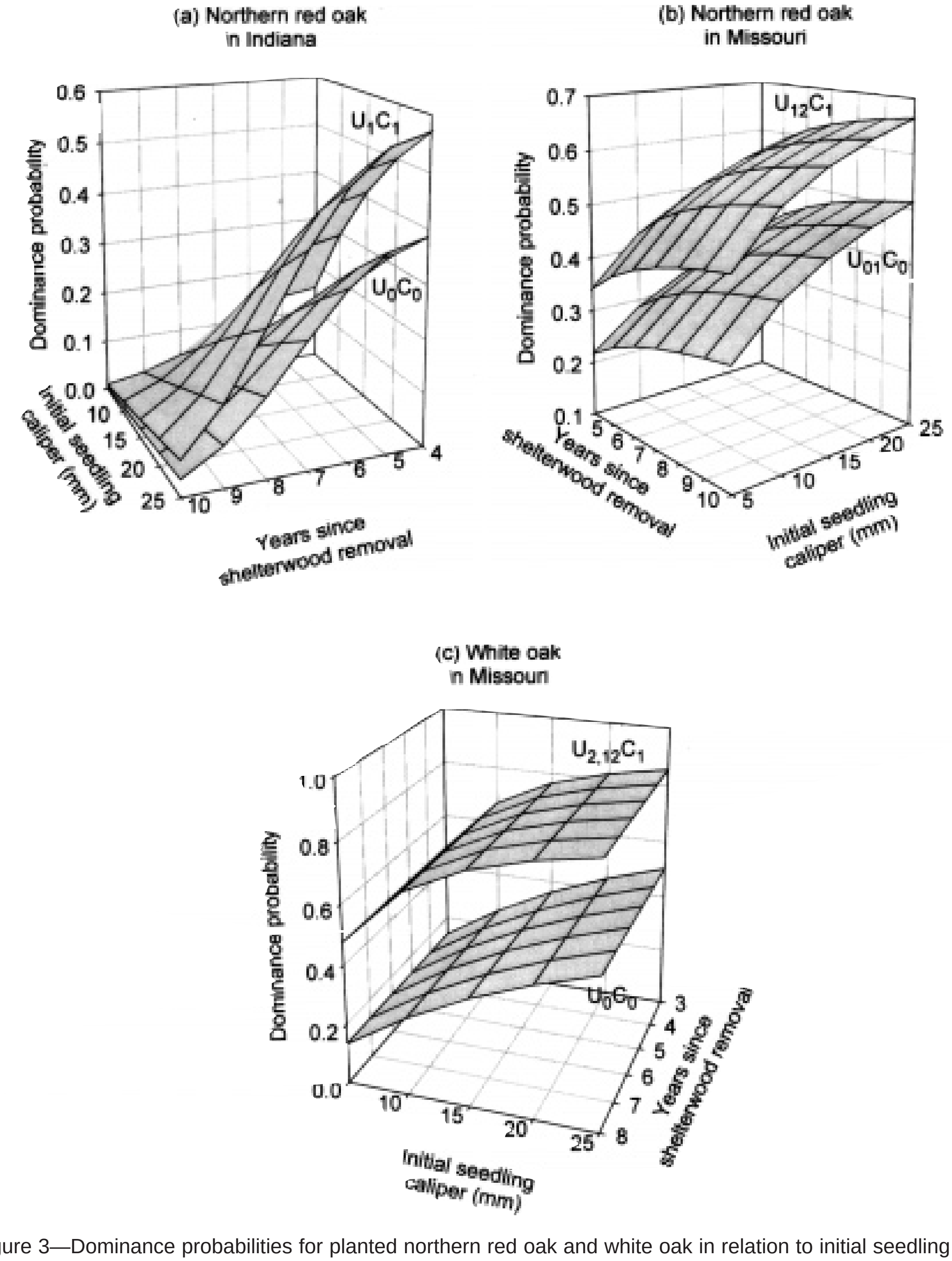

Figure 3—Dominance probabilities for planted northern red oak and white oak in relation to initial seedling caliper and years since shelterwood removal. For both regions and species, success probabilities are shown for the "best" and "worst" combination of undercut (U) and shoot clipping (C) treatments. Seedling treatments: U_0C_0= not undercut, not top-clipped; $U_{01}C_0$ = not undercut or undercut the first year, not top-clipped; U_1C_1 = undercut first year, top-clipped; $U_{12}C_1$ = undercut first and second year, top-clipped; $U_{2,12}C_1$ = undercut second year or first and second year, top-clipped. Note change in axis orientation between a and c and b.

successful seedling. Dominance probabilities give an indication of success by taking into account initial seedling caliper and seedling success against competition. Therefore, dominance probabilities integrate survival and growth into a single value which is silviculturally useful.

Different dominance probabilities between ecoregions can be explained by the relatively high site quality and the presence of rapid growing intolerant yellow-poplar (*Liriodendron tulipifera* L.) and aspen (*Populus* sp.) reproduction in Indiana. Those species comprised 75 percent of woody competitors on the Indiana planting sites.

Yellow-poplar (Beck 1990) and aspen (Perala 1990) are absent in the Ozark Highlands of Missouri. This height advantage of yellow-poplar and aspen over northern red oak in Indiana was at least 2.5 meters 10 years after shelterwood removal (figure 2). With the absence of yellow-poplar and aspen in Missouri, northern red oak was able to outgrow the competition and increase its dominance probabilities. Missouri woody competitors are characterized by low survival rates and limited size development relegating them to the sapling and reproduction layers (Dey and others 1996). However, in Indiana northern red oak was unable to compete with yellow-poplar and aspen which remain dominant throughout the rotation and thus its dominance probabilities decreased to near zero. Similar results were reported by McGee and Loftis (1986) in the southeast. White oak was able to compete successfully with woody competitors from time of shelterwood removal and thus its dominance probabilities remained constant.

Current results indicate that plantings in the Missouri Ozark Highlands will produce dominant and codominant trees. Successful plantings will not result in the two Indiana ecoregions. To improve planting results, in ecoregions dominated by rapid growing persistent woody competitors, post planting competition control may be needed. Burning the plantings while beneath the shelterwood (Brose and Van Lear 1998) or within 4 years following overstory removal should reduce the yellow-poplar competition. Application of herbicides to woody competitors 3-4 years following overstory removal should also provide the oaks with growing space. Repeated applications of competition control in these ecoregions may be necessary to maintain the planted oak in a dominant or codominant position.

The results of this research emphasize the importance of recognizing variation in competition among ecoregions where oak are to be planted. While a prescription for planting oak in one ecoregion may be successful, the same prescription in another ecoregion may result in entirely different results.

REFERENCES

Bardon, R.E.; Countryman, D.W. 1993. Survival and growth for the first-growing season of northern red oak (*Quercus rubra* L.) seedlings underplanted in mixed upland hardwood stands in south central Iowa. In: Gillespie, A.R.; Parker, G.R.; Pope, P.E.; Rink, G., eds. Proceedings, 9th central hardwood forest conference; 1993 March 8-10; West Lafayette, IN. Gen. Tech. Rep. NC-161. U.S. Department of Agriculture, Forest Service: 195-209.

Beck, D.E. 1990. Yellow-poplar. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. Agric. Handb. 654. U.S. Department of Agriculture, Forest Service: 406-416. Vol. 2.

Brose, P.H.; Van Lear, D.H. 1998. Response of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Canadian Journal of Forest Research. 28(3): 331-339.

Carmean, W.H. 1971. Site index curves for black, white, scarlet and chestnut oaks in the Central States. Res. Pap. NC-62. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 8 p.

Crow, T.R. 1988. Reproductive mode and mechanisms for self-replacement of northern red oak (*Quercus rubra*)-a review. Forest Science. 34: 19-40.

Crunkilton, D.D.; Garrett, H.E.; Pallardy, S.G. 1989. Growth of northern red oak seedlings planted in a central Missouri clearcut and shelterwood. In: Miller, J.H., comp. Proceedings of the 5th biennial southern silvicultural research conference; 1988 November 1-3; Memphis, TN. Gen. Tech. Rep. SO-74. U.S. Department of Agriculture, Forest Service: 81-85.

Dey, D.C.; Johnson, P.S.; Garrett, H.E. 1996. Modeling the regeneration of oak stands in the Missouri Ozark Highlands. Canadian Journal of Forest Research. 26(4): 573-583.

Dey, D.C.; Parker, W.C. 1996. Regeneration of red oak (*Quercus rubra* L.) using shelterwood systems: ecophysiology, silviculture and management recommendations. For. Res. Info. Pap. 126. Sault Ste. Marie, ON: Ontario Ministry of Natural Resources; Ontario Forest Research Institute. 59 p.

Dey, D.C.; Parker, W.C. 1997. Overstory density affects field performance of underplanted red oak (*Quercus rubra* L.) in Ontario. Northern Journal of Applied Forestry. 14(3): 120-125.

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

Gordon, A.M.; Simpson, J.A.; Williams, P.A. 1995. Six-year response of red oak seedlings planted under a shelterwood in central Ontario. Canadian Journal of Forest Research. 25: 603-613.

Hannah, P.R. 1987. Regeneration methods for oaks. Northern Journal of Applied Forestry. 4(2): 97-101.

Hosmer, D.W.; Lemeshow, S. 1989. Applied logistic regression. New York: Wiley. 307 p.

Johnson, P.S. 1984. Responses of planted northern red oak to three overstory treatments. Canadian Journal of Forest Research. 14(4): 536-542.

Johnson, P.S.; Dale, C.D.; Davidson, K.R.; Law, J.R. 1986. Planting northern red oak in the Missouri Ozarks: a prescription. Northern Journal of Applied Forestry. 3(1): 66-68.

Johnson, P.S.; Jacobs, R.D.; Martin, J.A.; Godel, E.G. 1989. Regenerating northern red oak: three successful case histories. Northern Journal of Applied Forestry. 6(4): 174-178.

Lantagne, D.O. 1995. Effects of tree shelters on planted red oaks after six growing seasons. In: Gottschalk, K.W.; Fosbroke, S.L., eds. Proceedings, 10th central hardwood forest conference; 1995 March 5-8; Morgantown, WV. Gen. Tech. Rep. NE-197. U.S. Department of Agriculture, Forest Service: 515-521.

Larrick, D.S.; Bowersox, T.W.; Storm, G.L.; Tzilkowski, W.M. 1997. Two-year survival and growth of artificial northern red oak regeneration at Gettysburg National Military Park. In: Pallardy, S.G.; Cecich, R.A.; Garrett, H.G.; Johnson, P.S., eds. Proceedings, 11th central hardwood forest conference; 1997 March 23-26; Columbia, MO. Gen. Tech. Rep. NC-188. U.S. Department of Agriculture, Forest Service. 349 p.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the Southern Appalachians. Forest Science. 36(4): 917-929.

Lorimer, C.G. 1989. The oak regeneration problem: new evidence on causes and possible solutions. For. Resour. Anal. 8, Publ. R3484. Madison, WI: University of Wisconsin, Department of Forestry. 31 p.

McGee, C.E.; Loftis, D.L. 1986. Planted oaks perform poorly in North Carolina and Tennessee. Northern Journal of Applied Forestry. 3(3): 114-116.

McNab, W.H.; Avers, P.E., comps. 1994. Ecological subregions of the United States: section descriptions. Admin. Publ. WO-WSA-5. Washington, DC: U.S. Department of Agriculture, Forest Service. 267 p.

McNeel, C.A.; Hix, D.M.; Townsend, E.C. 1993. Survival and growth of planted northern red oak in northern West Virginia. In: Gillespie, A.R.; Parker, G.R.; Pope, P.E.; Rink, G., eds. Proceedings, 9th central hardwood forest conference; 1993 March 8-10; West Lafayette, IN. Gen. Tech. Rep. NC-161. U.S. Department of Agriculture, Forest Service: 222-228.

McQuilkin, R.A. 1974. Site index prediction table for black, scarlet, and white oaks in southeastern Missouri. Res. Pap. NC-108. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 8 p.

Perala, D.A. 1990. Quaking aspen. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. Agric. Handb. 654. U.S. Department of Agriculture, Forest Service: 555-569. Vol. 2.

Rathfon, R.A.; Kaczmarek, D.J.; Pope, P.E. 1995. Site preparation for red oak plantation establishment on old field sites in southern Indiana. In: Gottschalk, K.W.; Fosbroke, S.L., eds. Proceedings, 10th central hardwood forest conference; 1995 March 5-8; Morgantown, WV. Gen. Tech. Rep. NE-197. U.S. Department of Agriculture, Forest Service: 349-362.

SAS Institute Inc. 1989. SAS/STAT user's guide. Version 6, 4th ed. Cary, NC: SAS Institute, Inc. 846 p. Vol. 2.

Teclaw, R.M.; Isebrands, J.G. 1993. Artificial regeneration of northern red oak in the Lake States with a light shelterwood: a departure from tradition. In: Gillespie, A.R.; Parker, G.R.; Pope, P.E.; Rink, G.; eds. Proceedings, 9th central hardwood forest conference; 1993 March 8-10; West Lafayette, IN. Gen. Tech. Rep. NC-161. U.S. Department of Agriculture, Forest Service: 185-194.

Weigel, D.R.; Johnson, P.S. 1997. Planting oaks in the central hardwood region: a shelterwood approach. In: Pallardy, S.G.; Cecich, R.A.; Garrett, H.G.; Johnson, P.S., eds. Proceedings, 11th central hardwood forest conference; 1997 March 23-26; Columbia, MO. Gen. Tech. Rep. NC-188. U.S. Department of Agriculture, Forest Service. 151 p.

Weigel, D.R.; Johnson, P.S. 1998a. Planting white oak in the Ozark Highlands: a shelterwood prescription. Tech. Brief TB-NC-5. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 8 p.

Weigel, D.R.; Johnson, P.S. 1998b. Planting northern red oak in the Ozark Highlands: a shelterwood prescription. Tech. Brief TB-NC-6. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 8 p.

Zaczek, J.J.; Steiner, K.C.; Bowersox, T.W. 1993. Performance of northern red oak planting stock. Northern Journal of Applied Forestry. 10(3): 105-111.

Zaczek, J.J.; Steiner, K.C.; Bowersox, T.W. 1997. Northern red oak planting stock: 6-year results. New Forests. 13: 177-191.

EFFECTS OF FROST ON HARDWOOD REGENERATION IN NORTHERN WISCONSIN

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Abstract—In northern Wisconsin, frosts often occur during the early growing season when shoot and leaf development are beginning and trees are most susceptible to freezing temperatures. We measured the intensity and duration of frosts in late May and early June 1994 in a vertical, 10 meter temperature profile in a clearcut, a stand with 50 percent crown cover, and a control. In addition, we monitored shoot development of northern red oak (*Quercus rubra* L), white ash (*Fraxinus americana* L.), ironwood (*Carpinus caroliniana* Walt.), sugar maple (*Acer saccharum* Marsh.), and aspen—both trembling (*Populus tremuloides* Michx.) and big tooth (*Populus grandidentata* Michx.) in these stands and in a stand with 75 percent crown cover. We also assessed frost damage on a broader scale as it was related to overstory condition, distance from clearcut edge, and topographic position in two areas with clearcut and partially cut hardwood forests. The frost damage was more severe in the clearcut, less in the 50 percent crown cover, and did not occur in the 75 percent crown cover and control. Northern red oak and white ash shoots were severely affected by frost with death of all new growth in the clearcut; ironwood was slightly damaged; and sugar maple and aspen were apparently unaffected by frost. All trees with frost damage produced new shoots; the location of the buds that flushed depended on the severity of frost damage. In addition to differences among overstory treatments, frost damage differed with topographic positions and with depth-of-edge between areas with different temperature environments. Silvicultural implications of our observations are discussed.

INTRODUCTION

Forest succession patterns are determined by many interrelated factors acting at different spatial and temporal scales. The most obvious factors, such as site productivity, herbivory, plant species growth rates, vegetative reproduction, and seed availability, are commonly used to explain the variation in composition and development of forest ecosystems. Other physical and biological factors may also be important, but we know little about their occurrence in space and time. Late spring frosts are one factor that may be more important than expected for some species and sites. Frost has long been implicated as a factor in hardwood regeneration and growth, particularly for oaks (Buckley 1994, Crow 1992, Emerson 1846, Gordon and others 1995, Johnson 1979, McGee 1975, Ward 1964). However, there is little documentation of the frequency and severity of frost damage to oaks at varying scales of resolution, e.g., within and among individuals, stands, and landscapes, although Geiger (1965) described general spatial and temporal variation of frost in forests. Early growing season frosts result from the interaction between cold air masses that generally affect a region (advective frost), radiative heat loss, and pooling of cold air in areas with restricted air flow. The duration and severity of sub-freezing temperatures are related to the local topography, amount and type of forest cover, and weather conditions. The inability of vegetation to return to some northern Wisconsin sites after forest clearing in the early 1900s was due, in part, to the presence of frost pockets. Plant response to frost can vary significantly depending on the inherent ability of species to withstand frost, the stage of plant development when frost occurs, the severity of frost, and the ability to recover after frost damage (Kramer and Kozlowski 1979).

This paper describes frost damage in 2 regenerating hardwood stands in northern Wisconsin. The purpose of these observations is to better understand the short term effects of frost and determine if different forest structures resulting from silvicultural treatments reduced damage.

METHODS

Willow Springs

The Willow Springs site is located on the Chequamegon National Forest near Park Falls, WI (T38N R3E, Sections 13-14). Our most detailed observations were made at this site, which has a habitat type of *Acer/Viola - Osmorhiza* (Kotar and others 1988). The topography is level to gently sloping (0 to 5 percent) and the soil is a moderately well-drained, sandy loam. Early results of an artificial oak regeneration study on this site are available (Teclaw and Isebrands 1991).

The study area consists of 4, 8 hectare overstory blocks—0 (clearcut), 50 percent, and 75 percent canopy cover and an uncut control. Basal area was 17.0, 20.7, and 32.4 square meters/hectare, respectively in the 50 percent, 75 percent, and control blocks. Temperature was measured in the clearcut, 50 percent, and control blocks at the surface and 0.25, 0.50, 1.0, and 2.0 meters above ground every 10 minutes. Hourly average and daily maximum and minimum values were recorded with Campbell CR10 data loggers. Weather stations were located in the center of each overstory treatment and temperature measurements were replicated three times within each block.

Shoot elongation was measured on sugar maple (*Acer saccharum* Marsh.), white ash (*Fraxinus americana* L.),

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ironwood (*Carpinus caroliniana* Walt.), red oak (*Quercus rubra* L), aspen—both trembling (*Populus tremuloides* Michx.) and big tooth (*Populus grandidentata* Michx.), and raspberry (*Rubus idaeus* L.) in the overstory blocks in which they were present. The terminal shoot was measured on each of six plants for each species. Plants were located near the center of the block, as close to the weather stations as possible, and at least 150 meters from the edge of the block. Moreover, red oak acorn germination was observed on six artificially seeded plots in each overstory treatment block. Shoot elongation and germination observations were made every 7 to 14 days from May 1 through early July and in early August and September.

Two weeks after the temperature in the clearcut was recorded at -9.2 °C on Julian days 146-147 (May 26-27), we established three transects in all overstory treatments to determine the spatial variation in frost damage. Sample points were located at 10 meter intervals beginning at the edge of the clearcut block and going into the adjacent 50 percent and untreated blocks. The transects in the 50 percent block spanned the block and ended in the adjacent uncut forest. Transects were 150 and 300 meters long in the control and 50 percent blocks, respectively.

White ash and red oak were the only tree species severely affected by the frost. Because of the small number of red oak and the large number of uniformly distributed white ash seedlings that regenerated naturally, sample trees were only white ash. When oak seedlings were encountered, they were evaluated and the degree of damage was compared to adjacent white ash. Based on these observations, red oak and white ash were similarly affected by the frost. At each sample point, up to 20 white ash seedlings ranging in height from 0.25 to 1.5 meters were evaluated. Stump sprouts were also sampled in the clearcut because seedlings in that block were rare. In the control, most seedlings were 0.25 meter tall or less due to poor performance of seedlings in full shade. Frost damage was classified according to the following scale: class 1—no damage; 2—up to 20 percent of leaves damaged; 3—21 to 40 percent damage; 4—41 to 60 percent damage; 5—61 to 80 percent damage; 6—81 to 99 percent damage; 7—all leaves killed. In classes 2 to 5, there was little obvious damage to the current year's shoot. In class 6, the current year shoot was frost-pruned to varying lengths; in class 7, the entire current year shoot and all leaves were killed.

Bird Lake

The Bird Lake site is located near Lake Tomahawk, WI, on the American Legion State Forest (T38N R7E, Section 18) and has a habitat type of *Acer/Vaccinium - Viburnum* (Kotar and others 1988). The topography is typical of ground moraine deposits in the region with irregular slopes varying from 0 to as much as 60 percent over short distances (fig. 1). The mesoscale topography at this site was much more complex than at Willow Springs. The Bird Lake site consists of three replications of three overstory treatments—clearcut, and 25 percent and 50 percent crown cover (fig. 1). Residual basal area in the 25 percent and 50 percent plots was 7 and 14 square meters/hectare, respectively. Each overstory plot is 0.56 hectare. Teclaw and Isebrands (1993) have reported the early results of red oak artificial regeneration studies on this site.

Three weeks after the frost on Julian days 146-47, we made frost damage observations in all nine overstory treatment plots. Within each plot, 12 sample points were established. At each sample point, up to five red oak seedlings and three red oak sprout clumps were evaluated using the damage classes described above. In addition, we measured total plant height and height of frost damage.

The contour map of the site was developed using a hand-held abney level. All elevations in figure 1 are referenced from the starting point of the survey, which was designated as the 100 isoline. The total relief for the area is approximately 28 meters (relative height of 88 to 116).

RESULTS

Willow Springs

In 1994, sub-freezing temperatures were recorded on Julian days 131-133 (May 11-13), 136 (May 16), 146-147 (May 26-27), 152-154 (June 1-3), and 159 (June 8). The growing degree days (dd) for the clearcut based on 5 °C for the frost dates above were 97 to 111, 128, 243, 310 to 329, and 380, respectively. Degree days in the 50 percent and control blocks were slightly less. Bud break occurred between May 7 and 11, and there was measurable shoot elongation on May 22. Neither red oak nor white ash showed signs of frost damage on May 21; thus, frosts before this did not appear to affect development.

The sub-freezing temperatures on days 146-147 caused an immediate wilted appearance and green tissues turned black within a few days. Temperature regimes differed considerably among the three overstory treatments. Minimum temperatures in the clearcut, 50 percent crown cover, and 100 percent crown cover blocks were -9.2 °C, -2.5 °C, and -0.6 °C, respectively. The duration of the freezing conditions was 10 hours in the clearcut, 8 hours in the 50 percent crown cover block, and only 1 hour in the 100 percent crown cover block. The depth of freezing temperatures in the inversion layer was up to 10 meters above the ground in the clearcut and 50 percent crown cover blocks and only up to 0.5 meter in the 100 percent crown cover block (fig. 2).

Species differed in their ability to withstand frost. In the clearcut, red oak and white ash were equally susceptible. The current year shoot and associated leaves were killed on every seedling and sprout clump throughout the 8 hectare block (figs. 3A, 3B, 4A, and 5A). Frost damage did not differ between plants in open areas and those present in the understory of stands of dense young aspen suckers. About 20 percent of the red oak acorns on artificially seeded plots in the clearcut had germinated at the time of the frost. They were subjected to the coldest temperatures, and their above ground stems were killed. Red oak germinants on the artificially seeded plots in the 50 percent and 75 percent blocks were not damaged. Of the other common tree and shrub species present, ironwood and

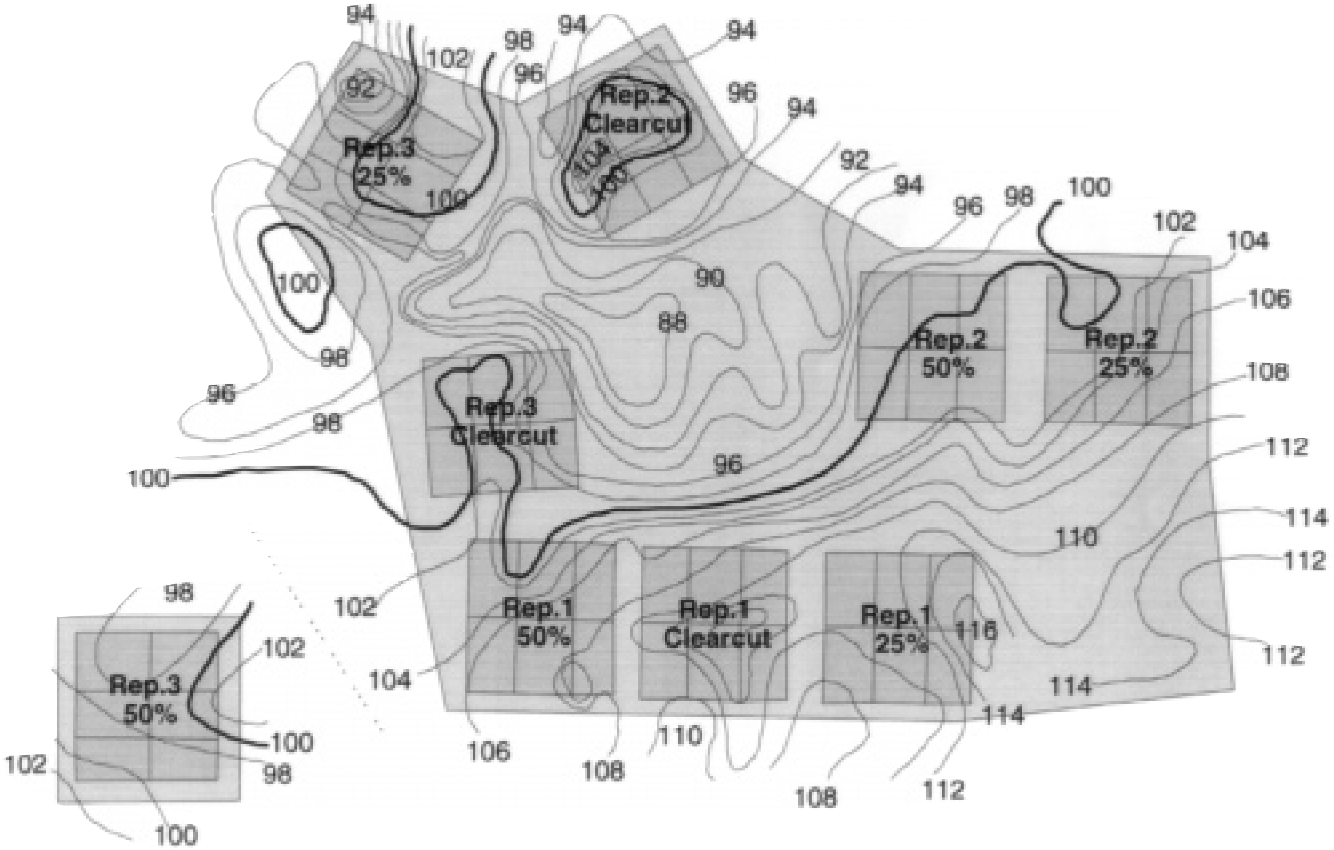

Figure 1—Contour map of the Bird Lake site with locations of clearcut, 25 percent crown cover and 50 percent crown cover plots. The elevations shown are in meters and relative to the 100 line (e.g., 88 indicates 12 meters lower than 100 and 116 indicates 16 meters higher). The total relief in the area is approximately 28 meters. Isolines in the Rep 3 - 50 percent plot are not continuous with other isolines in the figure as the plot is located several hundred meters away. See table 3 for estimates of frost damage in each treatment unit.

raspberry had some leaf and shoot damage. The damage to ironwood was not as immediately noticeable as in red oak and white ash. The most susceptible parts of the current growth of ironwood were killed by the frost, and there was a measurable reduction in shoot length (data not shown). In contrast, sugar maple and aspen did not exhibit signs of frost damage. Shoot growth patterns observed here were similar to those reported by Buech (1976) at another site in northern Wisconsin.

Severity of frost damage to white ash and red oak varied in the 50 percent and control blocks depending, in part, on the distance to the edge of the clearcut block. In the control, class 7 damage occurred to a minor degree up to 20 meters from the edge of the clearcut. From 30 to 150 meters from the edge, no damage was observed (table 1). In the 50 percent block, class 6 and 7 damage was common near the edge of the clearcut, but gradually declined with distance from the edge of the clearcut. Beyond 80 meters, class 7 damage did not occur and undamaged seedlings were most common, although class 2 damage occurred up to 210 meters from the clearcut (table 2). The damage from 80 to 190 meters from the edge of the clearcut block may be representative of that

Table 1—Percentage of frost damaged seedlings in the control block as related to distance from the edge of the clearcut

	Frost damage class						
Distance from edge	1	2	3	4	5	6	7
m							
0-10	24	-	6	-	9	1	60
11-20	88	2	-	7	-	-	3
21-30	96	-	2	-	2	-	-
31-150	100	-	-	-	-	-	-

Class 1 = no damage, 2 = up to 20 percent of leaves damaged, 3 = 21 to 40 percent damage, 4 = 41 to 60 percent damage, 5 = 61 to 80 percent damage, 6 = 81 to 99 percent damage, 7 = all leaves killed.

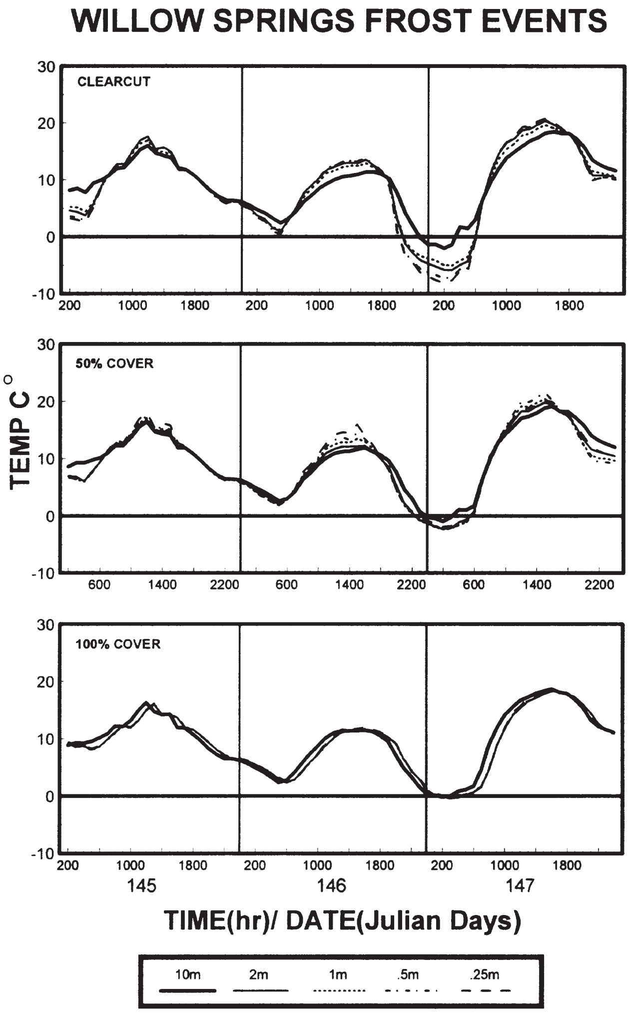

Figure 2—Temperature profiles in clearcut, 50 percent crown cover, and control blocks at the Willow Springs site on days 146 to 147.

Table 2—Percentage of frost damaged seedlings in the control block as related to distance from the edge of the clearcut

Distance from edge	Frost damage class						
	1	2	3	4	5	6	7
m							
0-10	-	-	-	-	-	-	100
11-20	-	-	-	-	-	-	100
21-30	-	-	-	-	10	-	90
31-40	-	-	-	-	8	-	92
41-50	2	-	6	2	6	10	74
51-60	25	14	7	4	10	14	25
61-70	29	19	12	9	12	12	7
71-80	61	19	5	5	4	5	2
81-90	74	18	4	4	-	-	-
91-100	93	6	-	1	-	-	-
101-110	78	19	-	3	-	-	-
111-120	93	6	-	1	-	-	-
121-130	82	18	-	-	-	-	-
131-140	100	-	-	-	-	-	-
141-150	87	13	-	-	-	-	-
151-160	78	22	-	-	-	-	-
161-170	80	20	-	-	-	-	-
171-180	89	11	-	-	-	-	-
181-190	73	27	-	-	-	-	-
191-200	87	13	-	-	-	-	-
201-210	96	4	-	-	-	-	-
211-280	100	-	-	-	-	-	-

Class 1 = no damage, 2 = up to 20 percent of leaves damaged, 3 = 21 to 40 percent damage, 4 = 41 to 60 percent damage, 5 = 61 to 80 percent damage, 6 = 81 to 99 percent damage, 7 = all leaves killed.

expected with a frost of this intensity (fig. 2) in larger blocks of continuous forest with these overstory conditions. Beyond 190 meters from the edge of the clearcut, the transect approached the uncut area and this may have affected microclimate (table 2). None of the other tree species or raspberry exhibited any evidence of frost damage in any part of the 50 percent block. The 75 percent block was surrounded by untreated forest and no damage to red oak, white ash or other species occurred.

The type of flushing after frost damage varied with the severity of damage. All plants with class 7 damage produced new flushes. Most commonly, the new shoot originated from dormant buds in the terminal bud cluster on the previous year's growth (fig. 3B) or from a dormant axillary bud on the previous year's growth or an axillary bud near the base of the seedling. Flushing in seedlings and stump sprouts was similar. Flushing began 1 to 2 weeks after the frost and was completed within about 2 weeks of budbreak. The post-frost flush was longer than the original flush in some trees (figs. 4 and 5).

Undamaged buds on the current year's shoot flushed (fig. 3C and 3D) on seedlings that sustained partial frost damage to leaves or shoots (e.g., red oak in the 50 percent block and ironwood in the clearcut block). Flushing occurred on stems with only damaged leaves and on those with damaged leaves and shoots pruned to varying lengths by the frost. For example, most of the leaves on the terminal shoot of tree 3 (fig. 4B) were killed and subsequently shed, but the shoot was not damaged. Following this damage, several axillary buds flushed, resulting in additional net shoot elongation for the year. Also, the tip of the shoot of tree 4 (fig. 4B) was killed and shoot length was reduced in the short-term; however, flushing of a bud on this shoot recovered the lost length and added an additional 10 centimeters.

Bird Lake

Red oak was the only species damaged by frost at Bird Lake. Other species present, but not damaged, were paper birch (*Betula papyrifera* Marsh.), red maple (*Acer rubrum*),

Figure 3—Examples of frost damage to northern red oak where (A) entire shoot and all leaves were killed, (B) close-up of frost killed shoot showing: (1) frost killed original flush and (2) new shoot formed by flush of dormant bud, (C) undamaged shoot with nodes where frost-damaged leaves have been shed and axillary buds flushing, (D) shoot with frost pruned tip and various degrees of leaf damage.

raspberry, and blackberry (*Rubus allegheniensis* Porter). Frost damage was not closely associated with any of the three overstory conditions in this area as it was at Willow Springs, but appeared associated with topographic position. However, seedlings and sprouts in 50 percent plots had the least damage (class 2-3), followed by the 25 percent plots and clearcuts, respectively. Although the number and size of plots in our study were not large enough to establish a definitive relationship between overstory condition and topographic position, frost damage and topographic position seem related. For example, in the clearcut plots, the least damage occurred in the plot with the highest relative elevation (clearcut plot 2, fig. 1, table 3). Seedlings and sprouts in the 25 percent and 50 percent plots tended to have more class 6 and 7 damage where the elevation was below 100 than in those with elevations greater than 100 (fig. 1, table 3).

The height on the plant to which frost damage occurred differed among the plots. Affected seedlings always exhibited damage over the entire plant, suggesting that they were completely within the portion of the inversion layer with temperatures low enough to freeze stem and leaf tissues. However, stump sprouts in 25 percent plot 2 and 50 percent plot 2 were of sufficient height so that their tops were above the freezing conditions of the inversion layer, while their lower branches were damaged by freezing conditions. Red oak trees along roads with the lower part of their crowns damaged and the upper part undamaged were common in this area. The oak stump sprouts in clearcut plot 2 and 25 percent plot 3 showed damage over their entire vertical structure.

We did not follow the post-frost flushing on this site. However, measurements made at the end of the growing season indicated that all seedlings had produced new shoots and no mortality was observed as a result of the frost damage.

DISCUSSION

In northern Wisconsin, frosts often occur during the early growing season when trees are most susceptible to freezing temperatures. Frost damage should be viewed at different scales of resolution and as variable in space and time. At Willow Springs and Bird Lake, frost damage differed among shoots on the same plant, among species and populations of the same species in an area, and among stands within a landscape. We also observed variation in frost damage related to a significant depth-of-edge (Chen and others 1995) between areas differing in temperature regimes as, for example, between the clearcut block and the adjacent 50 percent and control blocks at our Willow Springs site. The observed penetration of frost damage into the control and 50 percent blocks from the clearcut at Willow Springs may have important implications for silvicultural practices such as strip or group selection cuts. The patterns of frost damage that we observed substantiate the work of Geiger (1965) on the spatial variation in frost damage in forests, and they provide examples of how hardwood forests of the northern Great Lakes region are affected by frost.

Understanding the role of frost in determining the composition and structure of these forests requires long term observations. Several things do stand out, however, from short-term observations. First, there are great differences in susceptibility to frost among species, and the most susceptible species, e.g., oak and ash, are at a disadvantage on sites where growing season frosts are common. For example, at Willow Springs, red oak was affected by frost in each of the past 6 years in the clearcut. Stump sprouts appear to have an advantage over seedlings in that they grow faster and attain heights that are above the damaging inversion layer during frost events and may be the best way to maintain oak as a component of the forest where frost is a concern. Oak sprouts in lower Michigan recovered quickly from frost damage and there was little effect on net shoot elongation (Johnson 1979).

Frost is probably not a critical factor by itself in terms of seedling survival. Rather, it is one of a host of biotic and abiotic factors affecting survival and growth. Buckley (1994, 1998) found that frost damaged seedlings often died, but that a combination of frost and repeated browsing of the new shoots (produced after the frost) likely caused most of

Table 3—Percentage of frost damaged seedlings at the Bird Lake site relative to canopy cover and elevation

Frost damage class	Clearcut Plot 1	Clearcut Plot 2	Clearcut Plot 3	25 percent crown cover Plot 1	25 percent crown cover Plot 2	25 percent crown cover Plot 3	50 percent crown cover Plot 1	50 percent crown cover Plot 2	50 percent crown cover Plot 3
Elevation (m)	108-114	94-104	96-100	110-116	100-108	92-102	102-110	96-102	98-102
1	25	-	2	31	28	-	25	14	45
2	34	-	19	27	23	-	41	12	10
3	29	-	17	22	11	-	26	8	3
4	4	-	17	11	3	-	2	11	-
5	5	-	8	4	2	-	-	14	5
6	2	-	21	-	20	-	-	15	8
7	2	100	17	5	12	100	7	26	28

Class 1 = no damage, 2 = up to 20 percent of leaves damaged, 3 = 21 to 40 percent damage, 4 = 41 to 60 percent damage, 5 = 61 to 80 percent damage, 7 = all leaves killed.

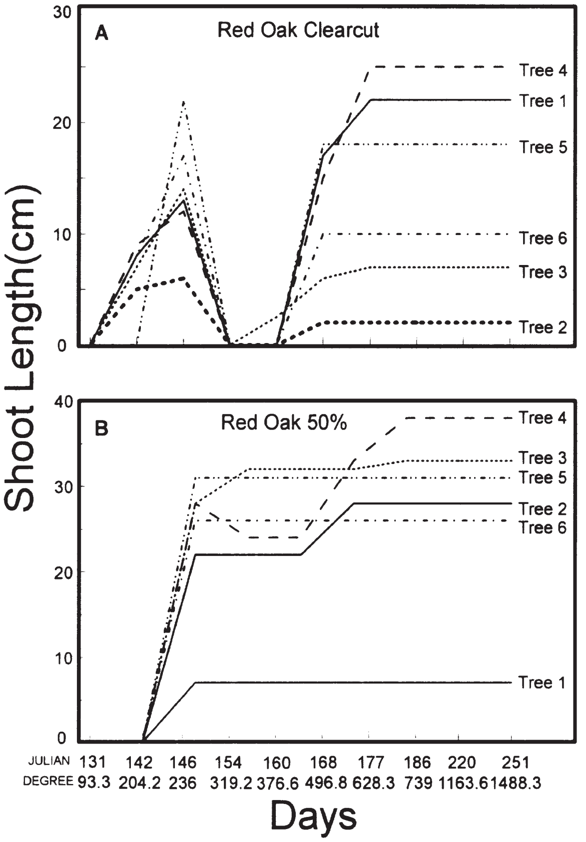

Figure 4—Shoot elongation of red oak in (A) clearcut, and (B) 50 percent crown cover block at the Willow Springs site. The frost on Julian days 146 and 147 killed all new shoot growth in the clearcut (fig. 3A), but affected only less mature tips of shoots in 50 percent crown cover (fig. 3C and 3D). Shoot growth of individual trees is shown for clearcut and 50 percent crown cover areas to better illustrate some of the variation within the seedling population. Note the effect of frost pruning on shoot length of tree 4 and on shoot elongation resulting from flushing in trees 2 and 4 in 50 percent crown cover.

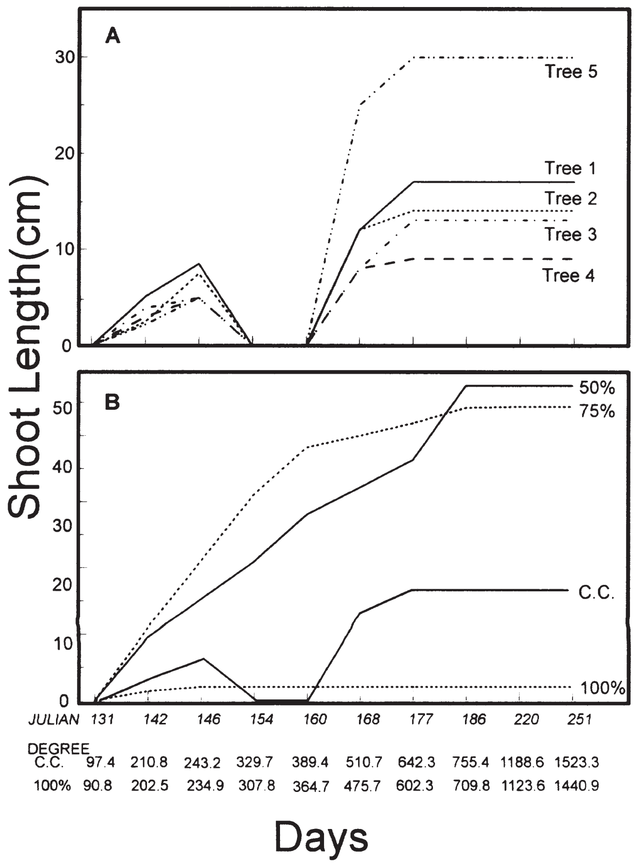

Figure 5—Shoot elongation of white ash (A) 5 individual trees in the clearcut, and (B) average of 5 trees in clearcut, 50 percent and 75 percent crown cover, and control blocks at Willow Springs.

the mortality. Where portions of terminal leaders are severely frost-pruned, reductions in height and photosynthate production may decrease growth rate and increase the period when a large proportion of the photosynthetic tissues of a seedling are susceptible to additional frosts, herbivory, and competition with understory vegetation.

Our work confirms the studies of Ringger (1972) on microclimate of forest gaps in northern hardwoods, as well as the broader experience of Geiger (1965) who found that silvicultural treatments significantly influence microclimate and more specifically the distribution, severity and effect of frost in the landscape. The importance of creating or maintaining an overstory to reduce frost damage to oak regeneration was recognized 150 years ago by Emerson (1846), Crow (1992), Buckley (1994, 1998), and Gordon and others (1995). The appropriate level of overstory cover and composition will depend on the local topography of the area and the frequency and severity of frosts in an area. For areas with relatively little relief, such as our Willow Springs site, it appears that a 50 percent canopy cover is at the lower end of a threshold for total prevention of frost. This agrees with observations from lower Michigan (Buckley 1994, 1998) of a significant decrease in frost damage as crown cover increased from 25 to 75 percent. However, on sites with more irregular topography, such as Bird Lake, there is a relationship between overstory condition, topographic position, and potential for frost. Small gaps or clearcuts in areas with highly irregular topography might best be situated on relatively higher areas where cold air drainage does not result in frost pockets. Because of the depth-of-edge between areas with substantially different microclimates, there could be considerable damage to susceptible species in stands that are not large enough to have a true interior (Chen and others 1995).

In summary, these two case studies provide insight into some aspects of how frost affects the development of hardwood forests and how silvicultural treatments can be used to ameliorate frost in northern hardwood forests. In areas subject to frequent late spring frosts, managers would benefit from increased information about the relationship of frost damage to canopy cover, canopy composition, and the size of canopy openings. Certainly more work is needed to characterize interactions between these factors and frost, and to understand the long-term impacts of frost on species in these forests.

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REFERENCES

Buckley, D.S. 1994. Relationships among competing vegetation, herbivores, and environmental conditions affecting northern red oak (*Quercus rubra* L.) regeneration. Houghton, MI: Michigan Technological University. 581 p. Ph.D. dissertation.

Buckley, D.S.; Sharik, T.L.; Isebrands, J.G. 1998. Ecology. 79(1): 65-78.

Buech, R.R. 1976. Tree shoot elongation in northern Wisconsin and relationships with temperature and precipitation. Canadian Journal of Forest Research. 6: 487-498.

Chen, J.; Franklin, J.F.; Spies, T.A. 1992. Growing season microclimatic gradients from clearcut edges into old-growth Douglas-fir forests. Ecological Applications. 5: 74-96.

Crow, T.R. 1992. Population dynamics and growth patterns for a cohort of northern red oak (*Quercus rubra*) seedlings. Oecologia. 9: 192-200.

Emerson, G.B. 1846. Report on the trees and shrubs growing naturally in the forests of Massachusetts report to the legislature. Boston: Dutton and Wentworth, State Printers.

Geiger, R. 1965. The climate near the ground. Cambridge, MA: Harvard Press. 611 p.

Gordon, A.M.; Simpson, J.A.; Williams, P.A. 1995. Six-year response of red oak seedlings planted under a shelterwood in central Ontario. Canadian Journal of Forest Research. 25: 603-613.

Johnson, P.S. 1979. Shoot elongation of black oak and white oak sprouts. Canadian Journal of Forest Research. 9: 489-494.

Kotar, J.; Kovanch, J.A.; Locey, C.T. 1988. Field guide to forest habitat types of northern Wisconsin. Madison, WI: University of Wisconsin, Department of Forestry. 153 p.

Kramer, P.J.; Kozlowski, T.T. 1979. Physiology of woody plants. New York: Academic Press. 811 p.

McGee, C.E. 1975. Change in forest canopy affects phenology and development of northern red oak and scarlet oak seedlings. Forest Science. 21: 175-179.

Ringger, D.L. 1972. The microclimate of various forest openings. Milwaukee: University of Wisconsin. 122 p. M.S. thesis.

Teclaw, R.M.; Isebrands, J.G. 1991. Artificial regeneration of red oak in the northern Lake States. In: Laursen, S.B.; DeBoe, J.F., eds. The oak resource in the upper Midwest—implications for management. Publ. NR-BU-5663-5. St. Paul, MN: University of Minnesota: 187-197.

Teclaw, R.M.; Isebrands, J.G. 1993. An artificial regeneration system for establishing northern red oak on dry-mesic sites in the Lake States. Annuls of Science Forestry. 50: 543-552.

Ward, W.W. 1964. Bud distribution and branching in red oak. Botanical Gazette. 125: 217-220.

RED MAPLE DYNAMICS IN APPALACHIAN HARDWOOD STANDS IN WEST VIRGINIA

Brian D. Tift and Mary Ann Fajvan[1]

Red maple (*Acer rubrum* L.) is increasing in understory abundance in Appalachian Hardwood forests in the eastern United States. Partial cutting practices are typically used to harvest the mature, second generation forests. These partial overstory removals may increase the importance of red maple in the overstory of the next forest generation. The ability of understory red maple to respond to minor disturbances may give it a competitive advantage over new regeneration. The growth strategies red maple uses to attain overstory canopy positions were investigated on mesic and dry sites, in two West Virginia Appalachian hardwood stands. In both stands, red maple comprised a small percentage of overstory basal area but was the most abundant understory species. Stem analysis and increment cores were used to determine disturbance history and to examine height and diameter growth relationships among species. The stands originated after a stand-replacing disturbance around 1925. During stand initiation, red maple invaded the sites for 10-15 years longer than most other species, however, all of the codominant red maples sampled were similar in age to other codominant species. Even though stump sprouting is a typical regeneration mechanism for red maple and other species in the region, none of the sampled stems showed evidence of sprout origin. Red maple may have been a better competitor on the dry site because even though red maple grew faster on the mesic site, the drier site had a higher percentage of codominant red maples. Regardless of site, codominant red maples had height growth rates similar to codominant oaks (*Quercus* spp.) throughout stand development. The height growth rates of codominant yellow-poplars (*Liriodendron tulipifera* L.) were usually higher than other species, especially during the first 20 years of development. Only one sampled codominant maple achieved codominance after being overtopped for 20 years (Figure 1a). Stem analyzed understory maples showed fluctuating height growth rates in response to small disturbances in the overstory. Age did not appear to decrease red maple's ability to respond to canopy disturbance. Three understory maples showed significant height growth increases when they were 40-50 years old (Figure 1c,e). Red maple's shade tolerance, understory abundance, and ability to respond to minor canopy disturbances, suggests that partial harvesting increases the likelihood that future stands could have a higher overstory red maple component.

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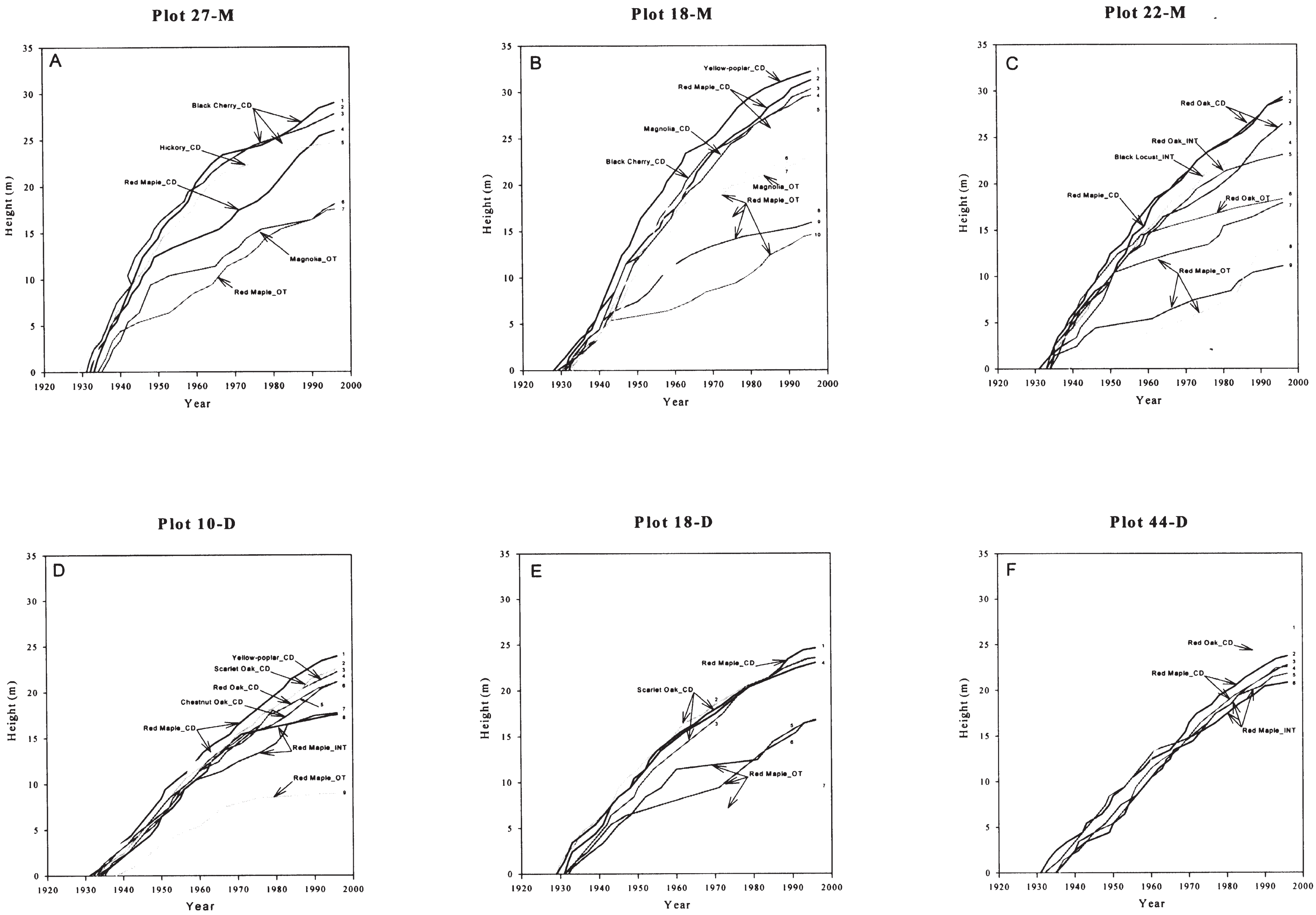

Figure 1— Height growth trajectories for stem analysis plots. Crown class designations CD-codominant, INT-intermediate, OT-overtopped.

Nutrient Dynamics

EFFECTS OF HARVESTING ON SOIL NITROGEN (N) DYNAMICS IN A N-SATURATED HARDWOOD FOREST

Frank S. Gilliam and Mary Beth Adams[1]

Abstract—Recent evidence suggests that soils of some central Appalachian hardwood forests have become nitrogen (N) saturated, a condition that develops when availability of soil N exceeds demand for N by plant roots and soil microbes. Among many environmental concerns associated with N saturation are the following: (1) greatly altered N cycle seen as a de-coupling of N dynamics from the high degree of biotic control that occurs in N-limited forest ecosystems and (2) greatly enhanced nitrification and leaching of NO_3^- along with calcium (Ca^{++}) and magnesium (Mg^{++}) from the soil and into streams, depleting base cation availability in the soil. Both of these have been demonstrated for hardwood forests of the Fernow Experimental Forest (FEF), West Virginia. Because increases in rates of nitrification and loss of soil cations often are associated with forest harvesting, it is important to determine whether the simultaneous effects of N saturation and harvesting may accelerate ecosystem degradation. Accordingly, we established a plot-based study to examine the effects of harvesting with and without addition of N and cations on soil N dynamics at FEF. Our treatments (each with four replicate 0.2-ha plots) were as follows: (1) Control (no treatment), (2) Harvest only (whole-tree harvesting [WTH] of entire plot), (3) Harvest+N (WTH plus 36 kg N/ha/yr), and (4) Harvest+N+cations (WTH plus 36 kg N/ha/yr plus 22.4 and 11.9 kg/ha/yr each of Ca and Mg, respectively). The cation addition treatment was done to simulate a mitigative treatment to replace loss of base cations to leaching. As expected at this N-saturated site, even our control plots exhibited high rates of N mineralization, mostly as net nitrification. However, these high rates were increased significantly by all harvest treatments. The addition of N in combination with harvesting did not increase nitrification significantly. By contrast, the addition of cations in combination with harvesting and N resulted in nitrification rates significantly higher than the other harvest treatments. These results illustrate the increasing complexities and challenges facing management of hardwood forests in the context of pollutant conditions.

INTRODUCTION

Deciduous forests of the eastern United States have exhibited an extremely high degree of resilience to a variety of historical perturbations and stresses. For example, although most of these forests originated after the heavy cutting and fires that occurred during the era of railroad logging (1880-1930), present forests have generally high species richness and productivity (Marquis and Johnson 1989, Gilliam and others 1995). While overall forest productivity remains reasonably high despite a growing demand for wood, the question of sustainable productivity, biodiversity, and health of forest ecosystems of the eastern U.S. is one of increasing concern. Indeed, recent data indicate that several eastern deciduous forest species are experiencing higher-than-normal rates of mortality (Twardus 1995). Despite on-going debate among forest ecologists, foresters, and resource managers, the observed mortality has been attributed to a variety of factors, alone or in combination. Some of these are acidic deposition, nutrient cation depletion, and excess nitrogen deposition, all of which have the potential to decrease survivorship of several forest species by weakening the trees and increasing their vulnerability to other stresses, including temperature and moisture extremes, disease, and insects (Adams and Eagar 1992, Eagar and Adams 1992).

One of the primary hypothesized agents of observed declines is base cation depletion of poorly buffered soils via forest harvesting and atmospheric inputs of nitrogen and sulfur (Eagar and Adams 1992, Gilliam and Richter 1991). Although typical central Appalachian soils are not considered infertile (Auchmoody 1972), recent concerns about sustainability of central Appalachian forests have been linked with soil fertility. Declines in nutrient cation levels in soil, particularly calcium (Ca) and magnesium (Mg), from intensive harvesting of forest products have been documented in some parts of the United States (Federer and others 1989, Mann and others 1988). Because timber harvesting is expected to increase in this region, and because the shift in forest utilization is toward fiber and more intensive harvesting, cation export in forest biomass could amount to a significant loss from some forest sites.

Earlier research on the effects of acidic deposition on forest systems indicated that the productivity of many soils in the humid east could decline significantly within 50-70 years (Binkley and others 1989), the span of a typical timber rotation for many stand types (Marquis and Johnson 1989). The central Appalachian region receives some of the highest inputs of acidic deposition in the United States (Adams and others 1993, Gilliam and Adams, 1996). The Clean Air Act contained provisions primarily targeting sulfur (S) emissions, not N emissions. Indeed, U.S. emissions of sulfur in 1990 were >25 percent less than those in 1970, while emissions of N were 2

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percent higher in 1990 compared to 1970 (World Resources Institute 1994). In short, pollutant emissions of oxides of N are not expected to decline greatly in the near future. Thus, considering the likelihood of several eastern deciduous forests becoming N saturated in the future and the increased demand on these forests for the production of wood products, information on interactions between N saturation and forest harvesting will be of great importance for forest ecologists, forest and resource managers, and policy makers. Research suggests strongly that some sites have reached or are nearing N saturation (Stoddard 1994, Gilliam and others 1996, Fenn and others 1998, Aber and others 1998). Furthermore, studies of stream chemistry at other eastern forest sites have shown that elevated N additions resulted in elevated losses of N (as NO_3) and base cations, especially calcium (Ca) and magnesium (Mg), suggesting a direct connection between N saturation (N additions) and the leaching of nutrient cations (Kahl and others 1993, Rustad and others 1993, Norton and others 1994).

The effects of forest harvesting on ecosystem nutrient cycling have been studied for many forest types and harvesting practices. Indeed, much has been learned to indicate that response of the biota (recovering vegetation) is largely responsible for regulating nutrient change as the forest recovers from the disturbance of the harvest regime (Binkley 1986, Tritton and others 1987, Martin 1995, Gilliam and Adams 1995, Johnson 1995). It also has been learned, however, that there is great inter-site variability in these responses, precluding broad generalizations across forest ecosystems. Despite their great ecological and economic importance, forests of the central Appalachian region have not been studied extensively with respect to nutrient responses to forest harvesting.

We know of no studies that have examined N saturation in the context of commercial forest harvesting. Accordingly, the purpose of this study was to examine the effects of harvesting on N mineralization and nitrification in a N-saturated forest. Harvesting was carried out alone and in combination with additional treatments to simulate different pollutant and mitigation scenarios. In addition, because mineralization and nitrification rates are often found to be directly related to soil temperature, the effect of harvesting on the temperature of surface soils was examined in a sub-set of sample plots. Finally, we will review current knowledge on N dynamics at the Fernow Experimental Forest as an example of a central Appalachian forest under conditions of N saturation.

METHODS

Study Site

This study was done on the Fernow Experimental Forest (FEF), Tucker County, West Virginia, a 1900-ha area of montane hardwood forests within the unglaciated Allegheny Plateau (39°03'N, 79°49'W). Prior to treatments, stands of all sample plots were typical of mature forests of the region (Gilliam and others 1995), dominated by sugar maple (*Acer saccharum* Marsh.) and northern red oak (*Quercus rubra* L.), with occasional large stems of black cherry (*Prunus serotina* Ehrh.) and yellow poplar (*Liriodendron tulipifera* L.). Soils of the study site within FEF have been described in great detail by Lusk (1998). In general, they are acidic (pH 4.1-4.7), shallow (30-40 cm to C horizon), and loamy in texture (Lusk 1998), similar to characteristics of soils of other watersheds of FEF (Gilliam and others 1994).

Field Design and Sampling

A single 4-ha mixed hardwood forest site was located on uniform soil at FEF. Slopes were gentle with a southeast aspect. Sixteen 0.2-ha plots, separated by 15 m buffer areas, were located and treatment assignments made prior to harvest. Twelve of the 16 plots were designated as harvest plots; four plots remained unharvested as controls. The treatments (each with four replicate plots) were as follows: (1) Control (C, no treatment), (2) Harvest only (H, whole-tree harvesting of entire plot), (3) Harvest+N (HN, harvesting as in 2 plus 36 kg N/ha/yr), and (4) Harvest+N+cations (HNC, as in 3 plus 22.4 and 11.9 kg/ha/yr each of Ca and Mg, respectively). The base cation treatment was carried out by adding dolomitic limestone by hand; rates were chosen to be twice the annual loss in stream flow from the reference watershed (WS4) at FEF (Adams and others 1997). All sampling and analyses were carried out during a one-year pre-treatment period and, for this paper, a one-year post-treatment period.

Available (extractable) N pools and rates of N mineralization and nitrification were measured with in situ incubations (buried bag technique of Eno 1960), based on the method described by Gilliam and others (1996). Following removal of forest floor material at a single sample point within each subplot, mineral soil was sampled by hand trowel to a 5-cm depth. Each sample was divided into two sub-samples and placed in polyethylene bags, one bag buried in the mineral soil at 5 cm for ~28 d and the other brought back to the lab. To minimize disturbance to the soil, samples were not sieved, but pebbles, rocks, and larger roots were excluded from sub-samples. All bags were refrigerated immediately in the field (using icepacks within backpacks) and then stored at 4 C prior to extraction. Sampling was carried out from approximately the middle of May to the middle of September for each of the 2 years.

Soil temperature was measured with HoboTemp (Onset Corporation) thermister temperature probes. Temperature was monitored continuously from 1 May to 30 November by placing probes at a 10-cm depth in four locations in each of two control and two harvest plots.

Laboratory and Data Analyses

Sub-samples of soil from paired sample bags were extracted for determination of net N mineralization and nitrification. Moist soils were extracted with 1N KCl at an extract:soil ratio of 10:1 (v:w). Extracts were analyzed colorimetrically for NH_4 and NO_3 with a Bran+Luebbe TrAAcs 2000 automatic analysis system. Net N mineralization was calculated as post-incubation (NH_4 and NO_3) minus pre-incubation (NH_4 and NO_3); net nitrification was calculated as post-incubation NO_3 minus pre-incubation NO_3.

To determine the relative amount of N mineralized that was eventually nitrified, N mineralization rates were compared to net nitrification rates using Pearson product-moment correlation and linear regression. Monthly rates of N mineralization and nitrification were averaged across all sample times in each year for each plot, four plots per treatment (i.e., N=4 for all means). Pre-treatment means were compared among treatments using ANOVA, whereas pre- versus post-treatment means were compared using t-tests for each treatment separately (Zar 1994).

RESULTS

Response of Soil Temperature to Harvesting

Seasonal patterns of soil temperature were similar among all four plots in which temperature was measured during the pre-treatment (pre-harvest) period (data not shown). Harvesting greatly increased maximum daily temperature of surface soil, especially from early June to mid October (fig. 1). Seasonal patterns of maximum soil temperature were highly similar between the two plots within each treatment type (harvest or control).

Response of Soil N to Harvesting and Additional Treatments

There was a highly significant correlation ($P<0.01$) between N mineralization and nitrification during the pre-treatment period ($r = 0.93$) (fig. 2). This was also highly significant for the post-treatment period ($r = 0.95$) (fig. 3).

Analysis of variance revealed no significant differences ($P<0.05$) among treatments during the pre-treatment period for net N mineralization (fig. 4). T-tests showed no differences in net N mineralization in pre- versus post-treatment periods for control (C) plots. However, net N mineralization was significantly higher ($P<0.05$) in post-treatment than pre-treatment plots for all harvest treatments, particularly for the HNC treatment.

The pattern for net nitrification was similar to that for mineralization. There were no significant differences ($P<0.05$) among treatments during the pre-treatment period, but there were significant differences between pre- and post-treatment means for most of the harvest treatments (fig. 5). Post-treatment nitrification rates were slightly (by about 25 percent), but significantly ($P<0.10$) higher in C plots. Net nitrification increased 2- to 2.5-fold from pre- to post-treatment periods on all harvest plots. This was significant ($P<0.10$) for all harvest treatments, except HN plots (fig. 5).

DISCUSSION

Although not always measured, soil temperature is assumed to increase following whole-tree harvesting. Because activity of nitrifiers is usually highly temperature-dependent,

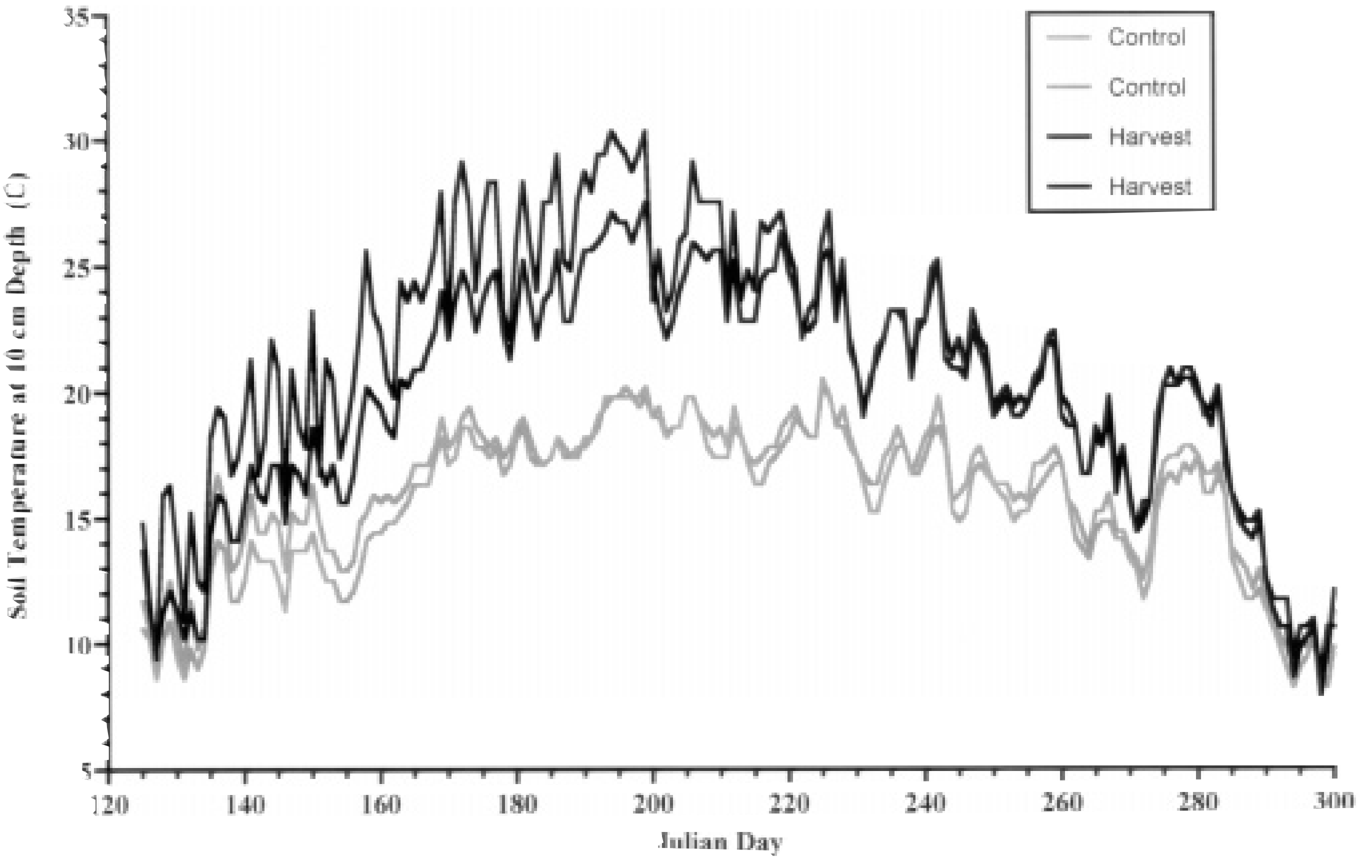

Figure 1—Soil temperature in two control and two harvest plots at Fernow Experimental Forest, WV. Data are daily maximum temperatures monitored continuously at a 10-cm depth.

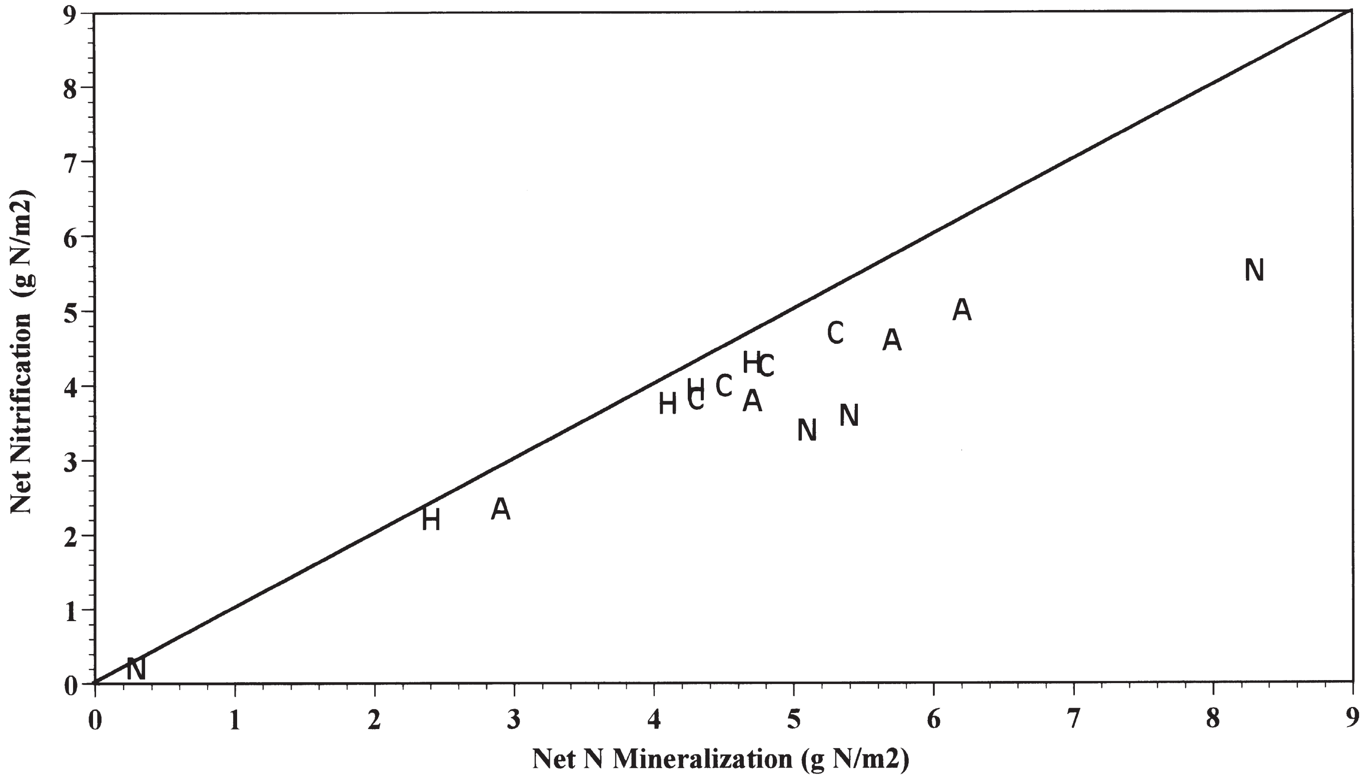

Figure 2—Nitrification versus N mineralization for individual plots during the pre-treatment period. Data are mean values for each of four plots for four treatments as follows: C = control; H = harvest; N = harvest + N; A = harvest + N + cations (i.e., A = all). Correlation was significant at P<0.01, r = 0.93. Line given is a 1:1 line representing a condition wherein N mineralization is 100 percent nitrification.

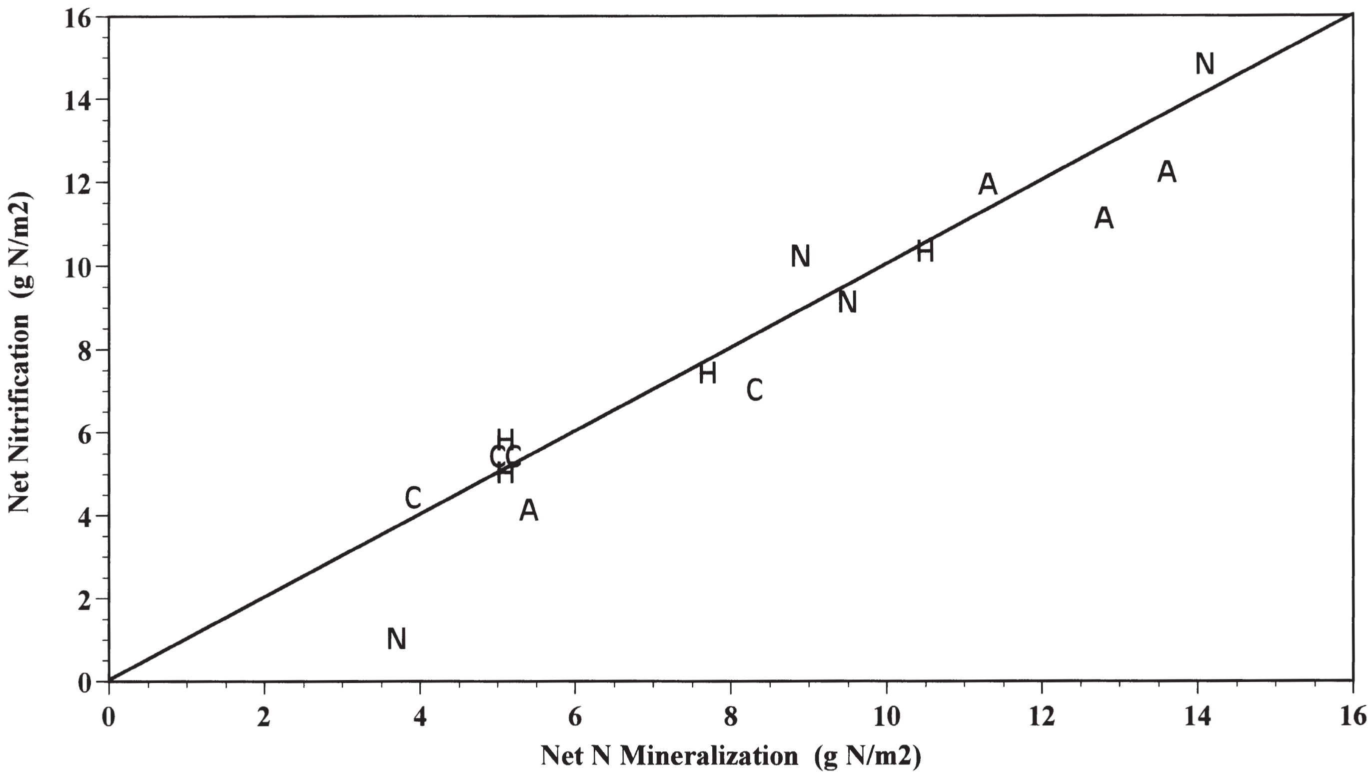

Figure 3—Nitrification versus N mineralization for individual plots during the post-treatment period. Data are mean values for each of four plots for four treatments; see fig. 2 for meaning of symbols. Correlation was significant at P<0.01, r = 0.95. Line given is a 1:1 line representing a condition wherein N mineralization is 100 percent nitrification.

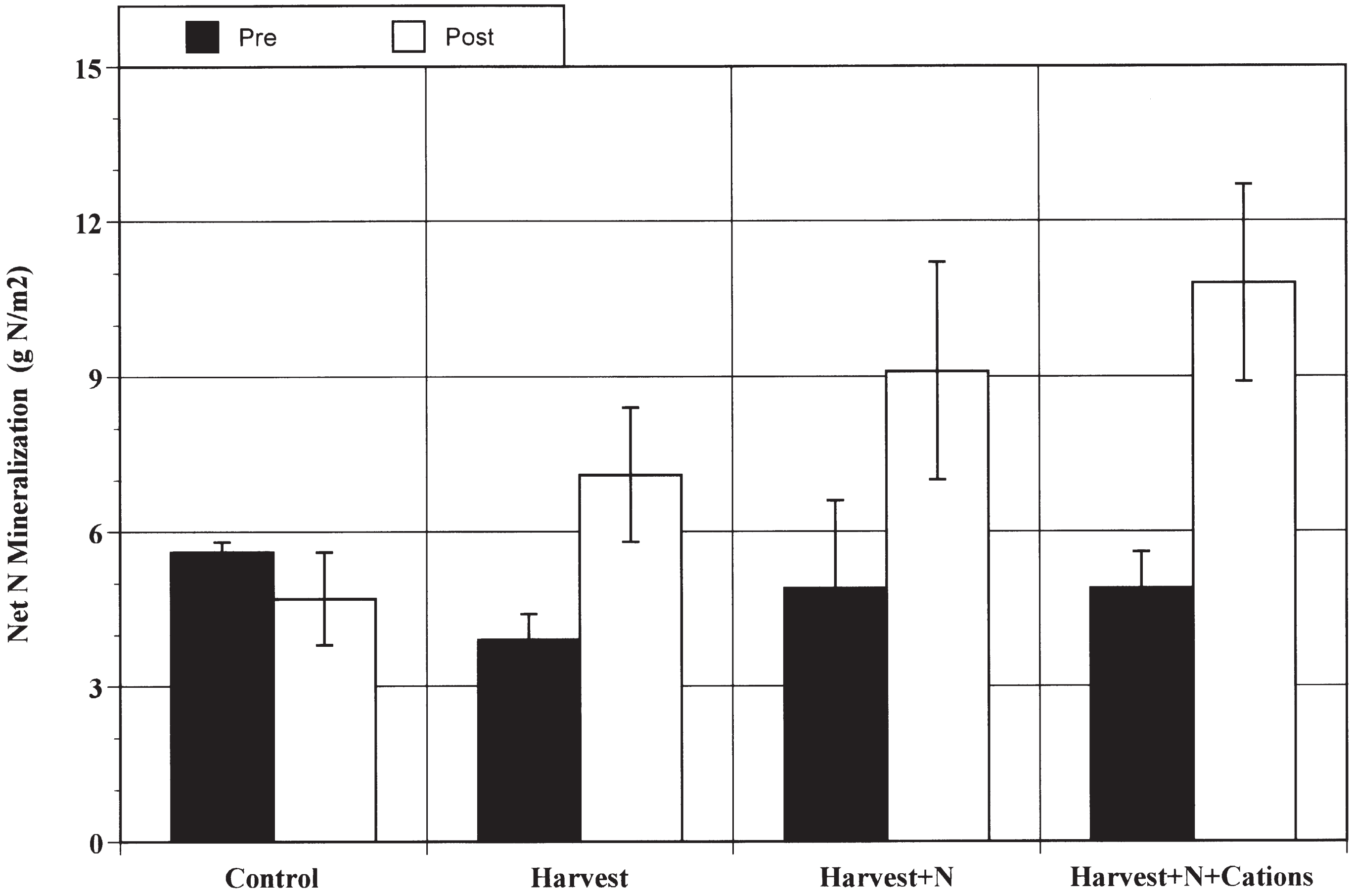

Figure 4—Net N mineralization for pre- and post-treatment periods for control and harvest treatment plots at Fernow Experimental Forest, WV. Data are means plus and minus one standard error of the mean. Analysis of variance revealed no significant differences among pre-treatment means. Pre- versus post-treatment differences were significant ($P<0.05$) for all harvest plots, but not for control plots.

however, it was important in this study to quantify harvest-mediated changes in soil temperature. Our data confirm that post-harvest increases in soil temperature can be prolonged and substantial (fig. 1). These increases likely arose from changes in the relationship between soil heat flux (S) and net radiation (Rn)(Rosenberg and others 1983). Idso and others (1975) showed that S (increases of which lead to increases in soil temperature) is highly positively correlated to Rn under a variety of soil conditions. The removal of the tree canopy during harvesting decreases interception of solar radiation and increases Rn, which increases S and results in the increased soil temperatures seen in fig. 1. These patterns have important implications for N dynamics and their responses to harvesting.

Not only was net nitrification highly correlated to net N mineralization, nitrification was also an extremely high proportion of N mineralization in both the pre- and post-treatment periods, regardless of treatment (figs. 2 and 3). Linear regressions of each of the sample periods closely approximated the theoretical 1:1 line that indicates a condition wherein nitrification is 100 percent of mineralization. These high rates of nitrification (relative to mineralization) are indicative of N saturation (Aber 1992), supporting findings and conclusions of Gilliam and others (1996) for three watersheds of FEF. This was especially the case during the post-treatment period for the harvest plots, suggesting that harvesting may enhance the relative predominance of nitrification in the N dynamics of N-saturated soils.

Patterns of soil temperature (fig. 1) combined with the relationships shown in figs. 2 and 3 are helpful in interpreting pre- versus post-treatment comparisons for N mineralization and nitrification rates (figs. 4 and 5). Koopmans and others (1995) have shown that increases in temperature greatly enhance N mineralization in N-saturated soils in Europe. Thus, the substantial increases in soil temperature caused by harvesting in our study, especially from early June to around mid September, caused significant increases in N mineralization in the soil. Furthermore, virtually all this enhancement was related to increases in nitrification, i.e., temperature-mediated increases in the activity of nitrifying bacteria.

Data from our study show also that there may be further increases in N mineralization/nitrification ratios under simulated conditions of additional N inputs and additions of

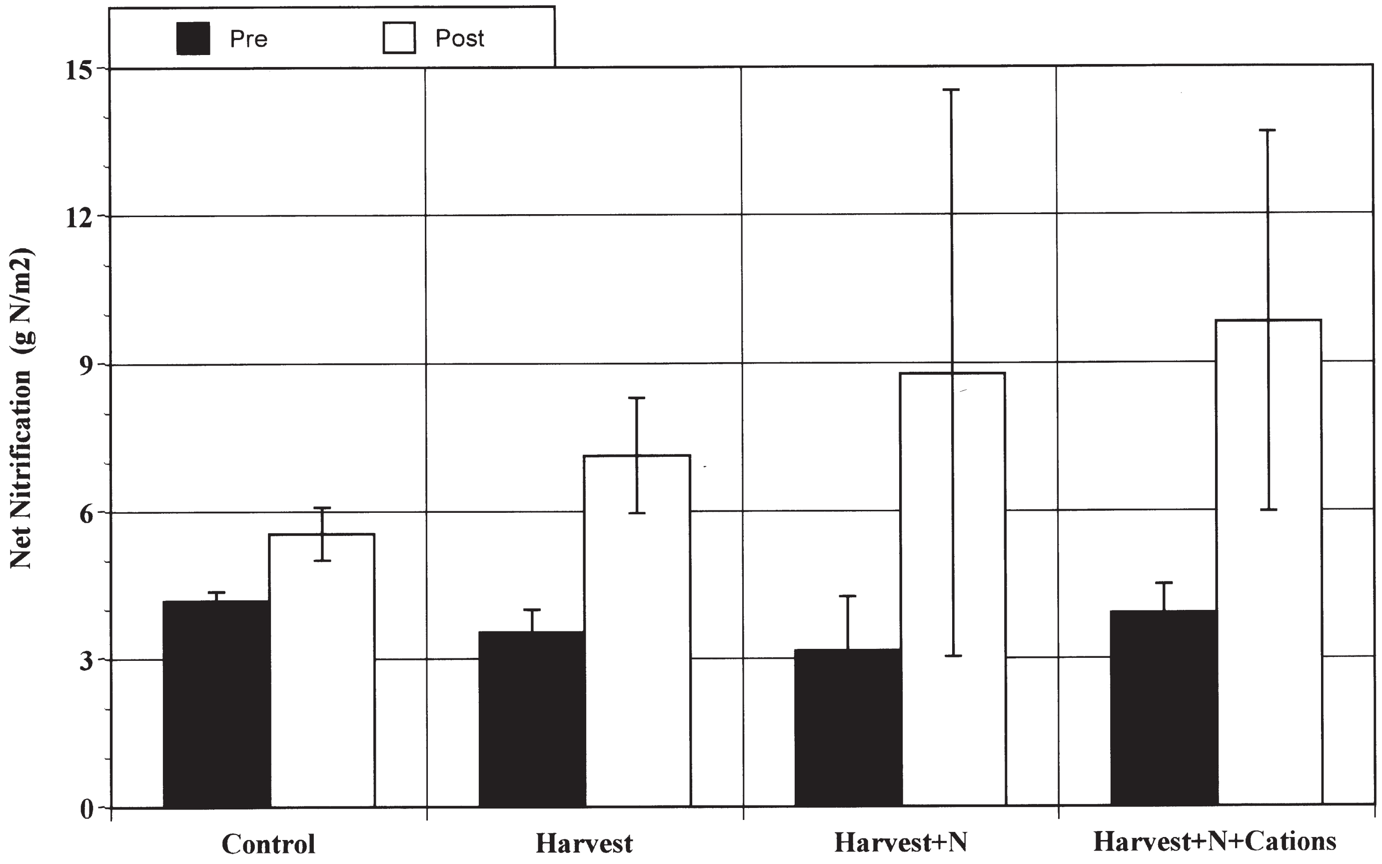

Figure 5—Net nitrification for pre- and post-treatment periods for control and harvest treatment plots at Fernow Experimental Forest, WV. Data are means plus and minus one standard error of the mean. Analysis of variance revealed no significant differences among pre-treatment means. Pre- versus post-treatment differences were significant for control plots at P<0.10 and for harvest and harvest+N+cations at P<0.05; difference for harvest+N plots was not statistically significant.

base cation designed to mitigate cation loss. Although differences were not significant among harvest treatments, there was a pattern of increases in N mineralization and nitrification related to N additions, and even greater increases with additions of base cations. This latter pattern suggests that mineralization and nitrification might be more pH limited than previously thought (Gilliam and others 1994, 1996). Future work at our site will test this hypothesis. Certainly, it suggests that managing central hardwood forests in a sustainable manner in the context of N saturation represents a complex challenge.

Implications for N Saturation in Central Hardwood Forests

Existing data from FEF strongly suggest a threat of base cation depletion and its relationship to N saturation in central Appalachian forests (Adams and others 1997, Gilliam and others 1996). Indeed, much evidence exists to suggest that soils at FEF already have become N saturated. For example, Gilliam and others (1996) found equally high rates of N mineralization based on one year's data from one N-treated (at rates identical to those in this study) and two untreated watersheds. Furthermore, nearly 100 percent of N mineralized was nitrified. On-going data from this study suggest a pattern of increasing rates through time for all three watersheds, regardless of treatment. Furthermore, available pools of N (extractable NH_4 and NO_3) were significantly higher on the treatment watersheds, suggesting that further additions of N to the treatment watershed are no longer resulting in increased uptake by plants or immobilization by microbes (Vitousek and Matson 1985).

Stoddard (1994) concluded that Watershed 4 (WS4—mature, second-growth forest) at FEF was one of the better examples of watersheds in the eastern United States which are at the later stages of N saturation. It is notable that WS4 supports a mature, mixed-age, second-growth hardwood stand and serves as the long-term control watershed at FEF, receiving no experimental treatments and, thus, it may be considered typical of many second-growth forests of the region (Gilliam and others 1996.

Using long-term (1971-1987) stream chemistry data from WS4 at FEF Edwards and Helvey (1991) found patterns of significant increases in specific conductivity with time. Using correlation analysis they concluded that most of this increase was related to increases in stream NO_3. Based on data from WS4 for an even longer time period (1969-1991), Peterjohn and others (1996) concluded that the patterns found by Edwards and Helvey (1991) were part of a larger scenario indicating N saturation for FEF.

The relative sensitivity of forest soils to accelerated cation leaching, pH depression, aluminum mobilization, and mineral weathering is largely unknown (Cronan and others 1990). Forest soils of the central Appalachians range from very resistant to very sensitive to such changes. Based on criteria established by Turner and others (1986), sensitive soils are those with combinations of low cation exchange capacity (CEC), intermediate base saturation, low pH, low levels of bases released via weathering, low sulfate adsorption capacities, and shallow soils. Given that a large percentage of soils in the Appalachian region falls within these criteria, data from this study strongly suggest that expected increases in timber harvesting, continued high levels of N deposition in the region, base cation depletion, and complications from N saturation all represent major threats to sustainability of central Appalachian forest ecosystems.

REFERENCES

Aber, J.D. 1992. Nitrogen cycling and nitrogen saturation in temperate forest ecosystems. Trends In Ecology And Evolution. 7: 220-223.

Aber, J.D.; McDowell, W.; Nadelhoffer, K. [and others]. 1998. Nitrogen saturation in temperate forest ecosystems: hypotheses revisited. BioScience. 48: 921-934.

Adams, M.B.; Angradi, T.R.; Kochenderfer, J.N. 1997. Stream water and soil solution responses to 5 years of nitrogen and sulfur additions at the Fernow Experimental Forest, West Virginia. Forest Ecology and Management. 95: 79-91.

Adams, M.B.; Eagar, C. 1992. Impacts of acidic deposition on high-elevation spruce-fir forests: Results from the Spruce-Fir Cooperative. Forest Ecology and Management. 51: 195-205.

Adams, M.B.; Edwards, P.J.; Wood, F.; Kochenderfer, J.N. 1993. Artificial watershed acidification on the Fernow Experimental Forest. Journal of Hydrology. 150: 505-519.

Auchmoody, L.R. 1972. Nutrient properties of five West Virginia forest soils. Res. Note NE-145. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station.

Binkley, D. 1986. Forest nutrition management. New York: John Wiley.

Binkley, D.; Driscoll, C.T.; Allen, H.L. [and others]. 1989. Acidic deposition and forest soils. New York: Springer-Verlag.

Cronan, C.S.; Driscoll, C.T.; Newton, R.M. [and others]. 1990. A comparative analysis of aluminum biogeochemistry in a northeastern and a southeastern forested watershed. Water Resources Research. 26: 1413-1430.

Eagar, C.; Adams, M.B. 1992. Ecology and decline of red spruce in the Eastern United States. New York: Springer-Verlag.

Edwards, P.J.; Helvey, J.D. 1991. Long-term ionic increases from a central Appalachian forested watershed. Journal of Environmental Quality. 20: 250-255.

Eno, C.F. 1960. Nitrate production in the field by incubating the soil in polyethylene bags. Soil Science Society of America Proceedings. 24: 277-279.

Federer, C.A.; Hornbeck, J.W.; Tritton, L.M. [and others]. 1989. Long-term depletion of calcium and other nutrients in Eastern U.S. forests. Environmental Management. 13: 593-601.

Fenn, M.E.; Poth, M.A.; Aber, J.D. [and others]. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecological Applications. 8: 706-733.

Gilliam, F.S.; Adams, M.B. 1995. Plant and soil nutrients in young versus mature central Appalachian hardwood stands. In: Gottschalk, K.W.; Fosbroke, S.L.C., eds. Proceedings, 10th central hardwood forest conference; 1995 March 5-8; Morgantown, WV. Gen. Tech. Rep. NE-197. U.S. Department of Agriculture, Forest Service: 109-118.

Gilliam, F.S.; Adams, M.B. 1996. Wetfall deposition and precipitation chemistry for a central Appalachian forest. Journal of the Air and Waste Management Association. 46: 978-984.

Gilliam, F.S.; Adams, M.B.; Yurish, B.M. 1996. Ecosystem nutrient responses to chronic nitrogen inputs at Fernow Experimental Forest, West Virginia. Canadian Journal of Forest Research. 26: 196-205.

Gilliam, F.S.; Richter, D.D. 1991. Transport of metal cations through a nutrient-poor forest ecosystem. Water, Air, and Soil Pollution. 57-58: 279-287.

Gilliam, F.S.; Turrill, N.L.; Adams, M.B. 1995. Species composition and patterns of diversity in herbaceous layer and woody overstory of clearcut versus mature central Appalachian hardwood forests. Ecological Applications. 5: 947-955.

Gilliam, F.S.; Turrill, N.L.; Aulick, S.D. [and others]. 1994. Herbaceous layer and soil response to experimental acidification in a central Appalachian hardwood forest. Journal of Environmental Quality. 23: 835-844.

Idso, S.B.; Aase, J.K.; Jackson, R.D. 1975. Net radiation-soil heat flux relations as influenced by soil water content variations. Boundary-Layer Meteorology. 9: 113-122.

Johnson, C.E. 1995. Soil nitrogen status 8 years after whole-tree clear-cutting. Canadian Journal of Forest Research. 25: 1346-1355.

Kahl, J.S.; Norton, S.A.; Fernandez, I.J. [and others]. 1993. Experimental inducement of nitrogen saturation at the watershed scale. Environmental Science and Technology. 27: 565-568.

Koopmans, C.J.; Lubrecht, W.C.; Tietema, A. 1995. Nitrogen transformations in two nitrogen saturated forest ecosystems subjected to an experimental decrease in nitrogen deposition. Plant and Soil. 175: 205-218.

Lusk, M.G. 1998. Sulfate dynamics and base cation release in a high elevation Appalachian forest soil. Blacksburg, VA: The Virginia Polytechnic Institute and State University. 204 p. M.S. thesis.

Mann, L.K.; Johnson, D.W.; West, D.C. [and others]. 1988. Effects of whole-tree and stem-only clearcutting on post-harvest hydrologic losses, nutrient capital and regrowth. Forest Science. 34: 412-428.

Marquis, D.A.; Johnson, R.L. 1989. Silviculture of eastern hardwoods. In: Burns, R.M., comp. The scientific basis for silvicultural and management decisions in the National Forest System. Gen. Tech. Rep. WO-55. U.S. Department of Agriculture, Forest Service: 9-17.

Martin, C.W. 1995. Development of a central hardwood stand following whole-tree clearcutting in Connecticut. In: Gottschalk, K.W.; Fosbroke, S.L.C., eds. Proceedings, 10th central hardwood forest conference; 1995 March 5-8; Morgantown, WV. Gen. Tech. Rep. NE-197. U.S. Department of Agriculture, Forest Service: 37-46.

Norton, S.A.; Kahl, J.S.; Fernandez, I.J. [and others]. 1994. Response of the West Bear Brook Watershed, Maine, USA, to the addition of $(NH_4)_2SO_4$: 3-yr results. Forest Ecology and Management. 68: 61-73.

Peterjohn, W.T.; Adams, M.B.; Gilliam, F.S. 1996. Symptoms of nitrogen saturation in two central Appalachian hardwood forest ecosystems. Biogeochemistry. 35: 507-522.

Rosenberg, N.J.; Blad, B.L.; Verma, S.B. 1983. Microclimate: the biological environment. New York: John Wiley.

Rustad, L.E.; Fernandez, I.J.; Fuller, R.D. [and others]. 1993. Soil solution response to acidic deposition in a northern hardwood forest. Agriculture, Ecosystems and Environment. 47: 117-134.

Stoddard, J.L. 1994. Long-term changes in watershed retention of nitrogen: its causes and aquatic consequences. In: Baker, L.A., ed. Environmental chemistry of lakes and reservoirs: ACS Advances in Chemistry Series 237. Washington, DC: American Chemical Society.

Tritton, L.M.; Martin, C.W.; Hornbeck, J.W.; Pierce, R.S. 1987. Biomass and nutrient removals from commercial thinning and whole-tree clearcutting of central hardwoods. Environmental Management. 11: 659-666.

Turner, R.S.; Olson, R.J.; Brandt, C.C. 1986. Areas having soil characteristics that may indicate sensitivity to acidic deposition under alternative forest damage hypotheses. Oak Ridge National Laboratory ORNL/TM-9917.

Twardus, D.B. 1995. Forest health assessment for eastern hardwood forests. In: Gottschalk, K.W.; Fosbroke, S.L.C., eds. Proceedings, 10th central hardwood forest conference; 1995 March 5-8; Morgantown, WV. Gen. Tech. Rep. NE-197. U.S. Department of Agriculture, Forest Service: 3-10.

Vitousek, P.M.; Matson, P.A. 1985. Disturbance, nitrogen availability, and nitrogen losses in an intensively managed loblolly pine plantation. Ecology. 66: 1360-1376.

World Resources Institute. 1994. World resources 1994-95. New York: Oxford University Press.

Zar, J.H. 1994. Biostatistical analysis, 3rd ed. Englewood Cliffs, NJ: Prentice-Hall.

N DYNAMICS ACROSS A CHRONOSEQUENCE OF UPLAND OAK-HICKORY FORESTS

T.W. Idol, P.E. Pope and F. Ponder, Jr.[1]

Changes in soil physical, chemical, and biological properties due to harvesting practices may have a significant impact upon N availability and cycling, thus future stand productivity and composition. Our study was designed to assess monthly rates of N min, nitrification, and N uptake by forest vegetation during the growing season. We selected five oak-hickory stands in Dubois County, Indiana for this study. We installed and collected PVC cores on a monthly basis from April to December. Soils were then analyzed for ammonium and nitrate.

Results show that seasonal N cycling rates are variable in all stands (figure 1). On an annual basis, the mature stand showed the highest rates of N min and N uptake and the lowest percentage of mineralized N converted to nitrate (29 percent) (table 1). The youngest stand had the lowest N min rate and the highest percentage of mineralized N converted to nitrate (77 percent). It was the only stand in which N min rates were appreciably lower than N uptake rates. The soil core method underestimates true N min rates within this stand because N min from logging slash is not reflected in the cores. For the other stands, there appears to be a tight internal cycling of N, preventing leaching losses.

Table 1—N cycling in a chronosequence of hardwood forests

Stand age	N mineralization	Nitrification	N uptake
	-----------------*g N/hectare/day*-----------------		
1	87.46	60.45	154.98
6	182.95	64.08	196.20
12	161.46	56.37	212.81
31	186.18	93.96	246.34
80-100	392.01	115.57	357.38

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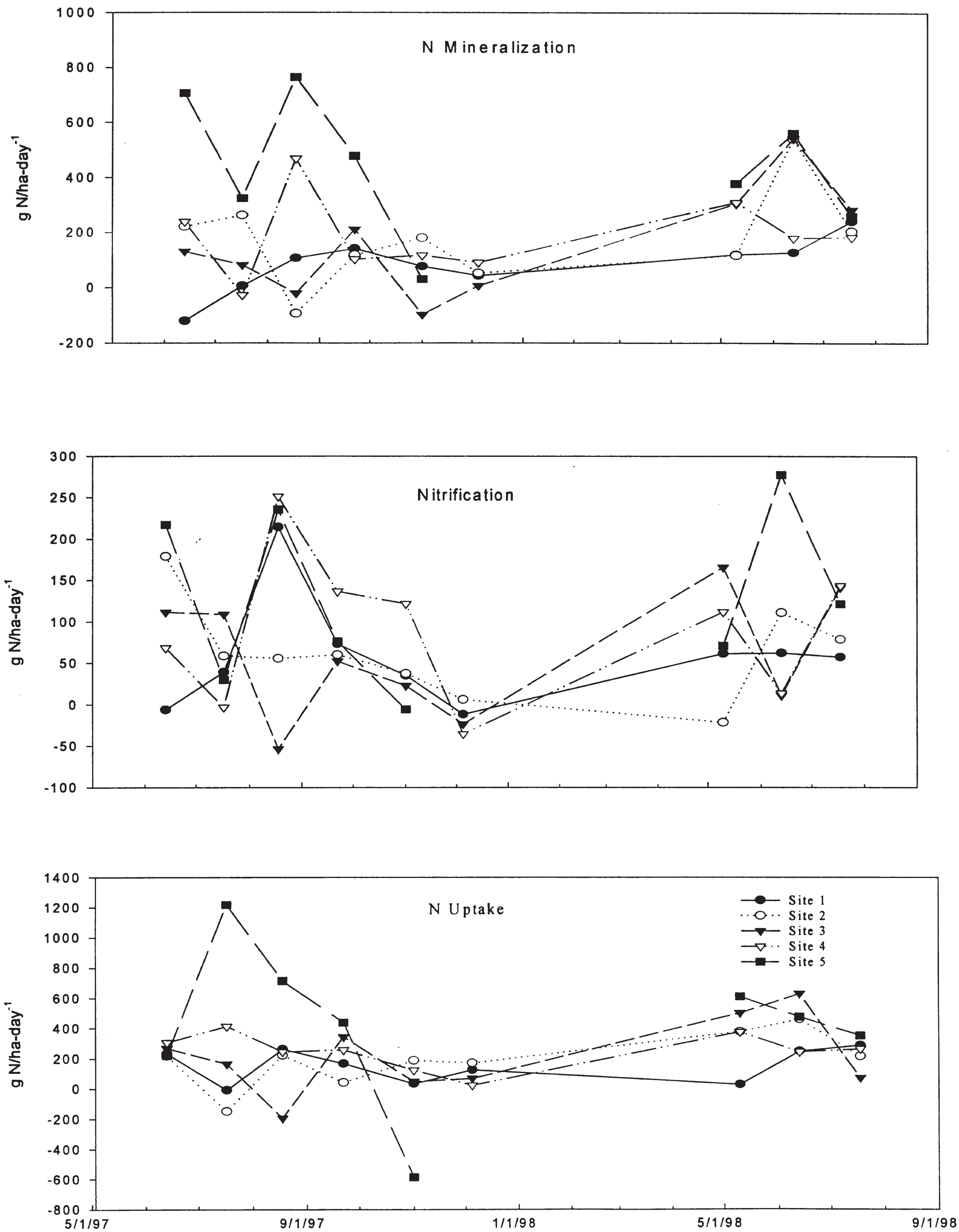

Figure 1. Nitrogen cycling across a chronosequence of hardwood forests. Stands cut in 1996, 1991, 1985, 1966 and 1900-1920, respectively.

SOIL NUTRIENT AND MICROBIAL RESPONSE TO PRESCRIBED FIRE IN AN OAK-PINE ECOSYSTEM IN EASTERN KENTUCKY

Beth A. Blankenship and Mary A. Arthur[1]

Abstract—On the Cumberland Plateau, periodic fire may be necessary to maintain oak-dominated stands on xeric ridgetops. However, thin organic horizons and nutrient poor soils may limit the beneficial effects of fire by further reducing nutrient pools. The USDA Forest Service has reintroduced fire to oak-pine ridgetops in the Daniel Boone National Forest in the form of single, late winter prescribed burns. Objectives of this study were to document effects of single prescribed fires conducted in March 1995 and March 1996 on soil nutrients and microbial biomass. In 1996, forest floor mass was determined before and after fire. In 1995 and 1996, total and available nitrogen, total carbon, pH, extractable cations, and microbial biomass were measured pre-burn and throughout the year after burning on burned and unburned sites. Thirty-two percent of the litter layer (Oi) was combusted in 1996, while no loss of the Oe and Oa horizons was found. A transitory increase in available nitrogen was detected in burned mineral soils. Increases in pH by 0.2-0.3 units were measured in the burned organic horizons. Lower concentrations of extractable cations were measured in burned soils than in unburned soils. Fire had a positive effect on active bacterial biomass, but no effect on fungal biomass. Our study suggests that single, late winter prescribed fire had minimal effects on belowground resources in these ridgetop ecosystems. Since repeated burning might be necessary to promote oak regeneration, future research must address the effects of repeated burning on soil resources.

INTRODUCTION

Fire has played an integral role in maintaining oak-pine forest communities in the southern Appalachians for as long as 3000 years BP (Delcourt and Delcourt 1997). Aboriginal use of fire was common and settlers of European descent continued fire use in the area; however, as forest use altered, fire regimes changed (Pyne 1982, Pyne and others 1996). This resulted in shifting forest composition, most noticeably as an increasing dominance of red maple (*Acer rubrum* L.) and decreasing regeneration of oak species (*Quercus* sp.) (Arends and McCormick 1987, Lorimer 1993). Ecologists and managers are increasingly promoting the use of fire to address problems of oak decline and flourishing fire-sensitive competitors like red maple.

In the Daniel Boone National Forest (DBNF) in eastern Kentucky, active fire suppression since the 1940s has successfully excluded fire on most oak-pine ridgetops (Martin 1989). Increasing regeneration of white pine (*Pinus strobus* L.) has been documented and ascribed to fire suppression (Wehner 1991), while charcoal and pollen analyses have demonstrated a growing presence of red maple and blackgum (*Nyssa sylvatica* Marshall) over the past 100 years (Delcourt and Delcourt 1997). Because of such changes in forest composition, the USDA Forest Service on the Stanton Ranger District has begun conducting late winter prescribed fires to restore fire to these ridgetop ecosystems (Richardson 1995). We have documented the effectiveness of late winter prescribed fire in reducing competition by fire-sensitive competitors (Arthur and others 1998, Blankenship [In Press]), and studies from other areas have recommended prescribed fire to improve oak regeneration (Barnes and Van Lear 1998, Brose and Van Lear 1998, Nyland and others 1983, Thor and Nichols 1974, Van Lear and Waldrop 1989).

Complicating the use of prescribed fire in eastern Kentucky is the problem of accidental and incendiary fires burning large acreage during the spring and fall seasons each year (Environmental Quality Commission 1992). Consequently, the use of prescribed fire on public lands is perceived by some to send a misleading message about fire to the general public. Because of thin organic soil horizons and low soil nutrient concentrations on ridgetops in the Red River Gorge, fire potentially could, if too hot, contribute to a loss of nutrients already in low supply. The effects of fire on soil microbial biomass may also be important because of the role fungi and bacteria play in mediating nutrient mineralization and availability. An improved understanding of the soil nutrient and biological response, as well as the plant community response, to prescribed fire is necessary to elucidate the ecological differences between the effects of prescribed fire and the potentially hotter and more frequent incendiary fires. This type of information is essential to the public debate regarding the role of prescribed fire in the management of public forestlands.

The objectives of this study were to document effects of a single, late-winter prescribed fire on soil nutrients and microbial biomass as a first step in addressing impacts of different fire regimes in these ridgetop ecosystems. Despite the plethora of studies regarding effects of fire on soil nutrients and microorganisms in forests throughout the United States and elsewhere, few studies have examined effects of burning on soil resources in oak-pine forests in the southern Appalachians (Clinton and others 1996, Vose and others [In Press], Vose and Swank 1993) and none on the Cumberland Plateau in eastern Kentucky.

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METHODS

Site Description

Permanent study plots are located on three noncontiguous ridgetops in the Stanton Ranger District of the DBNF in the Red River Gorge Geological Area. The Red River Gorge is located in the Cliff Section of the Cumberland Plateau (Braun 1950). The geological substrate is composed of shales and siltstones of the Upper and Lower members of the Breathitt Formation and the Corbin Sandstone of the Lee Formation (Weir and Richards 1974). The ridgetops, Pinch-Em-Tight Ridge, Whittleton Ridge and Klaber Ridge, are located in Powell, Wolfe and Menifee counties of eastern Kentucky. Study plots are located in stands mostly dominated by scarlet oak (*Q. coccinea* Muenchh.) and chestnut oak (*Q. prinus* L.), with some white (*Q. alba* L.) and black oak (*Q. velutina* Lam.). Pines (*P. rigida* Mill., *P. virginiana* Mill., and *P. echinata* Mill.) dominate the most xeric areas. Red maple is abundant in mid- to overstory positions. Sourwood (*Oxydendrum arboreum* (L.)DC), blackgum and dogwood (*Cornus florida* L.) are found in the understory. Eastern white pine seedlings and saplings are plentiful and a few large white pines occur in the overstory. Heath shrubs such as mountain laurel (*Kalmia latifolia* L.) and blueberry (*Vaccinium* spp.) are common throughout.

Soils at Whittleton Ridge are composed of Gilpin silt loam, a moderately deep, well drained soil with a lower subsoil of silty clay loam of the subgroup Typic Hapludult (Hayes 1993). Soils at Klaber Ridge are similar and classified as Latham-Shelocta silt loam, moderately deep, moderately well drained, slowly permeable clayey soils and of the subgroups Typic and Aquic Hapludults (Avers and others 1974). Soils at Pinch-Em-Tight are of Alticrest-Ramsey-Rock outcrop complex, moderately deep and shallow, well drained with surface layer and subsoil of sandy loam, and are Typic and Lithic Dystrochrepts (Hayes 1993, SCS 1975). The region has a temperate, humid and continental climate with an evenly distributed average annual precipitation of about 113 centimeters. Mean daily maximum and minimum temperatures in January are 6°C and –6°C, and in July are 30°C and 17°C. Mean annual temperature is 12°C (Hill 1976).

Experimental Design

Each ridge (Klaber, Pinch-Em-Tight and Whittleton) was divided into three areas: no burning, burned in March 1995, and burned in March 1996. This yielded two treatments (burned and unburned) per year of study (1995 and 1996). Study plots were located at random, with the condition that any location within 15 meters of the slope break, a trail, or any human disturbance was rejected. We installed eight 0.04 hectare plots per treatment except at Pinch-Em-Tight Ridge. That site is topographically narrow and traversed with trails and campsites, and we could fit only 6 plots per treatment. The 1995 burn treatment on Pinch-Em-Tight was especially narrow and these plots were laid out systematically in a line, 30 meters apart along the top of the ridge. Centers of all plots were permanently marked, and two trees in each plot were tagged and located (azimuth and distance to plot center) for future reference. While plots were located at random within each treatment area, treatments were applied nonrandomly to treatment areas because of the necessity of burning a contiguous finger ridge not in contact with privately owned land.

Fire Prescription

USDA Forest Service personnel of the Stanton Ranger District conducted the 1995 and 1996 prescribed fires. The fires were ignited with drip torch by firing line from the highest point and ridges first. From the ridges, strips were pulled downslope into the wind. If backing and flanking fires were not of sufficient intensity (>0.3 m flamelength), point-source and strip firing were used to increase intensity to acceptable levels (Richardson 1995). Wind speeds during the 1995 fires were from 0 to 3.2 km/h, and in 1996 were around 1.6-4.8 km/h. Flame heights for all burns were 0.3-0.9 meters (Richardson 1995, Richardson 1996). In 1996, the prescribed burning planned for Whittleton Ridge could not be conducted due to unsuitable wind direction that would have allowed smoke to blow over an adjacent highway.

Temperatures of prescribed fires were measured using 6 different Tempilac® temperature-sensitive paints chosen to represent a temperature range of 93°C to 576°C (Cole and others 1992). A stripe of each paint was applied on aluminum tags and vertically down strips of mica. The aluminum tags were stapled to stakes at heights of 15, 45 and 75 centimeters. The stakes and the mica sheets were placed at 4 meters from each plot center on north, south, east, and west bearings. The mica sheets were inserted into the ground to a depth of 9 centimeters below the litter surface.

Forest Floor Sampling

Forest floor was collected on all sites in February 1996 prior to burning, and in May 1996 following prescribed burning. One 27.5 x 27.5 centimeter sample was collected along three randomly selected azimuths in each plot (3 samples per plot, mean of the three used for analysis). Litter (Oi) was separated from the Oe and Oa horizons in the field and measured separately.

Soil Sampling

Soil samples for nutrient and microbial biomass analyses were taken within a 0.04 hectare plot by sampling along two randomly selected 11 meter radii originating at the center point of each permanently marked plot. Soil samples were stored at 4°C until processed. Nutrients (C, Ca, K, Mg, N, and P) were analyzed on soil samples from a composite of two 4.0 centimeter cores taken at 6.0 meters along each radius. The organic horizon (Oea) was separated from the mineral horizon in the field with mineral soils sampled to a depth of 5 centimeters. Samples for nutrient analyses were taken within 30 days pre-burn, after the first rainfall post-burn (1 week post-burn in 1995 and 2 weeks post-burn 1996), and in September.

For microbial biomass, two soil cores of 2.0 centimeter diameter were taken at 1.5 meter intervals along each radius beginning at 3.0 meters from the center for a total of 24 cores collected at 12 sampling points. Mineral soils were sampled to a depth of 5 centimeters. Organic and mineral horizons were separated and soil cores composited into a single sample for analysis from each plot. Microbial biomass samples were collected more frequently than nutrient samples because of the expectation of an immediate (1 day) response to fire as well as an attenuation of the effect with time. In 1995, samples for microbial biomass were collected within 30 days pre-burn, the day after the burn, 1 week post-burn (followed first rainfall post-burn), and every 4 weeks until July, then every 6 weeks thereafter through November. Sampling frequency was reduced after July 1995 following examination of results from earlier in 1995. In 1996, after sampling 2 weeks post-burn (following first rainfall post-burn), microbial biomass samples were collected post-burn only every 6 weeks through September 1996.

Available nitrogen (ammonium- and nitrate-nitrogen) was measured on mineral soil samples collected for microbial biomass estimates. In 1995 no pre-burn measurements of available nitrogen were made; both pre- and post-burn measurements of available nitrogen were made in 1996.

Laboratory Analysis

Forest floor samples were oven-dried at 60°C and then weighed. Mehlich III-extractable P, Ca, Mg, and K were determined with the Mehlich III reagent method (Tran and Simard 1993), and analysis by an inductive coupled plasma spectrophotometer. Organic horizon pH was measured in a 1:10 soil:$CaCl_2$ mixture, and a subset of these measured in a 1:2 soil:water mixture. Statistical analysis was conducted on the organic pH values from the $CaCl_2$ method. The mineral soil pH was measured in a 1:2 soil:water mixture (Hendershot and Lalande 1993). Total C and N were analyzed using a Leco C/N analyzer. Determination of ammonium- and nitrate-nitrogen in the mineral soil was made using a Technicon II auto-analyzer following KCl extraction (Maynard and Kalra 1993).

Active and total fungal and bacterial biomass were calculated from direct counts of hyphal length (fungi) and number of organisms (bacteria) using direct count epifluorescent microscopy (Ingham and Klein 1984, Ingham and others 1991, Lodge and Ingham 1991). Fluorescein diacetate was used to stain metabolically active fungi and bacteria; total fungal hyphae (metabolically active and inactive) were counted using direct light source. Direct counts of fungi and bacteria were carried out within one week of sample collection. Based upon the assumption that hyphal tissue density averages 410 mg cm^{-3} and bacterial tissue density averages 330 mg cm^{-3} (Ingham and others 1991), and using estimates of hyphal and bacterial diameters, active fungal and bacterial biomass were calculated from the volume of fungi or bacteria in 1 gram of dry forest floor.

Statistical Analysis

We analyzed the data in two ways: (1) by comparing pre-burn to post-burn within each treatment (burned or unburned) using a t-test, and (2) by comparing the burned treatment to the unburned by analysis of variance. A covariate analysis using pre-burn parameters as the covariate was used for the nutrient data to compare burned to unburned. Statistical analysis of pH was performed on $[H^+]$ and results were converted to pH values for presentation. On microbial biomass and available nitrogen, Levene's (1960) homogeneity of variance test revealed that often group variances within a treatment were significantly different, so the Welch ANOVA for means was used (Brown and Forsyth 1974, Welch 1951).

Initially, the three ridgetops were selected as replicates for treatment for each year with treatment area as the experimental unit (n=6; 3 burned and 3 not burned), but in 1996 one ridgetop was not burned as planned, so analysis of the 1996 data had only 2 replicates (n=4; 2 burned, 2 not burned). As the study progressed we became aware of inherent site differences. Therefore, statistical analyses were also conducted on a pseudo-replicated design (Hurlburt 1984) using the 8 (or 6) plots per treatment area as the experimental unit with ridge as block and significance determined at $p<0.10$ (Table 1). If these analyses revealed significant site-by-treatment interactions in addition to significant treatment effects, subsequent analyses were conducted on reduced models comparing the burned treatment to the unburned reference at each ridgetop location (Klaber and Whittleton: n=16; Pinch-Em-Tight: n=12). All statistical

Table 1—Experimental design for study of prescribed burns conducted in March 1995 and March 1996 in the Daniel Boone National Forest, Kentucky

1995			1996		
Ridge/block[a]	Plots	Treatments	Ridge/block	Plots	Treatments
Klaber Ridge	8	Burned/unburned	Klaber Ridge	8	Burned/unburned
Pinch-Em-Tight Ridge	6	Burned/unburned	Pinch-Em-Tight Ridge	6	Burned/unburned
Whittleton Ridge	8	Burned/unburned			
Total: 3	22	2	2	14	2

[a] Experimental unit was the ridge (n=3 in 1995, n=2 in 1996) or the plot with ridge as block (n=22 in 1995, n=14 in 1996).

tests were conducted with the JMP® statistical package (SAS Institute 1995).

RESULTS

Pre-Burn Soil Chemistry

Thin, acidic, nutrient-poor organic horizons characterized these ridges prior to burning. Organic horizons (Oea) ranged from <1 to 4-5 centimeters in thickness with a mean pH of 3.70. Total nitrogen averaged 0.66 percent and total carbon 54 percent. Mineral soil pH was slightly higher at 4.24, with total nitrogen averaging 0.14 percent and carbon 4.9 percent.

Fire Temperatures

Surface temperatures ranged from 316°C to 398°C in 1995 and from 204°C to 315°C in 1996. Temperatures within the Oea horizons at 0.5 centimeters below the surface ranged from 204-315°C in 1995 and from 93-203°C in 1996.

Forest Floor Mass

Pre-burn forest floor mass averaged 98.9 grams/meter^2 for the Oi layer and 184 grams/meter^2 for Oea layers. The prescribed fires of 1996 combusted an average of 32 percent of the litter layer (Oi) without combusting the underlying Oe and Oa layers. The decrease in Oi because of the fires was highly significant (p<0.01). Forest floor mass measurements were not made in 1995.

Soil Nitrogen

The inorganic nitrogen in the mineral soils primarily took the form of NH_4-N. Little, if any, NO_3-N was measured. In 1995, available nitrogen concentration was significantly higher in the burned treatments than in the unburned treatments 1 day after the burns on Klaber Ridge (12.5 vs. 1.89 µg/g dry wt soil; p<0.05) and Whittleton Ridge (8.43 vs. 4.89 µg/g dry wt soil; p<0.10), but not on Pinch-Em-Tight Ridge (Table 2). One week later there were no significant differences.

In 1996, available nitrogen concentration declined from the pre-burn sampling to the post-burn sampling dates, significantly so in the unburned areas (3.23 to 1.25 µg/g dry wt soil; p<0.0001), but not significantly in the burned treatments based on the pre- vs. post-burn t-tests. Two weeks later, available nitrogen concentrations were significantly lower in the burned treatments (1.28 µg/g dry wt soil) than in the unburned (2.43 µg/g dry wt soil; p<0.10) using covariate analysis. In September 1996, total nitrogen in the mineral horizon was significantly lower in the burned treatments (0.09 pct. vs. 0.15 pct; p<0.01). No consistent effect of fire on total carbon was found.

pH

O horizon pH was significantly higher in the 1995 burned treatments one week post-burn (p<0.05) and in September 1995 (p<0.10), but there were significant site by treatment interactions (p<0.05). Analysis on the reduced models

Table 2—Mean available N (NH_4-N + NO_3-N; µg/g dry wt soil) in mineral horizon of burned and unburned treatments on Klaber, Whittleton, and Pinch-Em-Tight Ridges, Daniel Boone National Forest, Kentucky, following 1995 prescribed fires and prior to and following 1996 prescribed fires

	Klaber		Whittleton		Pinch-Em-Tight	
Date	Burned	Unburned	Burned	Unburned	Burned	Unburned
1 day post-burn 1995	12.5[a] (3.1)[d]	1.89 (0.66)	8.43[a] (1.7)	4.89 (0.92)	3.59 (0.99)	4.17 (2.0)
1 wk post-burn 1995	3.72 (0.44)	3.37 (0.22)	4.90 (0.39)	4.86 (0.14)	4.14 (0.57)	3.46 (0.13)
Pre-burn 1996	1.95 (0.38)	2.37 (0.14)			2.79 (0.16)	4.10 (0.43)
1 day post-burn 1996[b]	1.60 (0.62)	1.24 (0.23)			2.21 (0.11)	1.26 (0.13)
2 wk post-burn 1996	1.49[c] (0.57)	3.01 (0.48)			1.40c (0.34)	1.64 (0.33)

[a] Burned treatment significantly different from unburned treatment of same ridge at p<0.10.
[b] Pre-burn vs. post-burn t-tests showed burned treatments not significantly different but unburned treatments significantly different at p<0.0001.
[c] Burned treatments significantly different from unburned treatments by covariate analysis at p<0.10.
[d] Standard errors (in parentheses) are based upon mean within a treatment.

revealed that burned treatments had significantly higher pH 1 week post-burn at Whittleton Ridge (3.64 vs. 3.18; p<0.001) and in September at Klaber Ridge (3.64 vs. 3.23; p<0.01) (Table 3). The 1995 fires did not affect pH in the upper mineral soils.

The 1996 fires significantly increased organic horizon pH from pre-burn values to 2 weeks post-burn (3.01 to 3.24; p<0.10), even though the pH of the burned treatments did not significantly differ from the unburned treatments 2 weeks post-burn (p=0.79). In September 1996, we found significant differences in pH between burned and unburned treatments in both the organic and mineral horizons with significant site by treatment interactions. The reduced model analyses showed significantly lower pH in Klaber burned organic horizon (3.20 vs. 3.53; p<0.01), while Pinch-Em-Tight burned treatments had significantly higher pH in organic (3.17 vs. 3.08; p<0.001) and mineral horizons (4.29 vs. 3.89; p<0.05).

Cations

Burned treatments contained significantly lower extractable cation concentrations than unburned treatments later in the season. Extractable potassium concentrations were significantly lower in burned organic horizons in September 1995 (362 vs. 439 µg/g dry wt soil; p<0.10) and September 1996 (362 vs. 490 µg/g dry wt soil; p<0.10) (Table 4). In September 1996, magnesium and calcium concentrations were also lower in burned organic horizons (164 vs. 246 µg/g dry wt soil; p<0.01; 1240 vs. 2500 µg/g dry wt soil; p<0.01, respectively).

Extractable cations in the burned upper mineral soil increased or were higher soon after the fires, but by September tended to be significantly lower than in unburned soils. In 1995, significantly higher potassium concentrations were found in the burned treatments one week post-burn (86.7 vs. 84.0 µg/g dry wt soil; p<0.05). In the 1996 burned treatments, significant increases of magnesium (19.4 to 25.2 µg/g dry wt soil; p<0.05) and calcium (100 to 135 µg/g dry wt soil; p<0.05) concentrations from pre-burn to 2 weeks post-burn were found. In September 1995, potassium was significantly lower in burned treatments (62.4 vs. 71.7 µg/g dry wt soil; p<0.10), and in September 1996 lower in the burned treatment of Klaber Ridge (64.6 vs. 89.6 µg/g dry wt soil; p<0.10). In September 1995 and 1996, burned treatments had significantly less magnesium than unburned treatments (1995: 21.6 vs. 25.0 µg/g dry wt soil; p<0.05; 1996: 16.9 vs. 28.7 µg/g dry wt soil; p<0.10). Calcium concentrations were also significantly lower in burned treatments than in unburned in September 1996 (84.5 vs. 169 µg/g dry wt soil; p<0.05).

Microbial Biomass

In the O horizon, microbial biomass was dominated by fungi (32 µg/g dry wt soil) rather than bacteria (14 µg/g dry wt soil), whereas in the mineral horizon, microbial biomass of bacteria (11.3 µg/g dry wt soil) and fungi (11.4 µg/g dry wt soil) were similar.

Table 3—Mean pH values for O horizon of burned and unburned treatments of Klaber, Whittleton, and Pinch-Em-Tight Ridges, Daniel Boone National Forest, Kentucky, prior to and following March 1995 and March 1996 prescribed fires

	Site					
	Klaber		Whittleton		Pinch-Em-Tight	
Date	Burned	Unburned	Burned	Unburned	Burned	Unburned
Pre-burn 1995	3.77	3.69	3.20	3.31	3.09	3.05
1 wk post-burn 1995	3.93	3.58	3.64	3.18[a]	3.21	3.16
September 1995	3.64	3.23[a]	3.05	2.93	2.88	2.85
Pre-burn 1996	3.10	3.30			2.94	3.10
2 wk post-burn 1996	3.32[b]	3.63			3.25[b]	3.14
September 1996	3.20	3.53			3.17	3.08

[a] Covariate analysis of $[H^+]$ shows burned treatment significantly different from unburned within ridge at p<0.01 using pre-burn $[H^+]$ as covariate.
[b] Analysis of pre-burn and post-burn $[H^+]$ shows post-burn significantly different from pre-burn at p<0.10.

Table 4—Mean extractable cations (µg/g dry wt soil) in organic and mineral horizons of Burned and Unburned treatments of three ridgetops in Daniel Boone National Forest, Kentucky, prior to and following March 1995 and March 1996 prescribed fires

	O horizon				Mineral soil			
	1995		1996		1995		1996	
	Burned	Unburned	Burned	Unburned	Burned	Unburned	Burned	Unburned
K								
Pre-burn	495	469	467	510	80.5	85.9	75.5	84.3
	(30)[a]	(29)	(39)	(39)	(6.7)	(6.3)	(5.3)	(4.9)
Post-burn	513	558	473	433	86.7	84.0	95.8	102
	(35)	(34)	(47)	(47)	(5.9)	(5.9)	(6.3)	(6.0)
September	362	439	362	490	62.4	71.7	61.1	75.3
	(25)	(24)	(22)	(23)	(5.0)	(4.7)	(4.7)	(4.4)
Mg								
Pre-burn	198	340	190	226	25.3	29.1	19.4	22.2
	(93)	(90)	(20)	(20)	(2.7)	(2.0)	(3.0)	(6.0)
Post-burn	228	216	209	204	24.7	25.6	25.2	29.5
	(12)	(12)	(22)	(22)	(1.8)	(1.3)	(1.5)	1.4)
September	165	190	164	246	21.6	25.0	16.9	28.7
	(13)	(12)	(13)	(14)	(1.4)	(1.7)	(2.5)	(2.4)
Ca								
Pre-burn	2010	2040	1440	2460	148	172	100	147
	(168)	(163)	(281)	(133)	(12)	(14)	(7.8)	(22)
Post-burn	2310	2060	1560	1800	144	130	135	159
	(182)	(173)	(263)	(262)	(11)	(7.5)	(11)	(11)
September	1900	1560	1240	2500	134	127	84.5	169
	(129)	(123)	(180)	(188)	(14)	(8.0)	(23)	(23)

[a] Standard errors (in parentheses) are based upon mean within a treatment.

The prescribed fires of 1995 and 1996 did not consistently affect active fungal biomass, but active bacterial biomass in the organic horizon appeared to be positively affected by both the 1995 and 1996 burns. In the 1995 burned treatments of Whittleton and Pinch-Em-Tight Ridges, active bacterial biomass was higher than the unburned for almost every sampling date post-burn. Burned treatment active bacterial biomass was significantly higher in July 1995 (13 vs. 10 µg/g dry wt soil; $p<0.05$), in September 1995 (15 vs. 9 µg/g dry wt soil; $p<0.01$), and in November 1995 (20 vs. 16 µg/g dry wt soil; $p<0.10$) (Table 5). In May 1995, active bacterial biomass was significantly lower in the burned treatments (17 vs. 20 µg/g dry wt soil; $p<0.10$). In the 1996 burned treatments, active bacterial biomass was significantly higher 2 weeks post-burn (17 vs. 12 µg/g dry wt soil; $p<0.05$) and in August 1996 (7.3 vs. 5.2 µg/g dry wt soil; $p<0.05$). No significant effect on active bacterial biomass was found in the mineral soils.

DISCUSSION

Increases in nutrient availability following fire have been found in some systems and fire regimes and not in others. More intense fires of logging slash in the southern Appalachians have combusted some of the organic layer without significant loss of C or N from the Oea horizon (Vose and Swank 1993), and have also increased available soil nitrogen (Knoepp and Swank 1993). A stand replacement fire in this region resulted in no detectable change in soil chemistry (Vose and others [In Press]). Other fire studies in other regions show significant increases in soil nutrient concentrations following burning (Austin and Baisinger 1955, Prieto-Fernandez and others 1993).

The oak-pine ridgetops in the Red River Gorge are characterized by a thin litter layer (Oi) atop a very acidic, thin, nutrient-poor organic horizon (Oea). As none of the organic layer (Oea) combusted in these prescribed fires, the fires caused no immediate nutrient loss from the Oea horizon, and with only a fraction of the litter layer consumed, the increase of nutrient concentrations was very small. Also on such a nutrient-poor site, any addition of nutrients may have been quickly taken up by plants or immobilized by microbes (Hungerford and others 1990). While cations in the O horizon did not significantly change soon after burning, significant increases of some cations in the mineral horizon after burning suggest increased mineralization and possibly leaching of nutrients from the

Table 5—Mean active microbial biomass (µg/g dry wt soil) in the O horizon of Burned and Unburned treatments of three ridgetops in Daniel Boone National Forest, Kentucky, following March 1995 and March 1996 prescribed fires

	O horizon			
	1995		1996	
	Burned	Unburned	Burned	Unburned
1 d post-burn	15.5 (0.99)[a]	14.9 (0.97)	10.4 (2.1)	9.36 (1.1)
1 wk post-burn	15.9 (1.7)	13.2 (1.3)	—	—
2 wk post-burn	—	—	17.4 (0.90)	11.8 (1.2)
April	25.2 (2.1)	20.5 (2.0)	—	—
May	16.5 (0.93)	19.8 (1.6)	13.0 (1.0)	11.7 (2.9)
June	5.53 (1.2)	11.7 (1.3)	11.4 (0.59)	10.4 (0.63)
July/August	12.8 (1.1)	10.1 (0.73)	7.33 (0.45)	5.16 (0.90)
September	14.7 (1.7)	8.66 (0.69)	8.27 (0.53)	7.84 (2.2)
November	20.3 (2.0)	15.6 (1.3)	—	—

[a] Standard errors (in parentheses) are based upon mean within a treatment.

ash through the thin organic horizon to the mineral soil. We detected a transient increase in available nitrogen, and we know that some increased nutrient availability occurs as evidenced in increased seedling foliar N, P, and K concentrations following burning (Gilbert and others 1998, Reich and others 1990). In another acidic, nutrient-poor ecosystem, the southeastern Coastal Plain pine flatwoods, herb layer vegetation had significant increases in tissue nutrient concentrations for N, P and K following winter fire (Gilliam 1991).

Availability of nutrients for plant uptake depends in part on soil pH, which often changes after burning as oxides in ash react with hydrogen ions to raise pH (Agee 1993, Ahlgren and Ahlgren 1960). We found significantly higher pH in burned soils both years. Higher pH may also have favored bacterial populations (Ahlgren 1974) which increased following burning. While many studies show decreased microbial populations after fire (Ahlgren 1974, Borchers and Perry 1990, Pietikäinen and Fritze 1993), other studies have found bacterial populations to increase following burning (Ahlgren and Ahlgren 1965, Jurgensen and others 1981). Lower concentrations of available cations measured late in the season may reflect increased immobilization by increasing bacterial biomass (Fritze and others 1993).

Fire is being reintroduced to these ridgetops in an attempt to maintain a community diversity that has been decreasing since fire suppression. Successful regeneration of oak species is integral to this diversity. Our data (Arthur and others 1998) as well as other studies (Barnes and Van Lear 1998, Thor and Nichols 1974, Van Lear and Waldrop 1989) suggest that multiple fires will be necessary to realize this objective. This study suggests that low intensity single fires where little, if any, organic matter is lost have little effect on soil nutrient status. Future research should examine whether repeated fire on these sites has similar effects on soil nutrients and microbial biomass or whether impacts of repeated burning on soil resources further degrade already nutrient-poor sites.

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REFERENCES

Agee, J.K. 1993. Fire ecology of Pacific Northwest forests. Washington, DC: Island Press. 493 p.

Ahlgren, I.F. 1974. The effect of fire on soil organisms. In: Kozlowski, T.T.; Ahlgren, C.E., eds. Fire and ecosystems. New York: Academic Press: 47-71.

Ahlgren, I.F.; Ahlgren, C.E. 1960. Ecological effects of forest fires. Botanical Review. 26: 483-533.

Ahlgren, I.F.; Ahlgren C.E. 1965. Effects of prescribed burning on soil microorganisms in a Minnesota jack pine forest. Ecology. 46: 304-310.

Arends, E.; McCormick, J.F. 1987. Replacement of oak-chestnut forests in the Great Smoky Mountains. In: Hay, R.L.; Woods, F.W.; DeSelm, H., eds. Proceedings of the central hardwood forest conference VI; 1987 February 24-26; Knoxville, TN. Knoxville: University of Tennessee: 305-315.

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

Austin, R.C.; Baisinger, D.H. 1955. Some effects of burning on forest soils of western Oregon and Washington. Journal of Forestry. 53: 275-80.

Avers, P.E.; Austin, J.S.; Long, J.K. [and others]. 1974. Soil survey of Menifee and Rowan Counties and northwestern Morgan County, Kentucky. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 88 p.

Barnes, T.A.; Van Lear, D.H. 1998. Prescribed fire effects on advanced regeneration in mixed hardwood stands. Southern Journal of Applied Forestry. 22: 138-142.

Blankenship, B.A.; Arthur, M.A. [In press]. Prescribed fire affects eastern white pine recruitment and survival on eastern Kentucky ridgetops. Southern Journal of Applied Forestry.

Borchers, J.G.; Perry, D.A. 1990. Effects of prescribed fire on soil organisms. In: Walstad, J.D.; Radosevich, S.R.; Sandberg, D.V., eds. Natural and prescribed fire in the Pacific Northwest. Corvallis, OR: Oregon State University: 143-157.

Braun, E.L. 1950. Deciduous forests of Eastern North America. New York: Hafner Press. 536 p.

Brose, P.H.; Van Lear, D.H. 1998. Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Canadian Journal of Forest Research. 28: 331-339.

Brown, M.B.; Forsythe, A.B. 1974. Robust tests for the equality of variances. Journal of the American Statistical Association. 69: 364-367.

Clinton, B.D.; Vose, J.M.; Swank, W.T. 1996. Shifts in aboveground and forest floor carbon and nitrogen pools after felling and burning in the Southern Appalachians. Forest Science. 42: 431-441.

Cole, K.L.; Klick, K.F.; Pavlovic, N.B. 1992. Fire temperature monitoring during experimental burns at Indiana Dunes National Lakeshore. Natural Areas Journal. 12: 177-183.

Delcourt, P.A.; Delcourt, H.R. 1997. Report of paleoecological investigations. Cliff Palace Pond, Jackson County, Kentucky, in the Daniel Boone National Forest. Forest Service Contract 43-531A-6-0389. 61 p.

Environmental Quality Commission. 1992. State of Kentucky's environment: a report of progress and problems. Frankfort, KY: Commonwealth of Kentucky.

Fritze, H.; Pennanen, T.; Pietikäinen, J. 1992. Recovery of soil microbial biomass and ativity from prescribed burning. Canadian Journal of Forest Research. 23: 1286-1290.

Gilbert, N.L.; Arthur, M.A.; Johnson, S.L. [and others]. 1998. Prescribed fire effects on foliar nutrients, photosynthesis, and growth of ridgetop maple and oak seedlings [Poster presentation]. In: Annual meetings of the Ecological Society of America; 1998 August 1-6; Baltimore, MD.

Gilliam, F.S. 1991. The significance of fire in an oligotrophic forest ecosystem. In: Nodvin, S.C.; Waldrop, T.A., eds. Fire and the environment: ecological and cultural perspectives: Proceedings of an international symposium; 1990 March 20-24; Knoxville, TN. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 113-122.

Hayes, R.A. 1993. Soil survey of Powell and Wolfe Counties, Kentucky. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 173 p.

Hendershot, W.H.; Lalande, H. 1993. Soil reaction and exhangeable acidity. In: Carter, M.R., ed. Soil sampling and methods of analysis. Ann Arbor: Lewis Publishers: 141-145.

Hill, J.D. 1976. Climate of Kentucky. Progress Rep. 221. Lexington, KY: University of Kentucky, College of Agriculture, Agricultural Experiment, Department of Agronomy. 88 p.

Hungerford, R.D.; Harrington, M.G.; Frandsen, W.H. [and others]. 1990. Influence of fire on factors that affect site productivity. In: Harvey, A.E.; Neuwnschwander, L.F., comps. Proceedings, management and productivity of western-montane forest soils; 1990 April 10-12; Boise, ID. Gen. Tech. Rep. INT-280. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 32-50.

Hurlburt, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs. 54: 187-211.

Ingham, E.R.; Griffiths, R.P.; Cromack, K.; Entry, J.A. 1991. Comparison of direct vs. fumigation incubation microbial biomass estimates from ectomycorrhizal mat and non-mat soils. Soil Biology and Biochemistry. 23: 465-471.

Ingham, E.R.; Klein, D.A. 1984. Soil fungi: Relationships between hyphal activity and staining with fluorescein diacetate. Soil Biology and Biochemistry. 16: 273-278.

Jurgensen, M.F.; Harvey, A.E.; Larsen, M.J. 1981. Effects of prescribed fire on soil nitrogen levels in a cutover Douglas-fir/western larch forest. Res. Pap. INT-275. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Research Station. 6 p.

Knoepp, J.D.; Swank, W.T. 1993. Site preparation burning to improve Southern Appalachian pine-hardwood stands: nitrogen responses in soil, soil water, and streams. Canadian Journal of Forest Research. 23: 2263-2270.

Levene, H. 1960. Robust tests for the equality of variances. In: Olkin, I., ed. Contributions to probability and statistics. Stanford: Stanford University Press. 517 p.

Lodge, D.J.; Ingham, E.R. 1991. A comparison of agar film techniques for estimating fungal biovolumes in litter soil. Agriculture, Ecosystems and Environment. 5: 131-144.

Lorimer, C.G. 1993. Causes of the oak regeneration problem. In: Loftis, D.; McGee, C.E., eds. Oak regeneration: serious problems, practical recommendations, symposium proceedings; 1992 September 8-10; Knoxville, TN. Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 14-39.

Martin, W.H. 1989. The role and history of fire in the Daniel Boone National Forest. Winchester, KY: U.S. Department of Agriculture, Forest Service. 131 p.

Maynard, D.G.; Kalra, Y.P. 1993. Nitrate and exchangeable ammonium nitrogen. In: Carter, M.R., ed. Soil sampling and methods of analysis. Ann Arbor: Lewis Publishers: 25-38.

Nyland, R.D.; Abrahamson, L.P.; Adams, K.B. 1983. Use of prescribed fire for regenerating white and red oak in New York. In: America's hardwood forests - opportunities unlimited. Proceedings of the 1992 convention of the Society of American Foresters; 1982 September 19-22; Cincinnati, OH. Bethesda, MD: Society of American Foresters: 163-167.

Pietikäinen, J; Fritze, H. 1993. Microbial biomass and activity in the humus layer following burning: short-term effects of two different fires. Canadian Journal of Forest Research. 23: 1275-1285.

Prieto-Fernandez, A.; Villar, M.C.; Carballas, M.; Carballas, T. 1993. Short-term effects of a wildfire on the nitrogen status and its mineralization kinetics in an Atlantic forest soil. Soil Biology and Biochemistry. 25: 1657-1664.

Pyne, S.J. 1982. Fire in America: a cultural history of wildland and rural fire. Princeton: Princeton University Press. 654 p.

Pyne, S.J.; Andrews, P.L.; Laven, R.D. 1996. Introduction to wildland fire. 2d ed. New York: John Wiley. 769 p.

Reich, P.B.; Abrams, M.D.; Ellsworth, D.S. [and others]. 1990. Fire affects ecophysiology and community dynamics of central Wisconsin oak forest regeneration. Ecology. 71: 2179-2190.

Richardson, D.M. 1995. Prescribed fire plan for Klaber Ridge, Pinch-Em-Tight Ridge, and Whittleton Ridge, Daniel Boone National Forest, Kentucky. R8-FS-5100-6. U.S. Department of Agriculture, Forest Service.

Richardson, D.M. 1996. Prescribed fire plan for Klaber Ridge and Pinch-Em-Tight Ridge, Daniel Boone National Forest, Kentucky. R8-FS-5100-6. U.S. Department of Agriculture, Forest Service.

SAS Institute Inc. 1995. JMP® User's guide. Cary, NC: SAS Institute, Inc. 593 p.

Soil Conservation Service. 1975. Soil taxonomy: a basic system of soil classification for making and interpreting soil surveys. Agric. Handb. 436. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 754 p.

Thor, E.; Nichols, G.M. 1974. Some effects of fire on litter, soil and hardwood regeneration. In: Proceedings of Tall Timbers Fire Ecology Conference 13; 1973 March 22-23; Tallahassee: Tall Timbers Research Station: 317-329.

Tran, T.S.; Simard, R.R. 1993. Mehlich III extractable elements. In: Carter, M.R., ed. Soil sampling and methods of analysis. Ann Arbor: Lewis Publishers: 43-39.

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

Vose, J.M.; Swank, W.T. 1993. Site preparation burning to improve Southern Appalachian pine-hardwood stands: aboveground biomass, forest floor mass, and nitrogen an carbon pools. Canadian Journal of Forest Research. 23: 2255-2262.

Vose, J.M.; Swank, W.T.; Clinton, B.D. [and others]. [In press]. Using stand replacement fires to restore Southern Appalachian pine-hardwood ecosystems: effects on mass, carbon, and nutrient pools. Forest Ecology and Management.

Wehner, R.R. 1991. The status of white pine (*Pinus strobus*) in the Clifty Wilderness, Kentucky. Lexington, KY: University of Kentucky. 67 p. M.S. thesis.

Weir, G.W.; Richards, P.W. 1974. Geologic map of the Pomeroyton quadrangle, east-central Kentucky. Reston, VA: U.S. Geological Survey.

Welch, B.L. 1951. On the comparison of several mean values: an alternative approach. Biometrika. 38: 330-336.

Stand Structure

COMPARISON OF ECOLOGICAL CHARACTERISTICS OF THREE REMNANT OLD-GROWTH WOODLOTS IN BELMONT COUNTY, OHIO

Ray R. Hicks, Jr. and Jeffrey Holt[1]

Abstract—Dysart Woods, North and South, and Collins Woods are small remnant old-growth stands with an old-growth cohort that has a significant oak component (> 50 percent) and includes mixed mesophytic species, such as yellow-poplar and white ash. All three stands are transitional to a northern hardwood type (sugar maple/American beech). Collins Woods is bounded on two sides by a surface mine highwall (20-60 feet high), and on another side by a reclaimed surface mine. Dysart Woods's stands are surrounded by second-growth hardwood forest and agricultural fields. We sampled trees larger than 2 inches dbh and measured woody regeneration. In addition, we conducted a 100 percent inventory of all large trees in the old-growth cohort. We found all three stands to be remarkably similar. For example, the basal area of trees at Collins Woods was 123.3 feet square/acre as compared to 101.2 and 102.5 for the South and North Woods at Dysart Woods. The oak component of the old-growth cohort at Collins Woods was 50.3 percent. For Dysart South and North, oak composition was 61.7 percent and 60.7 percent. Yellow-poplar made up 10.3 percent and 9.8 percent of Dysart South and North and 21 percent of the old-growth cohort at Collins Woods. The average dbh of old-growth trees at Collins Woods was 34.4 inches compared to 33.9 inches and 35.1 inches for the Dysart stands. Collins Woods, like Dysart Woods, had virtually no oak regeneration and the overall diameter distribution indicated a reverse J-shaped curve for all three stands with sugar maple predominating in the smaller diameter classes, and oaks and yellow-poplar in the larger classes. The species diversity of Collins Woods was slightly less than that at Dysart Woods, when expressed as Shannon's H', but about the same in species richness. Measures of health and vigor of the stands were very comparable. The estimated overall mortality rate for old-growth trees at Collins Woods was 1.49 percent/year as compared to 1.56 percent and 1.85 percent at the two stands at Dysart. The tree vigor rating, using the USDA, Forest Service vigor codes, (1 as vigorous and 7 as dead), averaged 2.05, 1.94 and 2.37 for Dysart South and North and Collins Woods, respectively.

The degree of similarity among the three stands suggests that they may have been part of a large contiguous forest. In spite of their different histories, the successional changes in these three stands have produced remarkably similar results. These stands represent what many present-day central hardwood forests could look like in about 200 years, if left relatively undisturbed.

INTRODUCTION

Old-growth central hardwood forests are interesting and valuable both from a natural history standpoint as well as examples of the successional process that affects them. The dynamics of these remnant old-growth stands provide clues as to the potential fate of many present-day stands that, like most old-growth remnants, developed from disturbances such as fire. The present study documents the current condition of three remnant old-growth stands in east-central Ohio. The composition of the old-growth cohort (the largest and oldest trees) is suggestive of a disturbance origin, but the gradual replacement of these stands by shade tolerant climax species is an on-going process that is documented in this study. These case studies, in addition to illustrating the process of succession, provide insight into the timing of changes in undisturbed forests.

DESCRIPTION OF THE SITE

The stands at Dysart Woods are old-growth remnants located approximately 5 miles south of St. Clairsville, Ohio, off state Route 9 and are spatially separated by about 1000 feet. The stands at Dysart Woods are surrounded by second-growth hardwoods except for one side of the North Woods, which is an agricultural meadow. Collins Woods is about 13 miles northwest of Dysart Woods, in Belmont County, Ohio, about 3 miles northwest of Morristown. This woodlot is surrounded on 3 sides by former surface mining activity and has highwalls of approximately 60 feet high or higher on the north and east borders. Mining operations that created these highwalls were conducted in the early to mid 1970s. All three woodlots are remnant old-growth stands that are estimated to be in the age range of approximately 300 years. Collins Woods has been purchased by The Nature Conservancy and is preserved as an example of what was most likely an extensive forest cover type about the time of European settlement of eastern Ohio while Dysart Woods is managed by Ohio University. Stands like Collins and Dysart Woods probably owe their origin to disturbance by fire which favors the regeneration of oaks and shade-intolerant species like yellow-poplar (*Liriodendron tulipifera*).

METHODS

The extent of the old-growth forests in the three stands was determined as the area inside the drip line of the area containing large old trees. Dysart North was found to be approximately 15 acres and the South Woods is about 13.5 acres in size while Collins Woods was 8.3 acres.

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A sampling grid was established over each stand in order to locate approximately 16 plots per stand. At each sample location a 0.1-acre circular plot was established. At plot center, a 10 BAF prism was used to estimate the basal area. The plot was classified as to the Society of American Foresters' cover type present (Eyre 1980). In addition, the slope position, aspect and slope percent were determined. All trees 2 inches dbh and larger were measured in the 0.1-acre plots. For each tree, its species, crown class, live crown ratio, USFS vigor rating[2], expected longevity and total height was evaluated and recorded.

In conjunction with the 0.1-acre plots, two 6-foot-radius understory plots were established, one at plot center and another midway between plot center and the perimeter of the 0.1-acre plot on a random azimuth. The percent of the regeneration plot covered by all vegetation (broken down by percent coverage of woody, herbaceous and fern) was estimated for each 6-foot-radius plot. In addition, all tree seedlings (< 2 inches dbh) in the plots were counted by species and size class (< 6 inches, 6 inches-4 feet, > 4 feet).

Finally all large trees in the "old-growth cohort" (27.5 inches dbh and larger) in each stand were measured (100 percent sample). This size limit was selected subjectively because it seemed to include all the large old trees (particularly oaks and yellow-poplar). Each tree was numbered and dbh, total height, live crown ratio, crown radius, vigor class and expected longevity (estimated as < 10 years, 10-20 years or > 20 years) was recorded. In addition to the large live trees, large standing dead trees were also measured by species for dbh and estimated years dead (based on bark and limb condition). All the large trees were located with laser survey equipment to determine their precise location within the stand so that they could be mapped and plotted.

Tree data from 0.1-acre plots and 100 percent samples were combined to produce the diameter distribution by species (number of trees by diameter). For the large trees (> 27.5 inches dbh), only the 100 percent sample data were used.

Seedling data were compiled by species to document the numbers of seedlings by species and size class, which should indicate the regeneration potential of the various species.

Data from the various samples were used to compute importance values of the tree species. Importance value is the sum of relative density, relative frequency and relative dominance. These are obtained as the number of stems of a species relative to the total number of stems, the number of plots containing the species relative to the total number of plots and the basal area of a species relative to the total basal area. This information was compared to importance values reported in previous studies of Dysart Woods.

[2] Vigor codes: 1-healthy/vigorous, 2-slight dieback, 3-moderate dieback, 4-slight decline, 6-moderate decline, 6-severe decline, 7-dead.

We used data for total number of species present and total individuals per species to develop measures of diversity in each stand (species richness and Shannon-Weiner index). These measures could be used to compare the diversity of the three stands and to compare them with other forests.

Finally, the data from standing dead trees and estimated longevity of large live trees were used to estimate the rate of mortality (percent per decade). This was done by averaging the class midpoints for estimated longevity for old-growth trees (e.g., 0-10 years = 5 years) and doing the same for estimated years dead for large standing dead trees. Thus, two estimates of rate of mortality were obtained (one based on expected longevity of live old-growth trees and the other based on standing dead trees). These two estimates were then averaged to obtain the value that was presented in subsequent discussions.

RESULTS

Overall

Collins Woods was very similar to the stands at Dysart Woods (Table 1) with an old-growth cohort of oaks and mesophytic hardwoods that is being gradually replaced by northern hardwoods [American beech (*Fagus grandifolia*)/ sugar maple (*Acer saccharum*)]. There were a few large stumps at Collins Woods that indicated limited selective logging had occurred in the past, similar to Dysart Woods, but in general most of the old-growth trees were left. The most unique feature of Collins Woods was the fact that the area had been circumscribed by contour surface mining, and along one side (approximately 800 feet), the highwall remained intact while on another side reclamation by back-filling, recontouring and seeding had been done.

The forested site at Collins Woods was an upper slope, primarily on a west-facing aspect with an average slope inclination of about 18 percent. West-facing aspects and upper slope positions, as occurred at Collins Woods, are typically associated with poorer growing sites (Hicks and Frank 1984). Dysart woods had a wider variety of aspects and slope positions, some of which are associated with better quality sites. The diameter distribution of trees at Collins Woods was a reverse J-shaped distribution, similar to the Dysart Woods stands (Figs. 1-3).

Regeneration

Like Dysart Woods, Collins Woods showed little or no oak regeneration—no oak seedlings were found in our survey. As with the Dysart stands, Collins Woods had abundant sugar maple, beech and elm (*Ulmus* spp.) regeneration (Figs. 4, 5). Unlike the Dysart stands, elms at Collins Woods have not grown into the small sapling-size class. Perhaps the higher overstory density at Collins Woods, as reflected by higher basal area, is responsible for shading out the elms before they reach sapling size. A variety of other woody regeneration occurred at Collins Woods, including bitternut hickory (*Carya cordiformis*), hophornbeam (*Ostrya virginiana*) and spicebush (*Lindera benzoin*). Seedling numbers at Collins Woods resembled more closely the North Woods at Dysart Woods, although Collins Woods had fewer maples and beech seedlings than

Table 1—Summary data for Dysart (North and South) and Collins Woods

Variable	Dysart Woods		
	South Woods	North Woods	Collins Woods
Basal area			
(live)(sq. ft./ac.)	101.25	102.50	123.30
SAF cover (pct. of plots)			
Type 54[a]	87.50	81.25	27.80
Type 60	0	12.50	55.50
Type 27	12.50	6.25	16.70
Slope percent (average)	16.75	16.40	17.80
Aspect class (pct. of plots)			
1 (N)	0	25.00	0
2 (NE)	0	0	0
3 (E)	31.25	0	0
4 (SE)	6.25	6.25	0
5 (S)	12.50	25.00	0
6 (SW)	37.50	.625	11.11
7 (W)	12.50	0	83.33
8 (NW)	0	37.50	5.55
Slope position (pct. of plots)			
Ridge	0	6.25	0
Upper slope	31.25	43.75	38.90
Lower slope	37.50	37.50	61.10
Hollow/bottom	31.25	12.50	0

[a] SAF cover type: 54=yellow-poplar; 60=white oak; 27=northern red oak.

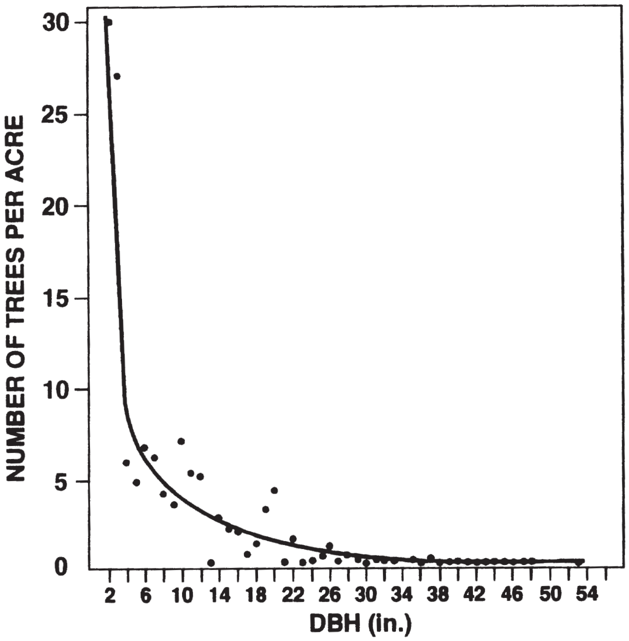

Figure 1—Diameter distribution of all trees > 2 inches dbh, South Woods, Dysart Woods.

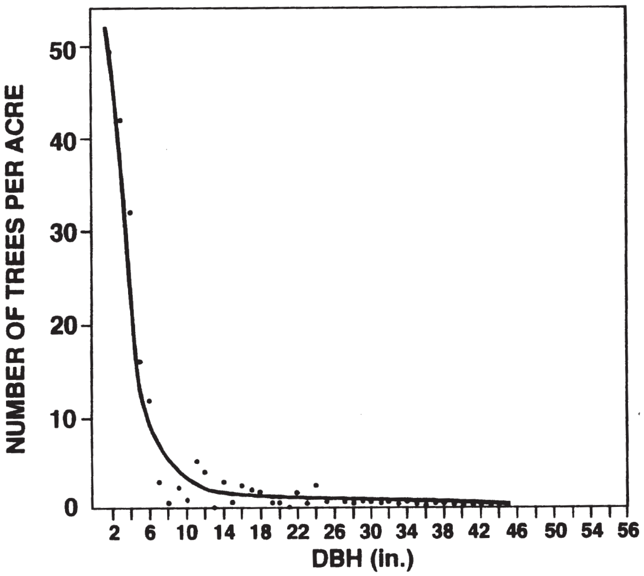

Figure 2—Diameter distribution of all trees > 2 inches dbh, North Woods, Dysart Woods.

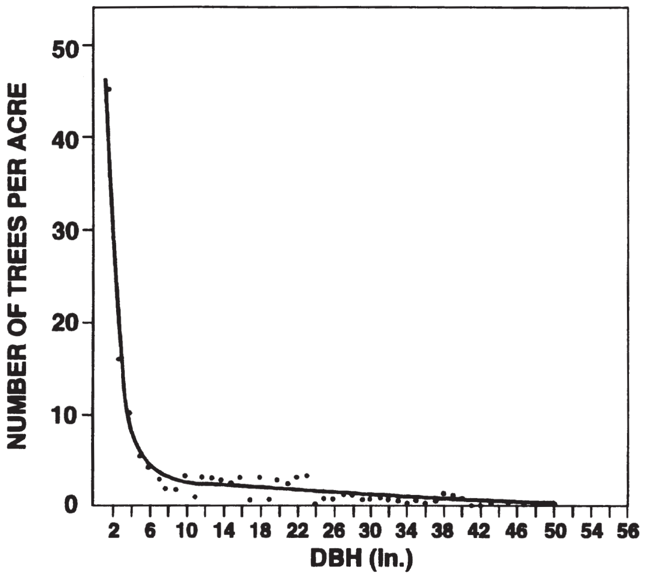

Figure 3—Diameter distribution of all trees > 2 inches dbh, Collins Woods.

either of the Dysart stands. This may reflect the effect of higher basal area and greater shade at Collins Woods or possibly other factors such as a higher deer density.

Overstory

Almost 70 percent of the overstory plots at Collins Woods were classed as northern hardwoods types (beech/maple and sugar maple), whereas the majority of plots at Dysart Woods were classed as mesophytic hardwoods. The large number of beech and sugar maple at Collins Woods that are in the diameter range of 10 to 27 inches is the basis for classifying the Collins Woods plots as northern hardwoods. Of the approximately 35 trees per acre in the 10-27-inch diameter range, 33 of them were beech or maples.

The importance values of species at Collins Woods (relative density + relative dominance + relative abundance) were very similar to those of Dysart stands. For example, sugar maple had an importance value at Collins Woods of 187.5 (Table 2). Its importance values for the South Woods and North Woods at Dysart Woods were 176.6 and 162.9, respectively. The ranking of importance value at Collins Woods was sugar maple, American beech,

Table 2—Relative densities (RD), relative frequencies (RF), relative dominance (RDo) and importance values (IV) for tree species sampled at Dysart and Collins Woods

	Dysart Woods														
	South Woods					North Woods					Collins Woods				
Species	RD	RF	RDo	IV	Rank	RD	RF	RDo	IV	Rank	RD	RF	RDo	IV	Rank
Sugar maple	46.95	100	29.63	176.58	1	55.04	90	17.89	162.93	1	69	100	18.5	187.5	1
American beech	14.24	70	23.04	107.28	2	13.51	80	29.27	122.78	2	15	67	15.1	85.6	2
Elms	17.37	50	3.29	70.66	3	6.65	40	3.25	49.90	4	2	22	3.6	27.6	5
Black cherry	6.37	40	0	46.51	4	4.11	40	0	44.11	6	1	11	0.1	12.1	13
White oak	3.06	20	19.75	42.81	5	1.72	40	26.02	67.74	3	1	17	15.6	33.6	4
White ash	3.24	30	1.65	34.89	6	1.61	20	3.25	24.86	9	1	11	3.5	15.5	10
Blackgum	2.24	20	4.94	27.18	7	1.87	20	1.63	23.50	10	1	11	0.4	12.4	12
Yellow-poplar	1.20	20	3.29	24.49	8	2.08	10	4.88	16.96	13	3	17	2.5	22.5	7
Northern red oak	1.78	10	1.64	13.42	9	1.61	20	4.88	26.49	8	2	39	25.0	66.0	3
Shagbark/shell Bark/bitternut Hickory	1.16	10	1.65	12.81	10	1.51	20	1.63	23.14	11	2	11	1.3	14.3	11
Flowering dogwood	0.72	10	0	10.72	11	4.26	30	0	34.26	7	1	6	0.1	7.1	15
Red maple	1.27	0	6.58	7.85	12	1.72	40	3.25	44.97	5	2	17	3.3	22.3	8
Black walnut	<0.01	0	0	<0.01	13	—-	—	—-	—-		—	—-	—-	—-	
Blue beech/ Hophornbeam	—-	—-	—-	—-		3.12	20	0	23.12	12	1	22	0.2	23.2	6
Serviceberry	—-	—-	—-	—-		0.32	10	0	10.32	15	—	—-	—-	—-	
Sourwood	—-	—-	—-	—-		—-	—	—-	—-		—	—-	—-	—-	
Hackberry	—-	—-	—-	—-		0.32	10	0	10.32	15	—	—-	—-	—-	
Basswood	—-	—-	—-	—-		0.52	10	0	10.52	14	—	—-	—-	—-	
Black oak	—-	—-	—-	—-		—-	—	—-	—-		2	11	6.0	19.0	9
Sassafras	—-	—-	—-	—-		—-	—	—-	—-		1	6	1.1	8.1	14

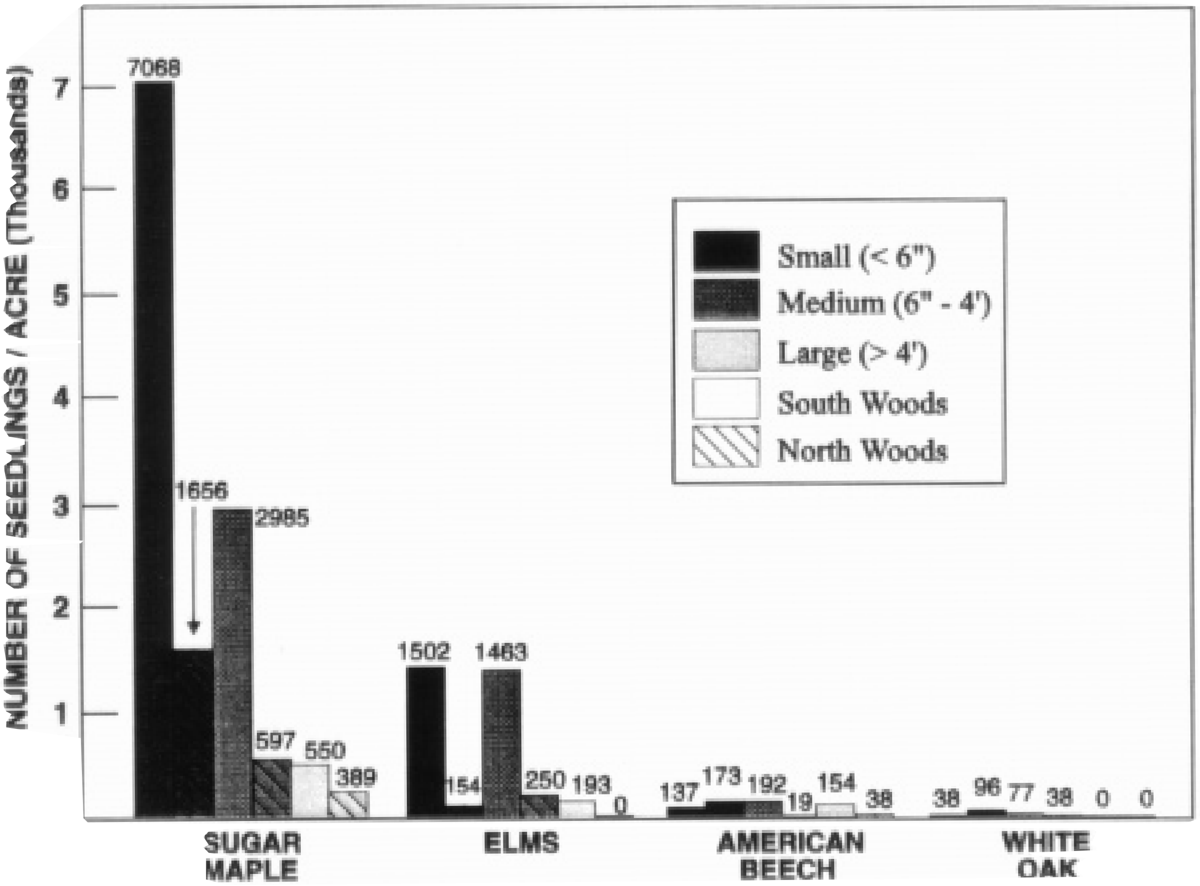

Figure 4—Number of seedlings per acre by species and height class for the North and South Woods, Dysart Woods.

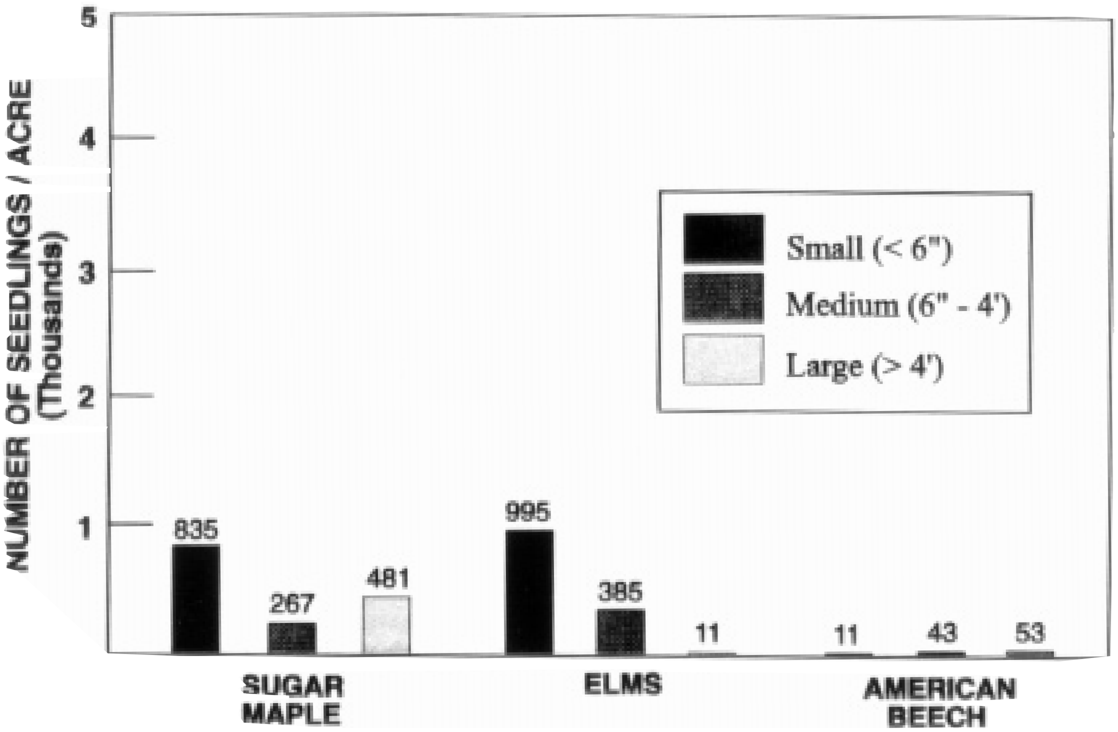

Figure 5—Number of seedlings per acre by species and height class for Collins Woods.

northern red oak (*Quercus rubra*), white oak (*Quercus alba*) and elm, whereas 4 of the top 5 species at Dysart North and South were the same, and sugar maple and American beech were the first- and second-ranked species at all these stands.

Previous studies performed at Dysart Woods (Benner 1971, Lafer 1968, Smith 1979) reported importance values. Figures 6 and 7 show the change over time of the relative ranking of species by importance values. In general, sugar maples and beech were ranked 1 and 2, while oak declined and elms increased. The species diversity at Collins Woods, as measured by Shannon's H' was lower than that of Dysart Woods (Table 3). But diversity based on species richness was very similar among the three stands.

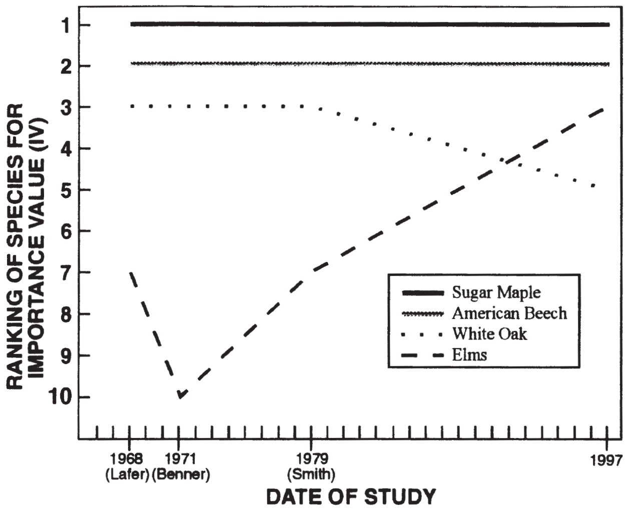

Figure 6—Relative ranking of species by importance values over 31 years for the South Woods, Dysart Woods.

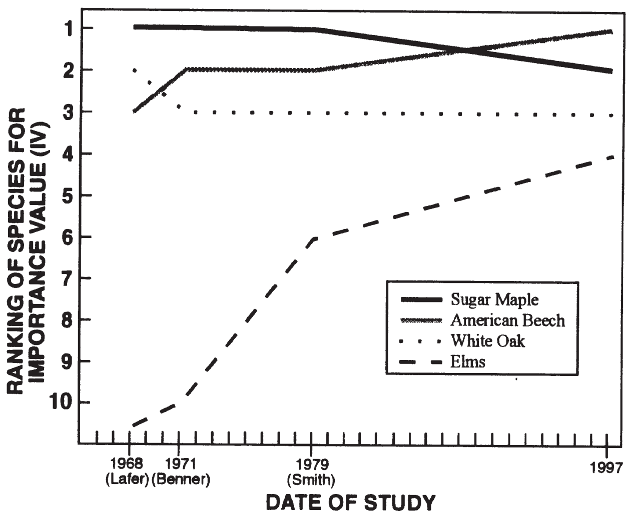

Figure 7—Relative ranking of species by importance values over 31 years for the North Woods, Dysart Woods.

Old-Growth

There was an average of 9.7 large trees per acre at Collins Woods as compared to 7.9 and 8.0 at Dysart South and North (Table 4). The average dbh of old-growth trees was very similar among all three stands but trees at Dysart Woods were, on average, about 15-20 feet taller, probably indicating an inherently better overall growing site at Dysart Woods. This difference in height is not related to any effect of mining over the past 30 years since old-growth trees, even on the best of sites, grow very little in height. For example, growth curves for yellow-poplar presented by Carmean and others (1989) indicate that trees on average sites grow about two feet per year. By age 100, yellow-poplar trees on these sites typically grow less than one foot per year, and by 250-300 years of age, trees have virtually ceased height growth due to the natural slowing with age. Thus the greater height at Dysart Woods most likely resulted from a steeper growth curve during the first 100 years of life rather than any occurrences in recent years. Collins Woods had a higher proportion of northern red oak and yellow-poplar and a lower proportion of white oak.

Comparing measures of forest health and vigor for the three stands, we found that they were very comparable. The overall rate of mortality (percent per year) for the large trees at Collins Woods was estimated to be 1.49 percent, whereas Dysart South and North were estimated to be 1.85 percent and 1.56 percent, respectively. The USDA, Forest Service vigor code averaged 2.37 (1 being most vigorous and 7 being dead) for Collins Woods (Table 5). The two stands at Dysart Woods averaged 1.94 and 2.05.

DISCUSSION AND CONCLUSIONS

The old-growth component of the forest at Collins and Dysart Woods regenerated in the late 17th or early 18th century. Many authors have linked oak regeneration to fire (Abrams 1992). It is acknowledged by a number of writers (Curtis 1959, Pallardy and others 1988) that presettlement fires, either caused by lightning or set by native people, were an important disturbance that led to the regeneration of oaks over vast areas. Native American populations were decimated by disease that was introduced by early European explorers which led to a lower incidence of fire. In the absence of fire, oaks regenerate poorly, mostly because they are unable to compete in the understory of established stands (Norwacki and others 1990).

Dysart and Collins Woods are old-growth "islands" that have persisted in spite of landscape changes, such as reduced fire, land clearing, subsistence farming and logging, that occurred throughout the central hardwood region.

The diameter distribution and structure of the stands gives important clues as to the origin of the various cohorts present. For example, the white oaks have a diameter distribution typical of an even-age stand in spite of the fact that the aggregate distribution of all species has the reverse J-shaped distribution typical of all-age stands. Stout (1991) observed a similar phenomenon on a much wider scale that characterizes so-called "transition" stands, currently covering a large area in the Allegheny Plateau of

Table 3—Species diversity [species richness and Shannon-Weiner Index (H')] for Collins Woods, Dysart Woods and other old-growth central hardwood forests

Forest	Shannon-Weiner H' $(H'=-1\sum_{i=1}^{n} pi \ln pi)$	Species richness No./0.1 ac.	Species richness Total no.	Reference
Collins Woods	1.25	—-	15	Present study
Dysart, South	1.69	4.0	13	Present study
Dysart, North	1.72	5.8	18	DeSelm and Sherman (1982)
Savage Gulf (TN)	3.70	—-	22	Monk (1967)
Beech-white oak	2.77	—-	—	Bryant (1987)
Ault Park (OH)	2.83	—-	21	Bryant (1987)
California Woods (OH)	3.11	—-	20	Bryant (1987)
Caldwell Park (OH)	3.35	—-	21	Bryant (1987)
Winton Woods (OH)	2.18	—-	15	Bryant (1987)
Bowles Woods (OH)	1.94	—-	15	Bryant (1987)
Melborne Forest (OH)	3.07	—-	17	Bryant (1987)
Davis-Purdue (IN)	2.50	—-	30	Ward and Parker (1987)

Table 4—Summary data for attributes of live large trees (> 27.5 inches d.b.h.) for Dysart Woods (South and North) and Collins Woods

	Dysart Woods[a]											
	South Woods				North Woods				Collins Woods[a]			
Species	Total no. trees	RD[b]	D.b.h.	Ht.	Total no. trees	RD[b]	D.b.h.	Ht.	Total no. trees	RD[b]	D.b.h.	Ht.
			In.	Ft.			In.	Ft.			In.	Ft.
White oak	55	51.4	35.3	120.9	59	48.4	37.5	121.4	18	22.2	36.9	100.8
Northern red oak	11	10.3	36.2	123.5	13	10.6	38.1	122.6	20	24.8	37.3	108.6
Black oak	0	0	—	—	2	1.7	29.0	117.0	3	3.7	31.7	102.0
Sugar maple	8	7.5	29.8	121.3	3	2.5	32.3	116.7	7	8.6	29.6	104.6
Red maple	1	0.9	32.0	121.3	1	0.8	32.0	88.0	0	0	—	—
American beech	18	16.8	30.9	101.0	17	13.9	30.0	106.0	12	14.8	29.9	98.9
Yellow-poplar	11	10.3	31.2	115.0	12	9.8	34.2	134.7	17	21.0	34.7	110.7
White ash	3	2.8	34.7	136.4	12	9.8	31.9	123.1	1	1.2	29.0	110.0
Blackgum	0	0	—	—	1	0.8	28.0	120.0	0	0	—	—
Shagbark hickory	0	0	—	—	2	1.7	30.5	128.0	0	0	—	—
Elm	0	0	—	—	0	0	—	—	3	3.7	32.7	108.3
Total/overall	107[c]	100	33.9	122.3	122[c]	100	35.1	120.5	81[c]	100	34.4	105.1

[a] All figures are averages.
[b] Relative density (percent).
[c] Number of large trees per acre—South Woods @ 13.43 ac. = 7.97 large trees/ac., North Woods @ 15.18 ac. = 8.04 large trees/ac., and Collins Woods @ 8.36 ac. = 9.69 large trees/ac.

Table 5—Health/vigor indices for Dysart and Collins Woods

Health/vigor indices	Dysart, North	Dysart, South	Collins Woods
Estimated tree mortality[a] (pct./yr.)	1.56	1.85	1.49
Average[b] USFS vigor rating	2.05	1.94	2.37

[a] Based on all trees in the old-growth cohort.
[b] Vigor assessed on a scale of 1-7, where 1=healthy and 7=dead.

Pennsylvania. These stands contain representatives of oak-hickory as well as northern hardwood types, and the oak component is generally in the larger diameter classes and possesses a normal or bell-shaped distribution.

The diameter distribution of the American beech component of the stands in Dysart Woods provides some interesting questions. In both stands, beech displays a reverse J-shaped distribution, typical of shade-tolerant species, but with a wave of larger diameter trees. This suggests that a cohort of beech ingrowth responded to a disturbance that produced fairly substantial canopy gaps about 75-150 years ago. Disturbances, such as drought, insect or disease attacks, ice or wind damage, could trigger such a phenomenon. Beech is representative of so-called "role-2" species (Shugart 1987) and is shade tolerant enough to regenerate in dense shade (Harcombe and others 1982) and capable of responding to canopy gaps as they develop.

The diameter distribution of sugar maple in all stands was typical of the reverse J-shaped distribution expected of shade-tolerant species in an all-age stand. However, the larger number of overtopped saplings in the Dysart North Woods compared to the South Woods seems to suggest that the regeneration of sugar maple in the two stands has become dissynchronous, and this relates to some fairly recent disturbance events that are not common to both stands. A ground fire would be a likely explanation but no charcoal or fire scars on trees was evident to point to such an occurrence and no reports of fires were found in published accounts of the history of Dysart Woods. An alternative explanation might be a different woodland grazing regime for the two stands.

The regeneration inventory of the stands reinforces our conclusions regarding the occurrence of a gradual shift from oaks to a maple-beech forest. This is a process that is on-going throughout the central hardwood region. Loftis (1989) indicates that the process is best described by combining the concepts of "initial floristics composition" and "vital ecological attributes" of species. That is to say that propagules of all the species present today were present when the stands first developed, but the shifts in dominance over time reflect a change in conditions that favor some species over others. Currently, the balance has shifted toward sugar maple and American beech and away from oaks and yellow-poplar. This transition from oaks to maples and other northern or mesophytic hardwoods is a trend that is well documented in old-growth forests throughout the central hardwood region (Downs and Abrams 1991, Schlesinger 1989). The interesting phenomenon relating to greater overall abundance of regeneration (especially sugar maple) in the Dysart South Woods may be part of the same process that has caused the reverse to be true of trees in the sapling size class (2 inches-6 inches dbh). In-other-words, the dense low shade created by the cohort of maples and beech in the sapling size class may be preventing regeneration, even of shade-tolerant species, in the North Woods. The high shade from canopy-level trees in the South Woods has allowed better seedling establishment there. Elms are apparently producing more seedlings and saplings now than were reported in past studies of Dysart Woods, but due to their intermediate status in shade tolerance and their susceptibility to disease, they will probably not become significant contributors to any future overstory.

The remarkable similarity between the old-growth components of the three stands indicates that strong forces are at work driving the ecosystem changes that have taken place. These stands were probably historically part of a single large stand but even so, in spite of differences in average aspect and soils (Lafer 1968) the resulting old-growth cohorts have maintained a high degree of similarity. The dissynchrony in regeneration between the two Dysart stands, however, indicates that it is the intrinsic characteristics of the system rather than particular disturbance events that drive the process.

Regarding species diversity, Dysart and Collins Woods is on the low end of the scale for old-growth forests with values for the Shannon-Weiner index of 1.69 and 1.72. There are two reasons for the mathematically low values. First, these stands have relatively low woody species richness compared to other old-growth forests reported in the central hardwood region (Table 3) and secondly, as Loucks (1970) proposes, northern hardwood types become dominated by sugar maple which reduces their diversity.

The process of oak replacement by maple and beech can be illustrated by plotting the change in ranking of importance values of species over the past 30 years (Figs. 6 and 7). As can be seen on these two figures, white oak is generally declining in rank whereas sugar maple is either increasing or maintaining. In both stands, sugar maple is now the top ranked species and American beech is in second position. An interesting rise in the importance value ranking of elms is primarily related to a large influx of sapling-sized trees—many of which are presently showing symptoms of declining vigor.

The future of these stands, barring a major disturbance event, is likely to continue on a course where large senescent oaks and yellow-poplars are replaced by sugar maple and American beech trees presently in subordinate canopy positions. Stresses, such as drought, air pollution/acid deposition, insects [gypsy moth (*Lymantria* *dispar*), etc.] and diseases (beech bark disease, Dutch elm disease, dogwood anthracnose, etc.) will play a role in the future of Collins and Dysart Woods, and these factors, although unpredictable, will probably alter the course of events by affecting the balance in competitive advantage among species.

Perhaps one of the greatest values of these remnant old-growth stands is the fact that they offer a glimpse into the future. The central hardwood region, following logging, fires and agricultural abandonment since the turn of the century, has rebounded with millions of acres of second-growth forests dominated by oaks and mesophytic hardwood species (MacCleery 1992). Stands like Collins and Dysart Woods give a good indication of what these forests will look like in 200 years with minimal disturbance (fire exclusion and limited cutting). They also, when viewed in the context of a collective whole of remnant deciduous forests, illustrate that some rather consistent processes (vital ecological attributes) are moving these stands in the same direction and these same forces will probably shape today's second-growth forests in a similar way.

REFERENCES

Abrams, M.D. 1992. Fire and the development of oak forests. Bioscience. 42(5): 346-353.

Benner, D.K. 1971. Effects of canopy openings within a mature oak forest in southeastern Ohio. Athens, OH: Ohio University. Unpublished M.S. thesis.

Bryant, W.S. 1987. Structure and composition of the old-growth forests of Hamilton County, Ohio and environs. In: Hay, R.L.; Woods, F.W.; DeSelm, H.R., eds. Proceedings, 6th central hardwood forest conference: 317-324.

Carmean, W.H.; Hahn, J.T.; Jacobs, R.D. 1989. Site index curves for forest tree species in the Eastern United States. Gen. Tech. Rep. NC-128. U.S. Department of Agriculture, Forest Service, Northcentral Forest Experiment Station. 142 p.

Curtis, J.T. 1959. The vegetation of Wisconsin. Madison, WI: University of Wisconsin Press.

DeSelm, H.R.; Sherman, M.D. 1982. Species diversity and site position at Savage Gulf, Cumberland Plateau, Tennessee. In: Muller, R.N., ed. Proceedings, 4th central hardwood forest conference: 356-366.

Downs, J.A.; Abrams, M.D. 1991. Composition and structure of an old-growth versus a second-growth white oak forest in southwestern Pennsylvania. In: McCormick, L.H.; Gottschalk, K.W., eds. Proceedings, 8th central hardwood forest conference. Gen. Tech. Rep. NE-148. U.S. Department of Agriculture, Forest Service, Northeastern Experiment Station: 207-223.

Eyre, F.H., ed. 1980. Forest cover types of the United States and Canada. Society of American Foresters. 148 p.

Harcombe, P.A.; White, B.D.; Glitzenstein, J.S. 1982. Factors influencing distribution and first-year survivorship of a cohort of beech (*Fagus grandifolia*) Ehrl. Castanea. 47(2): 148-157.

Hicks, R.R., Jr.; Frank, P.S., Jr. 1984. Relationship of aspect to soil nutrients, species importance and biomass in a forested watershed in West Virginia. Forest Ecology and Management. 8: 281-292.

Lafer, N.G. 1968. Tree composition of Dysart Woods. Athens, Ohio: Ohio University. Unpublished. M.S. thesis.

Lafer, N.G.; Wistendahl, W.A. 1970. Tree composition of Dysart Woods, Belmont County, Ohio. Castanea. 35(4): 302-308.

Loftis, D.L. 1989. Regeneration of southern hardwoods: some ecological concepts. In: Proceedings, national silviculture workshop; Petersburg, AK.

Loucks, O. 1970. Evolution of diversity, efficiency and community stability. American Zoology. 10: 17-26.

MacCleery, D.W. 1992. American forests: a history of resilience and recovery. FS-540. U.S. Department of Agriculture, Forest Service. 59 p.

Monk, C.D. 1967. Species diversity in the eastern deciduous forest with particular reference to north-central Florida. American Naturalist. 101: 173-187.

Norwacki, G.J.; Abrams, M.D.; Lorimer, C.G. 1990. Composition, structure and historical development of northern red oak stands along an edaphic gradient in northern Wisconsin. Forest Science. 36: 276-292.

Pallardy, S.G.; Nigh, T.A.; Garrett, H.E. 1988. Changes in forest composition in central Missouri: 1968-1982. American Midland National. 1120: 380-390.

Schlesinger, R.C. 1989. Dynamics of the sugar maple component of a white oak-yellow-poplar community. In: Rink, G.; Budelsky, C.A., eds. Proceedings, 7th central hardwood forest conference. Gen. Tech. Rep. NC-132. U.S. Department of Agriculture, Forest Service, Northcentral Experiment Station: 262-266.

Shannon, C.E.; Weaver, W. 1949. The mathematical theory of communication. Urbana, IL: University of Illinois Press. 125 p.

Shugart, H.H. 1987. Dynamic ecosystem consequences of tree birth and death patterns. Bioscience. 37(8): 596-601.

Smith, J.R. 1979. Composition and structure of Dysart Woods, a mature forest ecosystem in southeastern Ohio. Athens, OH: Ohio University. Unpublished M.S. thesis.

Stout, S.L. 1991. Stand density, stand structure, and species composition in transitional oak stands of northwestern Pennsylvania. In: McCormick, L.H.; Gottschalk, K.W., eds. Proceedings, 8th central hardwood forest conference: 194-206.

Ward, J.S.; Parker, G.R. 1998. Fifty-five years of association and diversity dynamics in an old-growth forest in Indiana: Davis-Perdue Research Forest 1926-1981. In: Hay, R.L.; Woods, F.W.; DeSelm, H., eds. Proceedings, 6th central hardwood forest conference: 369-376.

CHARACTERIZATION OF COARSE WOODY DEBRIS ACROSS A 100 YEAR CHRONOSEQUENCE OF UPLAND OAK-HICKORY FORESTS

Travis W. Idol, Phillip E. Pope, Rebecca A. Figler, and Felix Ponder, Jr.[1]

Abstract—Coarse woody debris is an important component influencing forest nutrient cycling and contributes to long-term soil productivity. The common practice of classifying coarse woody debris into different decomposition classes has seldom been related to the chemistry/biochemistry of the litter, which is the long term objective of our research. The objective of this preliminary study was to measure the volume, mass and nutrient content of the different decay classes of the down dead wood (DDW) component of coarse woody debris in upland hardwood stands of different ages. Three oak-hickory stands in southern Indiana: aged 1, 31, and 80-100 years since harvest were chosen for this study. Volume, mass, and C, N, S, and P content were determined on DDW from each decay stage in each stand. Results show that there is a large decrease in DDW volume and mass from recently harvested stands to more mature stands. The dominant decay stage shifts from Class II in the 1 year-old stand to Class III in the 31 year-old stand. The decay stages also have significantly different DDW density and C:N ratios, but only if outer and inner woody material are separated. The decomposition classes used to distinguish DDW correspond to distinguishable stages of DDW decay, as indicated by different C:N ratios and wood densities. The outer woody material seems to decay more quickly than the inner material, which is likely due to lower initial C:element ratios. Further work is needed in order to relate these patterns of coarse woody debris decay to nutrient mineralization and immobilization patterns.

INTRODUCTION

Maintaining the long-term productivity of managed forest soils is essential for the conservation of our forest resources. Finding ways to prevent soil erosion and soil compaction and to maintain soil structure and soil organic matter content are some of the goals of long-term soil productivity research. These indicators of soil quality are relatively easy to measure and quantify, but one aspect of soil quality that is not so easily assessed is nutrient cycling.

Some aspects of nutrient cycling in temperate forests have been well studied in the past. Nutrient availability (Powers 1990, Roy and Singh 1995), nutrient uptake rates and nutrient partioning (Habib and others 1993, Robinson 1986), nutrient leaching (Jordan and others 1993, Yin and others 1993), and returns of nutrients from leaf (Gholz and others 1985, Taylor and others 1989) and fine root turnover (Joslin and Henderson 1987, McClaugherty and others 1982) have been studied by numerous researchers. A topic that has received less attention in nutrient cycling studies is the contribution of coarse woody debris (CWD). Coarse woody debris is generally defined as dead woody material with a diameter of 10 centimeters or greater. This includes a range of woody debris from fallen logs and branches to standing dead trees and stumps. As a subset of this material, down dead wood (DDW) is considered to be those branches, logs, and stumps that are in contact with the soil. Many studies have determined the amount and relative state of decay of either CWD or DDW in both managed and old-growth forests of the Central Hardwoods Region (Jenkins and Parker 1997, McCarthy and Bailey 1994, Muller and Liu 1991, Richards and others 1995, Shifley and others 1995), but fewer have attempted to characterize the nutrient content or decay rate of this material (Abbott and Crossley, Jr. 1982, MacMillan 1988). This information is important for our understanding of the role of large dead woody material in forest nutrient cycling and forest soil productivity.

Although there is relatively little information regarding the nutrient content and decay rate of CWD, there are visual evaluations of the state of decay of CWD that are used by both university researchers (Muller and Liu 1991, Jenkins and Parker, 1997) and U.S. Forest Service personnel (Shifley and others 1995). These evaluations are based upon many visual cues, including bark slippage, penetration of visible decay into the core of the log, the number and size of branches remaining on the log, the shape of the log, the physical integrity of the log, and the degree of burial in the soil for DDW. Although these visual classification systems are useful, they are qualitative assessments and are subject to interpretation by the investigator. There is also the possibility that changes with increasing decay in certain CWD characteristics differ by species. Quantitative information about the elemental and biochemical nature of CWD at the different decay stages is necessary in order to assess the role of CWD in forest nutrient budgets and nutrient cycling.

Finally, most coarse woody debris studies have been conducted in either mature or old growth forest stands. Few have attempted to characterize the changes in CWD

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at different stages of stand development (Jenkins and Parker 1997, McCarthy and Bailey 1994). These studies have shown that there are significant and important differences in the volume, biomass, and distribution of CWD into the different decay classes with stand age. Understanding the dynamics of CWD as a stand develops from recently-harvested to mature or old growth stages is important if we are to accurately assess the role of CWD in forest regeneration and development.

We decided to restrict our study of coarse woody debris to down dead wood (DDW), i.e., fallen dead logs and branches in contact with the soil and tree stumps less than one meter high. This material is greatly influenced by the biotic and abiotic soil environment and in turn should directly influence biological and chemical processes in the soil. Therefore, our designation of down dead wood differs from the general definition of coarse woody debris in that our study does not include standing dead trees or broken treetops touching the ground that are still attached to the trunk. The two objectives for our research were: 1) to assess the volume and mass of DDW in upland oak-hickory forests of different age; and 2) to determine the nutrient content of DDW from the different decomposition classes as used by Thomas (1979).

MATERIALS AND METHODS

Study Site Descriptions

This study was implemented at the Southern Indiana Purdue Agricultural Center in Dubois County, Indiana. The soils in this area are different families within the family of fine-silty, mixed mesic Ultic Hapludalfs. The mean annual temperature is 12 degrees Celcius, and the mean annual precipitation is 1150 millimeters. The ecological land-type phase of these stands is classified as a Quercus alba-Acer saccharum Parthenocissus dry mesic ridge (USDA 1995). Three historically oak-hickory dominated stands were chosen for this study. The first was selectively harvested in 1996 (Site 1) with a post-harvest herbicide of undesirable trees and coppicing of desirable trees during the same year; thus we consider this stand to have undergone clearcutting. The second stand was clearcut harvested in 1966 (Site 2). The third is a mature stand dominated by white oak (*Quercus alba*) in the overstory and has not been harvested for the last 80-100 years (Site 3). The vegetation, climatic, and soil data for the three stands is given in Table 1. For the stand harvested in 1996, pre-harvest vegetation data is listed.

Field Sampling

Down dead wood was sampled according to the protocol of Thomas (1979) as used by Jenkins and Parker (1997). Three circular plots measuring 500 square meters were established in Sites 1 and 3, the mature and recently-harvested stands. In order to sample from locations with similar physiographic and soil characteristics, we had to restrict our sampling to two plots in Site 2, the stand harvested in 1966. The length and mid-point diameter of all DDW at least 10 centimeters in diameter were sampled. If a log or branch tapered to a diameter of less than 10 centimeters, only that portion of the DDW at least 10 centimeters in diameter was included. Lateral branches greater than 10 centimeters in diameter on down logs were also measured. Where a piece of DDW crossed the boundary of the circular plot, only that portion within the plot was measured. Each piece of DDW was evaluated and was assigned a decomposition class. The criteria for this classification scheme are listed in Table 2. The length and mid-point diameter of the DDW were used to calculate DDW volume, using the equation for the volume of a cylinder.

Table 1—Vegetation inventory for upland hardwood forests in southern Indiana. Overstory trees are all 10 cm or greater dbh. Understory trees are all 2.5 to 9.9 cm dbh

		Saplings		Overstory		
Age	N	Species		Species		
			Stems/ha		*Stems/ha*	(m^2/ha)
1	3	*Acer saccharum*	2370	*Acer saccharum*	123	(5.6)
		Nyssa sylvatica	554	*Quercus alba*	71	(15.9)
				Quercus rubra	40	(12.6)
				Carya glabra	40	(4.7)
		All species	3780	All species	384	(47.9)
31	2	*Acer saccharum*	741	*Acer saccharum*	515	
		Asimina triloba	546	*Prunus serotina*	300	
				Sassafras albidum	143	
		All species	1950	All species	1430	
100	3	*Acer saccharum*	740	*Acer saccharum*	253	(3.9)
				Quercus alba	103	(19.0)
		All species	770	All species	445	(27.4)

Table 2—Classification scheme for down dead wood decomposition stage, taken from Thomas (1979)

Character	Class I	Class II	Class III	Class IV	Class V
Bark	Intact	Mostly intact	Mostly absent	Absent	Absent
Structural integrity	Sound	Sapwood rotting	Heartwood sound	Heartwood rotten	None
Branches	All twigs present	Larger twigs present	Larger branches present	Branch stubs present	Absent

Laboratory Methods

In order to estimate biomass and nutrient content of the down dead wood, two cross sections from one log per decomposition class were taken. For Class II and III material, we chose logs that were approximately 20-30 centimeters in diameter. We were only able to locate a single piece of Class I material, found in Site 3. Two cross-sections from this Class I log, each approximately 10-15 centimeters in diameter, were taken for analysis. For Classes IV and V DDW, there were no intact cylinders from which to take a cross-section; therefore, various irregularly shaped pieces of material from these classes were taken for analysis. This sampling was done within each stand so that differences in DDW characteristics by stand age as well as by decomposition class could be assessed.

Each cross section or piece of DDW was cut into smaller pieces and dried at 65 degrees Celsius to constant weight (approximately 1 week). Because differences may exist in DDW characteristics between the inner and outer wood of Class I, II, and III material, within each cross-section we separated the outer 2-3 centimeters of wood from the inner wood. The outer wood consisted mainly of the bark and sapwood; the inner wood consisted mainly of the heartwood.

To determine the biomass of down dead wood, the density of the material at the different decomposition stages was determined using the soil clod bulk density method (Blake and Hartge 1986). Pieces of oven-dried DDW material were weighed, dipped in liquid Saran resin, dried overnight at 105 degrees Celsius, and the dry weight and displacement volume measured. Becaue the outer wood diameter varied between 2 and 3 centimeters, we used a diameter of 2.5 centimeters to calculate the mass of entire logs.

To determine the nutrient content of down dead wood, pieces of each cross section were ground in a Wiley mill until the material passed through a 1 millimeter diameter mesh screen. The ground DDW was re-dried at 65 degrees Celsius for at least 24 hours. Total C, N, and S content were determined using a LECO CNS 2000 elemental analyzer. Total P content was determined using the phospho-molybdate blue colorimetric procedure (Olsen and Sommers 1982) after digestion of the material in perchloric acid and hydrogen peroxide. Nutrient concentrations (micrograms per gram of tissue) were then multiplied by the estimated biomass in order to determine the total nutrient content of DDW within each of the stands.

Statistics

Our initial hypotheses were: 1) The volume, mass, and nutrient content of DDW will decrease with increasing stand age; 2) the dominant decay stage of DDW will also increase with stand age; and 3) the density, C:N, C:S, and C:P ratios of DDW will decrease with increasing decay stage.

All statistical analyses were carried out using the ANOVA procedure in SAS (SAS Institute, Inc. 1989) with an alpha-level of 0.05. Where a significant difference was indicated by the ANOVA, Duncan's multiple range test with an alpha-level of 0.05 was used as the means separation test. Because this study does not include true replication of stand age, the true error associated with differences in DDW characteristics by stand age cannot be known. We used plot within stand as the replication for stand age. The error term associated with this type of analysis may be biased, but without true replication of stand age, we cannot determine the degree of bias, if any.

In our initial analysis, DDW density and concentrations of C, N, S, and P were the dependent variables and the outer or inner DDW portion of each decomposition class within each stand age was the independent variable. For Class IV and V DDW no distinction between inner or outer wood was made. Although the density and element concentration of DDW in different classes differed significantly, there was no difference by stand age within a decomposition class. Therefore the values within a decomposition class across stand ages were combined and the average used to calculate DDW mass and element content for the individual decomposition classes and the location within a decomposition class.

In order to test hypothesis 1, we compared the total volume, mass, and C, N, S, and P content of all classes of DDW combined within each stand. The high degree of plot to plot variability coupled with low degrees of freedom for the error term (based on number of plots) led to our finding no significant differences by stand age. Therefore, we decided to compare DDW within each decomposition class within each stand to other classes of DDW within the same stand and across all stands. In this analysis, DDW volume, mass, and element content (C, N, S, P) were the dependent variables and decomposition class by stand age was the independent variable. This analysis was also used to test hypothesis 2.

Because the inner and outer wood of DDW may have different chemical characteristics, we repeated the above analysis, subdividing Classes I, II, and III DDW into inner and outer material. In order to test the third hypothesis, this classification of DDW was used to compare DDW density and DDW C:N, C:S, and C:P ratios. Comparisons of DDW classes and the location within a class were made with this analysis. No comparison of stand ages was made with respect to these variables; rather, the average values across the stands were used.

RESULTS

Although there is a striking difference between the total amount of DDW in the 1 year-old stand and the 31 and 80-100 year-old stands (table 3), the high degree of variability among the plots masked any statistical differences by stand age. We compared our results to those of Jenkins and Parker (1997), who estimated the volume of DDW in numerous hardwood forest stands in southern Indiana (fig. 1). The total volume of DDW in our study agrees with their results; however, differences are evident in the distribution of DDW between the two studies. In this study the mass and volume of Class II material was greater in the 1 year-old stand and Class IV material was greater in all stands (age 1, 31, and 80-100 years) of our study than in their study.

The dominant decay class differed somewhat by stand age (table 3). The volume and mass of Class II DDW in the 1

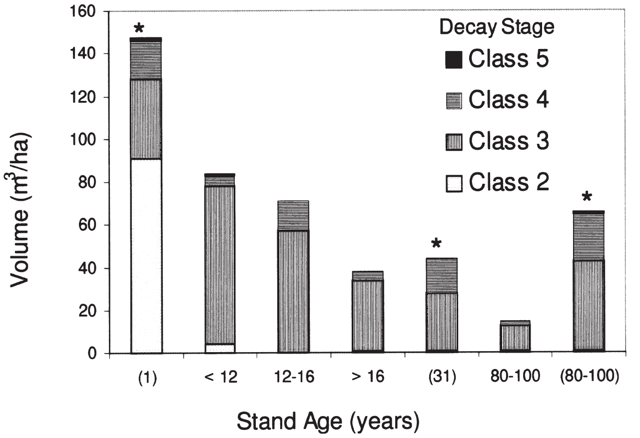

Figure 1—Down dead wood volume by decay stage across a chronosequence of upland hardwood forests in southern Indiana. (*) indicates stands measured in this study. All others taken from Jenkins and Parker (1997).

Table 3—Total volume, mass, and nutrient content of down dead wood by decay class across a 100-year chronosequence of upland oak-hickory forests in southern Indiana

Age	Decay class	N	Volume	Mass	C	N	S	P
Yrs			*m³/ha*	-------*Mg/ha*-------		-----------------*Kg/ha*-----------------		
1	II	3	90.8a	85.2a	42.8a	80.0a	26.4a	4.7abc
1	III	3	38.3bc	35.3b	18.2bc	42.4abc	17.5ab	1.8bc
1	IV	3	17.8bc	15.7b	7.5bc	27.9abc	14.6abc	4.9ab
1	V	3	1.2c	1.0b	0.5c	5.5c	7.4bcd	4.6abc
Total			148.1	137.2	69.0	155.8	66.0	16.0
31	II	3	0.6c	0.6b	0.3c	1.4c	0.2d	0.1c
31	III	3	27.1bc	24.8b	12.7bc	60.9ab	8.9bcd	1.7bc
31	IV	3	15.9bc	14.0b	6.7bc	51.7abc	7.3bcd	4.6abc
31	V	3	0.5c	0.4b	0.2c	5.1c	0.6cd	0.3bc
Total			44.2	39.8	19.9	119.1	17.0	6.7
100	II	3	0.7c	0.7b	0.3c	1.6c	0.2d	0.1c
100	III	3	41.7b	37.9b	19.5b	72.8a	11.7bcd	1.7bc
100	IV	3	22.3bc	19.6b	9.4bc	72.5a	10.3bcd	6.5a
100	V	3	1.0c	0.8b	0.4c	9.7bc	1.1cd	0.5bc
Total			65.7	59.0	29.7	156.5	23.3	8.8

Columns followed by the same letters do not differ significantly using Duncan's multiple range test with alpha = 0.05. All values are based on plot averages; thus "N" refers to the number of plots sampled in each stand.

year-old stand was significantly greater than any other class within that stand and across all stands. In the 80-100 year-old stand, the volume of Class III DDW was significantly greater than the volume of Class II or Class V DDW, but there was no significant difference in DDW mass by decomposition class in either the 31 or 80-100 year-old stands.

The C, N, S, and P content of the down dead wood followed a somewhat different trend than the distribution of DDW volume and mass. Although the mass and C content of Class III material is greater than Class IV material in all three stands, the amount (kilograms per hectare) of N, S, and P held in these two classes of DDW is similar. Lower C:element ratios (table 4) in Class IV material account for the similarity in the amount of nutrients held in DDW of these two classes despite the difference in DDW mass. Class V DDW had the least mass (megagrams per hectare) and nutrient content (kilograms per hectare). Class II material was also a very small component of the DDW mass and nutrient content in the 31 and 80-100 year-old stands. The amount of down dead wood mass (metric tons per hectare) and nutrients generally did not differ significantly by location within a log (table 5).

The bulk density and C:N ratio of down dead wood, however, did show significant trends by decay stage and location (table 4). The density of outer sapwood and bark of Class I, II, and III DDW is significantly greater than Class IV and V DDW. The density of the inner heartwood of Class I and II material is also significantly greater than Class IV and V material. The C:N ratio of inner heartwood decreased significantly from Class I to III DDW and was significantly greater than Class IV and V material. Outer sapwood and bark C:N ratios of Classes I, II, and III were significantly lower than the inner heartwood C:N ratios of Class I, II, and III material.

DISCUSSION

The mass and distribution of down dead wood (DDW) in this study was consistent with that found in other studies in the Central Hardwood Region. The trend of decreasing DDW mass with increasing stand age is supported by the work of Jenkins and Parker (1997), which was also conducted in southern Indiana hardwood forests. They found that Class III DDW dominated the volume of total DDW in stands of different ages. Although the number of pieces of Class IV and V DDW may be expected to increase with stand age, the volume and mass of DDW may still be dominated by Class III logs, which retain much more of their original mass and volume than DDW in later decay stages.

Although our study was of stands in the same geographical region as those of Jenkins and Parker (1997), our results suggest a greater volume and mass of Class II DDW in recently-harvested stands and a greater volume and mass of Class IV DDW in all age stands. Part of this discrepancy is probably due to the limited nature of our study. We investigated only one stand per age class, whereas Jenkins and Parker (1997) studied many stands per age class. However, part of this discrepancy may be due to the fact that different stand ages were investigated in the two studies. Our study included stands aged 1, 31, and 80-100 years since harvest. Jenkins and Parker (1997) investigated stands 8-12, 12-16, and 80-100 years of age. Although they did not study stands younger than 8 years after harvest, they hypothesized that Class III material would dominate stands less than 12 years of age. McCarthy and Bailey (1994) assessed the coarse woody

Table 4—Indicators of down dead wood decay across a 100-year chronosequence of upland oak-hickory forests in southern Indiana

Decay class	In/out	N	Density	C:N	C:P
			Mg/m^3		
I	Out	3	0.930ab	220:1c	2903:1c
II	Out	3	0.938ab	172:1c	2533:1c
III	Out	3	0.932ab	144:1c	2893:1c
I	In	3	0.921ab	839:1a	20622:1bc
II	In	3	0.950a	861:1a	35533:1ab
III	In	3	0.901bc	419:1b	48400:1a
IV	—	3	0.878c	130:1c	1550:1c
V	—	3	0.816c	42:1c	854:1c

Values within a column followed by the same letter do not differ significantly using Duncan's multiple range test with alpha = 0.05.
All values are based on the average across all three stands; thus "N" refers to the number of stands sampled.
"In" refers to the inner heartwood of a log cross-section, more than 2.5 cm from the outer edge of the log. "Out" refers to the outer 2.5 cm of a log cross-section, including the bark and sapwood.

Table 5—Volume, mass, and nutrient content of down dead wood by decay stage and location across a 100-year chronosequence of upland oak-hickory forests in southern Indiana

Age	Decay class	In/out	N	Volume	Mass	C	N	S	P
Yrs				*m³/ha*	-----------*Mg/ha*----------		-------------------*Kg/ha*-----------------		
1	II	In	3	49.3a	46.9a	23.9a	20.6bcd	4.2bc	0.1c
1	II	Out	3	41.5ab	38.3ab	18.9ab	59.4ab	22.2a	4.7ab
31	II	In	2	0.2d	0.2d	0.1d	0.2e	0.0c	<0.0c
31	II	Out	2	0.4d	0.4d	0.2d	1.3de	0.2c	0.1c
100	II	In	3	0.2d	0.2d	0.1d	0.2e	0.0c	<0.0c
100	II	Out	3	0.5d	0.4d	0.2d	1.4de	0.2c	0.1c
1	III	In	3	22.0bcd	20.1bcd	10.4bcd	13.2cde	3.2c	0.1c
1	III	Out	3	16.3bcd	15.2bcd	7.7bcd	29.2bcde	14.3ab	1.8bc
31	III	In	2	14.5bcd	13.1bcd	6.8bcd	16.5cde	3.2c	0.2c
31	III	Out	2	12.6cd	11.7cd	6.0bcd	44.3abcd	5.7bc	1.5bc
100	III	In	3	31.1abc	28.0abc	14.5abc	35.5abcde	6.9bc	0.4c
100	III	Out	3	10.6cd	9.9cd	5.0cd	37.3abcde	4.8bc	1.2bc
1	IV	—-	3	17.8bcd	15.7bcd	7.5bcd	27.9bcde	14.6ab	4.9ab
31	IV	—-	2	15.9bcd	14.0bcd	6.7bcd	51.7abc	7.3bc	4.6ab
100	IV	—-	3	22.3bcd	19.6bcd	9.4bcd	72.5a	10.3bc	6.5a
1	V	—-	3	1.2d	1.0d	0.5d	5.5de	7.4bc	4.6ab
31	V	—-	2	0.5d	0.4d	0.2d	5.1de	0.6c	0.3c
100	V	—-	3	1.0d	0.8d	0.4d	9.7cde	1.1c	0.5c

Values within a column followed by the same letters do not differ significantly using Duncan's multiple range test with alpha = 0.05.
All values are plot averages; thus "N" refers to the number of plots sampled within each stand.
"In" refers to the inner heartwood of a log cross-section, more than 2.5 cm from the outer edge of the log. "Out" refers to the outer 2.5 cm of a log cross-section, including the bark and sapwood.

debris volume and mass of forest stands in the Central Appalachians that ranged in age from clearcut to old-growth. As in our study, they found that Class II CWD dominated the clearcut stand. They also found that Class IV and V material was more abundant in older forest stands than is suggested by Jenkins and Parker (1997), also similar to the results of our study.

The differences we found with respect to DDW density and C:N ratios among the different classes, I-V, lends chemical support to the class distinctions of coarse woody debris made in the field. If these class distinctions are to have any meaning with respect to patterns of nutrient cycling, one would expect there to be significant differences with respect to C:element ratios. The differences between the bark plus sapwood and heartwood C:N ratios, however, illustrate the importance of distinguishing these two components of DDW. The initial chemical composition of these two substrates is different, and this may affect their decay dynamics. By the time DDW reaches Class III, the bark and sapwood are porous, loose or absent, indicating that there is substantial decay of this material. However, the inner heartwood is often still intact. This visual evaluation is supported by the lower C:element ratios of outer woody material versus inner woody material in Classes I and II. Although tree species may differ with respect to outer wood and inner wood decay rates, higher C:N ratios in outer wood than inner wood is to be expected among most live woody plants.

It is generally thought that a C:N ratio of 15:1 to 30:1 is necessary for net mineralization of nitrogen from organic residues in soil systems (Foth, 1978). The C:N ratios of DDW of all decomposition classes in this study were greater than 40:1. However, there is no experimental evidence to substantiate claims concerning the critical C:element ratios for net mineralization of N, P, or S from coarse woody debris. There have been studies in a variety of forest types that have investigated mass loss and nutrient concentrations in CWD at different stages of decomposition (Macmillan 1988, Abbott and Crossley 1982, Lang and Forman 1978). None of these, however, have investigated N, P, or S mineralization and immobilization patterns directly. Laboratory and field studies on the decomposition and nutrient mineralization of leaf, forest floor, and fine root litter are abundant, but studies with CWD are surprisingly absent from the literature.

Although nutrient mineralization and immobilization patterns of decaying DDW are difficult to assess at present, there is evidence that suggests DDW decay rates are related to C:N ratios. Macmillan (1988) found that DDW density was highly correlated with the DDW C:N ratio for oak (*Quercus*), hickory (*Carya*), maple (*Acer*), and beech (*Fagus*) DDW. We also found a significant relationship between density of different decomposition classes and the C:N ratio, but only when inner heartwood was examined (data not shown).

One of the visual cues that distinguishes Class II from Class III DDW is the onset of significant bark slippage and the absence of most smaller limbs. Because the bark and outer woody tissues are higher in N, S, and P, they would be expected to decay at a rate much faster than the inner heartwood material. This more rapid decay means that Class II DDW quickly progresses into Class III DDW. A major difference between Class III and Class IV DDW is the integrity of the inner woody material. Because this material has a much lower N, S, and P content initially, it decays much more slowly than the sapwood and bark. Therefore, the transition between Class III and Class IV is more gradual and lengthy than the transition from Class II to Class III. Given this set of circumstances, it is easy to understand why DDW in most stands is dominated by Class III material. Future research on DDW decay and nutrient dynamics should focus on direct examinations of nutrient mineralization and immobilization patterns so that critical C:element ratios can be established. Also, Van Lear (1993) has pointed out that we know almost nothing about the amount and decay of coarse root systems after a harvest or canopy tree death. Studying this type of coarse woody debris presents methodological challenges, but we will not have a full picture of forest nutrient cycling until this gap in our knowledge is filled.

CONCLUSIONS

Our first hypothesis stated that we expected the volume and mass of down dead wood in the recently clearcut stand to be significantly greater than in the 31 and 80-100 year-old stands. Although there is a large difference in the volume and mass between the recently clearcut and the other stands, we did not find a significant difference. This is most likely due to the high degree of variation between plots within a stand and the low number of plots (2 or 3) in each stand used to estimate volume and mass of DDW. However, our results did follow the same general trend that Jenkins and Parker (1997) and McCarthy and Bailey (1994) found for other Central Hardwood forest stands.

Our second hypothesis stated that the most abundant down dead wood class in the 1 year-old stand would be Class II; whereas, decomposition classes III and IV would be the most abundant DDW classes in the 31 and 80-100 year-old stands. This hypothesis was supported by our results. This is somewhat different than what was found by Jenkins and Parker (1997), but it agrees well with the findings of McCarthy and Bailey (1994). The rapid decomposition of the sapwood and bark and the slow decomposition of the inner heartwood are probably the main reasons Class III DDW is so abundant in forest stands of all ages. Our final hypothesis stated that we expected the density and C:element ratios to decrease with increasing decomposition class. This hypothesis is true for density and the C:N ratios of inner heartwood DDW. There was a significant decrease in the C:N ratio from Classes I and II to Class III and from Class III to Classes IV and V for the inner heartwood material. These differences in density and C:N ratio between decomposition classes lend direct chemical support for the classification schemes used to distinguish the different decomposition stages of coarse woody debris. However, they also illustrate the heterogeneous nature of material within a piece of DDW and the need to study different fractions within a log as well as different decay classes of whole logs.

REFERENCES

Abbott, D.T.; Crossley, D.A., Jr. 1982. Woody litter decomposition following clear-cutting. Ecology. 63: 35-42.

Blake, G.R.; Hartge, K.H. 1986. Bulk density. In: Klute, A., ed. 1986. Methods of soil analysis. Madison, WI: American Society of Agronomy: 363-375.

Donaldson, J.M.; Henderson, G.S. 1990. Nitrification potential of secondary-succession upland oak forests: II. Regulation of ammonium-oxidizing bacteria populations. Soil Sci. Soc. Am. J. 54: 898-902.

Foster, N.W. 1989. Influences of seasonal temperature on nitrogen and sulfur mineralization/immobilization in a maple-birch forest floor in central Ontario. Can. J. Soil Sci. 69: 501-514.

Foth, H.D. 1978. Fundamentals of soil science. New York: John Wiley. 436 p.

Gholz, H.L.; Perry, C.S.; Cropper, Jr., W.P.; Hendry, L.C. 1985. Litterfall, decomposition, and nitrogen and phosphorus dynamics in a chronosequence of slash pine (*Pinus elliottii*) plantations. Forest Science. 31: 463-478.

Habib, R.; Millard, P.; Proe, M.F. 1993. Modelling the seasonal nitrogen partitioning in young sycamore (*Acer pseudoplatanus*) trees in relation to nitrogen supply. Annals of Botany. 71: 453-459.

Idol, T.W. 1997. Fine root and nutrient dynamics across a chronosequence of oak-hickory forests. West Lafayette, IN: Purdue University. 137 p. M.S. thesis.

Idol, T.W.; Pope, P.E.; Tucker, J.; Ponder, F., Jr. [In press]. The role of fine root dynamics in the N and P cycles of regenerating upland oak-hickory forests. Proceedings of the 9th biennial southern silvicultural research conference.

Jenkins, M.A.; Parker, G.R. 1997. Changes in down dead wood volume across a chronosequence of silvicultural openings in southern Indiana forests. Proceedings of the 11th central hardwood forest conference: 162-169.

Jordan, T.E.; Correll, D.L.; Weller, D.E. 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. Journal of Environmental Quality. 22: 467-473.

Joslin, J.D.; Henderson, G.S. 1987. Organic matter and nutrients associated with fine root turnover in a white oak stand. Forest Science. 33: 330-346.

Kronzucker, H.J.; Siddiqi, M.Y.; Glass, A.D.M. 1997. Conifer root discrimination against soil nitrate and the ecology of forest succession. Nature. 385: 59-61.

Lang, G.E.; Forman, R.T. 1978. Detrital dynamics in a mature oak forest: Hutcheson Memorial Forest, New Jersey. Ecology. 59: 580-595.

Lawrence, G.B.; David, M.B. 1997. Response of aluminum solubility to elevated nitrification in soil of a red spruce stand in eastern Maine. Environ. Sci. Technol. 31: 825-830.

MacMillan, P.C. 1988. Decomposition of coarse woody debris in an old-growth Indiana forest. Canadian Journal of Forest Research. 18: 1353-1362.

Marques, R.; Ranger, J.; Villette, S.; Granier, A. 1997. Nutrient dynamics in a chronosequence of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) stands on the Beaujolais Mounts (France). 2. Quantitative approach. For. Ecol. Manage. 92: 167-197.

Matson, P.A.; Boone, R.D. 1984. Natural disturbance and nitrogen mineralization: wave-form dieback of mountain hemlock in the Oregon Cascades. Ecology. 65: 1511-1516.

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

McClaugherty, C.A.; Aber, J.D.; Melillo, J.M. 1982. The role of fine roots in the organic matter and nitrogen budgets of two forested ecosystems. Ecology. 63: 1481-1490.

Muller, R.N.; Liu, Y. 1991. Coarse woody debris in an old-growth deciduous forest on the Cumberland Plateau, southeastern Kentucky. Canadian Journal of Forest Research. 21: 1567-1572.

Olsen, S.R.; Sommers, L.E. 1982. Phosphorus. In: Page, A.L.; Miller, R.H.; Keeney, D.R., eds. 1982. Methods of soil analysis part 2-chemical and microbiological properties. Madison, WI: 2nd American Society of Agronomy: 403-430.

Powers, R.F. 1990. Nitrogen mineralization along an altitudinal gradient: interactions of soil temperature, moisture, and substrate quality. For. Ecol. Manage. 30: 19-29.

Pritchett, W.L.; Fisher, R.F. 1987. Properties and management of forest soils. New York: John Wiley. 494 p.

Richards, R.H.; Shifley, S.R.; Rebertus, A.J.; Chaplin, S.J. 1995. Characteristics and dynamics of an upland Missouri old-growth forest. Proceedings of the 10th central hardwood forest conference: 11-22.

Robinson, D. 1986. Limits to nutrient inflow rates in roots and root systems. Physiol. Plantarum. 68: 551-559.

Roy, S.; Singh, J.S. 1995. Seasonal and spatial dynamics of plant-available N and P pool and N-mineralization in relation to fine roots in a dry tropical forest habitat. Soil Biol. Biochem. 27: 33-40.

Ruess, R.W.; Van Cleve, K.; Yarie, J.; Viereck, L.A. 1996. Contributions of fine root production and turnover to the carbon and nitrogen cycling in taiga forests of the Alaskan interior.

SAS Institute Inc. 1989. SAS for windows. Release 6.12. Cary, NC: SAS Institute Inc.

Shifley, S.R.; Roovers, L.M.; Brookshire; B.L. 1995. Structural and compositional differences between old-growth and mature second-growth forests in the Missouri Ozarks. Proceedings of the 10th central hardwood forest conference: 23-36.

Taylor, B.R.; Parsons, W.F.J.; Parkinson, D. 1989. Decomposition of *Populus tremuloides* leaf litter accelerated by addition of Alnus crispa litter. Canadian Journal of Forest Research. 19: 674-679.

Thomas, J.W. 1979. Wildlife habitat in managed forests in the Blue Mountains of Oregon and Washington. Agric. Handb. 553. Washington, DC: U.S. Department of Agriculture, Forest Service.

U.S. Department of Agriculture. 1995. Field guide: ecological classification of the Hoosier National Forest and surrounding areas of Indiana. Washington, DC: U.S. Department of Agriculture, Forest Service. 111 p.

Van Lear, D.H. 1993. Dynamics of coarse woody debris in southern forest ecosystems. Proceedings of the workshop for coarse woody debris in southern forests: effects on biodiversity.

Willard, K.W.J. 1997. Indicators of nitrate export from forested watersheds of the Chesapeake Bay region. In: Proceedings of the 11th central hardwood forest conference. 284 p.

Yin, X.; Foster, N.W.; Arp, P.A. 1993. Solution concentrations of nutrient ions below the rooting zone of a sugar maple stand: relations to soil moisture, temperature, and season. Canadian Journal of Forest Research. 23: 617-624.

Reforestation / Reclamation

RENEWING A FOREST ECOSYSTEM IRRIGATED WITH TREATED WASTEWATER

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Abstract—Treated wastewater is being irrigated on declining upland forest ecosystems in central Pennsylvania. To renew the forest community, four clearcut treatments were administered to determine the effects of season of harvest and residue on the resulting vegetation. Two years after the clearcut treatments, forb, grass and vine plants covered > 90 percent of the ground surface. Densities of naturally occurring seedling- and sapling-sized stems of shrub and tree species were insufficient for a fully stocked forest stand. Two years after clearcutting, the average densities of shrub and tree seedling-sized stems were 3,056 and 1,319 stems per hectare, respectively. The average densities of shrub and tree sapling-sized stems were 1,209 and 846 stems per hectare, respectively. There were some significant differences in densities among clearcut treatments, however, there was no consistently better treatment after two years.

Artificial regeneration was used to augment natural regeneration. Green ash (*Fraxinus pennsylvanica* Marsh.), quaking aspen (*Populus tremuloides* Michx.), northern red oak (*Quercus rubra* L.) and pin oak (*Q. palustris* Muenchh.) were planted to evaluate their potential survival and growth in the wastewater conditions. At the end of the two growing seasons, overall average survival was 98 percent for green ash and 88 percent for quaking aspen. These two species responded to the additional nutrients and water, and after two growing seasons had overall average total heights of 156 centimeters and 222 centimeters, respectively. Neither northern red oak nor pin oak survived or grew as well as the green ash and quaking aspen. After two growing seasons, the oaks were smaller than when originally planted. Green ash and quaking aspen showed excellent potential to respond to the wastewater irrigation conditions and can be used to renew the forest stands.

INTRODUCTION

Disposal of sewage effluent has become a major problem for many municipalities due to escalating human populations and stricter regulatory controls on disposal methods. Nutrient-rich wastewater from sewage treatment plants is usually discharged into nearby rivers or lakes, often leading to serious environmental problems through eutrophication of aquatic systems. Disposal of secondary treated sewage effluent on land has several potential advantages, including reducing stream pollution and recharging ground water reserves (Sopper and Kardos 1973).

Experiments conducted in forested communities have shown that spray irrigation with chlorinated sewage effluent caused changes in the structure and species composition of vegetation. Increased rates of tree growth and plant biomass, altered species composition of ground-level vegetation, and decreased production and survival of native shrub and tree species have all been reported (Brister and Schultz 1981, Epstein and Sawhill 1977, Sopper and Kardos 1973). It is believed that these changes are due to the addition of nutrients and water to the soil and damage caused by ice loading during winter.

In 1983, The Pennsylvania State University engaged in a full-scale spray-irrigation system for municipal wastewater on 209 hectares of farm and forest land owned by the Pennsylvania Game Commission. The forest land contains ecosystems that are important landscape features to a comprehensive land application system. These include important noise and visual screens for adjacent landowners, all-season wastewater receptors, nutrient accumulators, and habitats for wildlife. The irrigation schedule of 5 centimeters of wastewater per week for 52 weeks, since 1983 had reduced the upland forest overstory density by 33 percent. Many of the remaining 75 to 125 year old trees were in a poor state of health (Larrick and Bowersox 1999). The irrigated forest generally had upper soil nitrate-nitrogen levels > 10 milligrams/liter, which was above the U.S. Public Health potable water standards. This illustrated that the irrigated forest was not renovating the wastewater and needed to be renewed (Storm 1995).

In 1992, an exploratory study with 0.4-hectare units was designed to examine the potential to replace the deteriorating overstory with young vegetation. This study investigated how the amount of irrigation and seasonal timing of application could affect species composition of a forest community being regenerated by clearcutting. This study indicated that woody vegetation was markedly reduced in clearcuts irrigated with 5 centimeters of treated wastewater for 52 weeks per year; forb and vine plants dominated these areas. Clearcuts irrigated with 5 centimeters of wastewater for 26 weeks (during the growing season only) contained more shrub and tree stems, but were still dominated by forb and vine plants. The soil nitrate-nitrogen in the young clearcuts was < 10 milligrams/liter, demonstrating that the young vegetation can reduce the amount of nitrogen in the soil percolate (Storm 1995).

In 1995, a research study was implemented to determine how the wastewater irrigated forest stands could be

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renewed. Three, 8-hectare clearcuts were installed to examine the effect of four harvest treatments on the resulting plant community which was being irrigated weekly with 5 centimeters of wastewater from May through October. This paper presents the first and second year responses of the ground cover plants, naturally occurring understory shrubs and trees, and artificially regenerated tree species to four clearcut harvest treatments.

PROCEDURES

The study area was located in central Pennsylvania, 2 kilometers north of State College. The gently rolling terrain is in the Ridge and Valley region (Lull 1968) on lands managed by the Pennsylvania Game Commission. The Pennsylvania State University operates a wastewater irrigation system over the study area. Treated wastewater is distributed by aboveground sprinkler-heads that rotate 360 degrees for even distribution. The soil was Morrison sandy loam (Ultric, Hapludalf; fine, loamy, mixed, mesic). Site quality was about average, and if occupied by even-aged mixed oak stands, the site index would be about 20 meters at 50 years.

Plant response to season of cutting and residue were studied in an area clearcut in 1995-96. There were three, 8-hectare replications of four, 2-hectare treatment units for a total area of 24 hectares. Each replication contained four harvest treatments which were: Summer/Remove with complete tree removal (cut in August 1995); Summer/Leave with no tree removal (cut in September 1995); Winter/Remove with complete tree removal (cut in January 1996); and Winter/Leave with no tree removal (cut in February 1996). Removal treatments were done by feller-buncher and grapple skidder combination. Leave treatments were done by directional felling with chainsaws, and all residue was lopped to within 1 meter of the ground. All shrubs and trees > 2 centimeters d.b.h. were severed at < 30 centimeters above ground in all treatments. There was greater disturbance in the Remove treatments than in the Leave treatments caused by the dragging of felled stems across the seedbed. Between the Summer/Remove and Winter/Remove treatments, the latter had less impact because the presence of snow on the ground protected the seedbed.

Inventories of ground cover, and abundance of shrub and tree seedling-sized stems (1 to 150 centimeters in height) and sapling-sized stems, including stump sprouts (151 centimeters in height and 1 to 10 centimeters in diameter), were conducted in 1995 to determine the understory composition prior to harvest treatment. During the first (1996) and second (1997) growing seasons after harvest treatment, these inventories were continued on randomly located permanent plots. There were 144 pairs of fenced and unfenced 2-square meter plots (12 pairs per treatment per replication) for the ground cover and seedling inventories, and 96, 100-square meter plots (8 per treatment per replication) for the sapling inventories. Ground cover was separated into three categories: forbs (including broadleaf herbaceous plants), grass, and vine plants (both woody and non-woody). Annual estimations of cover, to the nearest 5 percent were made in August.

Densities of seedling-sized stems by height and species, and densities of sapling-sized stems by d.b.h and species, were measured at the end of each growing season.

Seedlings of four hardwood tree species were planted in spring of 1996 to determine their potential to survive and grow in the wastewater irrigated clearcuts. The species planted as bareroot seedlings were green ash, quaking aspen, northern red oak and pin oak. Plantings were in four pairs of 18-square meter fenced and unfenced plots per treatment unit. A 2 percent solution of glyphosate was sprayed prior to planting to reduce competition. Hand-weeding was done during the rest of the growing season. The fence was 1.3-meters tall and used to deter deer browsing of the planted seedlings. A total of 1,152 bareroot seedlings of each species were planted at a 1-tree/0.5 square meter density. The initial height of each of the seedlings was recorded shortly after planting. Survival and height were measured at the end of the first and second growing seasons. Evidence of browsing was recorded at the end of each growing season, and also in April of 1997 to determine damage caused by white-tailed deer (*Odocoileus virginianus*).

From 1983 to 1995, the study area received 5 centimeters of wastewater irrigation for 52 weeks each year. In 1996 and 1997, these areas were irrigated with 5 centimeters of wastewater weekly for 26 weeks (May 1 to October 31). Irrigation during the winter months was avoided so that regeneration was not damaged by ice loading. The nutrient concentration of the applied effluent can vary considerably depending on the volume of incoming wastewater to the treatment facility, as well as the procedures used to analyze it. Table 1 is a coarse estimate of some of the nutrients added to the study site.

A completely random experimental design was used. Analysis of variance ($\alpha = 0.05$) was used to test replication, treatment and fence (where appropriate)on: ground cover, density of naturally occurring seedlings and saplings, and survival and growth of planted seedlings. Percentage data was arcsine transformed prior to analysis. Tukey's HSD mean separation procedure ($\alpha = 0.05$) was used to determine significant differences between treatments.

RESULTS AND DISCUSSION

Naturally Occurring Plants

Ground cover—Prior to overstory removal, forb, grass and vine coverages were not significantly different among harvest treatments (Table 2). Overall, forb (22 percent), grass (1 percent), and vine (8 percent) plants covered about 30 percent of the ground surface prior to clearcutting.

In 1996, the average coverage of both forb and vine plants was significantly different among treatments (Table 2). The Winter/Leave treatment had the least amount of forb coverage (60 percent) and the most vine coverage (24 percent) of all the treatments. Grass coverage did not differ significantly among treatments in 1996, and the overall average coverage was 2 percent (Table 2).

Table 1—Amount of nitrite-N, nitrate-N, phosphorus, calcium and magnesium annually added in the wastewater from 1983 through 1995[a] and 1996-1997[b]. Values were based on treatment plant output concentration values, which varied considerably, depending on analytical method, supply to and procedures at the treatment plant

Period	Nitrite-N	Nitrate-N	Phosphorus	Calcium	Magnesium
	Kg/ha				
1983-1995	79	69	95	1,188	581
1996	1	32	53	647	330
1997	1	37	79	568	277

[a] Based on 52 weeks of 5 cm of wastewater per week
[b] Based on 26 weeks of 5 cm of wastewater per week

Table 2—Average[a] forb, grass and vine coverage for the harvest treatments, by species group, for before (1995), one year after (1996) and two years (1997) after clearcut

Year and harvest treatment	Forb	Grass	Vine
	Cover pct		
1995			
Summer/remove	24a	0a	10a
Summer/leave	23a	0a	7a
Winter/remove	20a	0a	6a
Winter/leave	20a	2a	8a
Overall	22C	1A	8C
1996			
Summer/remove	75a	2a	9b
Summer/leave	77a	4a	14ab
Winter/remove	72a	1a	12ab
Winter/leave	60b	0a	24a
Overall	71A	2A	15B
1997			
Summer/remove	66a	3a	26a
Summer/leave	67a	5a	22a
Winter/remove	63a	1a	30a
Winter/leave	63a	1a	30a
Overall	65B	3A	27A

[a] Values within each species group by year with the same lower case letter were not significantly different at the 0.05 level. Overall treatment values among years within each species group with the same upper case letter were not significantly different at the 0.05 level.

In 1997, there were no significant differences among harvest treatments as to average forb, grass and vine coverage values (Table 2). Overall, the average coverage of forb (65 percent), vine (27 percent), and grass (2 percent) plants amounted to a total of 94 percent of ground cover during the second growing season.

Approximately twelve forb species were found in the irrigated clearcut areas. Garlic mustard (*Alliaria petiolata* (Bieb.) Cavara & Grande) and spotted touch-me-not (*Impatiens capensis*) comprised most of the forb cover in the clearcuts. These two species tend to dominate any area when present. There were five vine species found in the clearcuts. Bittersweet nightshade (*Solanum dulcamara* L.) was the most abundant vine species followed by Virginia creeper (*Parthenocissus quinquefolia* (L.) Planch).

There were distinctly different post-treatment seedbed conditions among the four harvest treatments, which we expected would have influenced the different forb and vine communities. It was our observation that the Winter/Leave and Summer/Leave treatments had more vine coverage than the Summer/Remove and Winter/Remove treatments, however this was not reflected in the data. Sampling intensity may not have been adequate to detect significant differences in the ground coverage among treatments.

Averaged over all harvest treatments, there were significant, temporal differences in forb and vine coverages (Table 2). The coverage values were all significantly different after the clearcut treatments. Forb coverage increased from 22 percent before the clearcut to 71 percent in the first year after the clearcut and then declined to 65 percent in the second year. This suggests that forb coverage may have reached a maximum level during the first growing season after the clearcuts and was beginning to decline. Vine coverage was significantly different among all inventories. Coverage increased with time in all of the treatments and was becoming a major component of the plant community.

Seedling-sized stems—Inventories conducted prior to clearcutting indicated that there was an overall average density of 365 shrubs and 538 tree seedling-sized stems per hectare (Table 3). There were no significant differences among treatments in the density of shrub and tree seedling-sized stems per hectare prior to overstory removal.

Table 3—Average[a] density of shrub and tree species seedling-sized stems for the harvest treatments, before (1995), one year after (1996) and two years (1997) after clearcut

Species and harvest treatment	1995	1996	1997
	------------ Stems/*ha* ------------		
Shrubs			
Summer/remove	139a	555a	1,181b
Summer/leave	972a	1,181a	5,208a
Winter/remove	69a	349a	3,472ab
Winter/leave	278a	138a	2,361b
Overall avg.	365B	556B	3,056A
Trees			
Summer/remove	69a	0a	208b
Summer/leave	1,806a	833a	2,639a
Winter/remove	69a	139a	2,083a
Winter/leave	208a	139a	347ab
Overall avg.	538B	278B	1,319A

[a] Harvest treatment values by species group within each year with the same lower case letter were not significantly different at the 0.05 level. Overall treatment values among years within each species group with the same upper case letter were not significantly different at the 0.05 level.

One year after clearcutting (1996), the overall density of shrub seedling-sized stems increased from 365 to 556 stems per hectare (Table 3). There was no significant difference among treatments in the density of shrub seedling-sized stems after overstory removal. The overall average density of tree seedling-sized stems decreased from 538 to 278 stems per hectare one year after overstory removal. There were no significant differences among treatments.

Two years after the clearcut treatments (1997), the overall average density of shrub seedling-sized stems increased to 3,056 from 556 stems per hectare, and there were significant differences among treatments (Table 3). The density of tree seedling-sized stems also increased in the second year after treatment. Overall, the average density of tree seedling-sized stems was 1,319 per hectare and there were significant differences among treatments. The Summer/Leave and Winter/Remove treatments had the greatest densities of both shrub and tree seedling-sized stems. Both of these treatments had two replications that bordered existing forest stands and as such may have recruited more individuals.

Over all the harvest treatments, the density of seedling-sized stems of both shrub and tree species were significantly different among inventories. The average density of shrub seedling-sized stems in 1997 (3,056 stems/hectares) was significantly different than in 1996 (556 stems/hectares) and prior to clearcutting (365 stems/hectares). Tree seedling-sized stem average density was also significantly different in 1997 (1,319 stems/hectares) than in 1996 (278 stems/hectares) and prior to the clearcut treatments (538 stems/hectares) (Table 3). Except for the Winter/Remove treatment, there were no marked changes in the densities of shrub and tree seedling-sized stems among the three inventories (Table 3). The Winter/Remove treatment had a substantial increase in both shrub and tree seedling-sized stems in 1997 compared to 1995 and 1996. With this one exception and the overall low densities, there was no strong evidence that any of the four treatments really made a difference in the success of securing a desirable stocked stand.

Sapling-sized stems—Prior to overstory removal, the overall average density of shrub sapling-sized stems was 1,318 per hectare, and there were significant differences among treatments (Table 4). The overall average density of tree sapling-sized stems was 392 stems per hectare and there were no significant differences among treatments.

One year after the clearcut treatments (1996), the overall average density of shrub sapling-sized stems decreased from 1,318 to 309 stems per hectare (Table 4). There were significant differences among the four treatments. The Winter/Remove treatment had a significantly different density of sapling-sized stems (1,004 stems/hectares) than the Summer/Leave, Summer/Remove, and Winter/Leave treatments (8, 191, and 33 stems/hectares, respectively). The overall average tree sapling-sized density also decreased one year after the clearcut treatments from 392

Table 4—Average[a] density of shrub and tree species sapling-sized stems for the harvest treatments, before (1995), one year after (1996) and two years (1997) after clearcut

Species and harvest treatment	1995	1996	1997
	------------ Stems/*ha* ------------		
Shrubs			
Summer/remove	458b	191b	321b
Summer/leave	758b	8b	1,058ab
Winter/remove	2,608a	1,004a	933ab
Winter/leave	1,446ab	33b	2,525a
Overall avg.	1,318A	309B	1,209A
Trees			
Summer/remove	361a	13a	549a
Summer/leave	294a	8a	975a
Winter/remove	349a	8a	935a
Winter/leave	575a	29a	950a
Overall avg.	392B	15C	846A

[a] Harvest treatment values by species group within each year with the same lower case letter were not significantly different at the 0.05 level. Overall treatment values among years within each species group with the same upper case letter were not significantly different at the 0.05 level.

to 15 stems per hectare and there were no significant differences among treatments.

In 1997, two years after clearcutting the average density of shrub sapling-sized stems increased to 1,209 from 309 stems per hectare and there were significant differences among the four treatments (Table 4). The Winter/Leave treatment had a significantly different average density of shrub sapling-sized stems (2,525 stems/hectares) than the Summer/Remove treatment (347 stems/hectares). The Summer/Leave and Winter/Remove treatments were not significantly different from the other two treatments with 1,770 and 933 shrub sapling-sized stems per hectare, respectively. The overall average density of tree sapling-sized stems two years after clearcutting was 846 per hectare, which was more than twice the amount of stems found prior to the clearcut treatments.

There were significant differences among years in the densities of shrub and tree sapling-sized stems (Table 4). The overall average density of shrub sapling-sized stems in 1997 (1,209 stems/hectares) was not significantly different from the average density in 1995, prior to clearcutting (1,318 stems/hectares). The density of shrub sapling-sized stems in 1996 (309 stems/hectares) was significantly different from both the 1995 and 1996 densities. The average density of tree sapling-sized stems in 1997 (846 stems/hectares) was significantly different than in 1996 (15 stems/hectares)and prior to overstory removal (392 stems/hectares).

The density of tree and shrub sapling-sized stems initially decreased after the clearcut because all sapling-sized stems were intentionally removed during harvest. Both the densities of shrub and tree sapling-sized stems increased during the second year of the study, but were still insufficient for stocking the forest stands.

Forb and vine plants prior to clearcutting covered about 30 percent of the ground layer. By the end of two growing seasons, the coverage had increased to nearly 90 percent. Forb plants were frequently > 1 meter tall and vines completely entangled seedlings. The degree to which these plants were influencing the survival and growth of the shrub and tree reproduction was not measured in this study. Over all harvest treatments, there were 1,337 seedling-sized and 846 sapling-sized tree stems per hectare two years after clearcutting. The reproduction contained a greater proportion of black cherry (*Prunus serotina* Ehrh.), red maple (*Acer rubrum* L.) and white ash (*Fraxinus americana* L.), and fewer oak (*Quercus* spp.) and hickory (*Carya* spp.) stems than were present in the pre-treatment overstory. Stump sprouting accounted for very few individuals that occurred in the natural regeneration of the stand. The greater diversity indicated that there was a potential for other tree species to become established and grow in the wastewater conditions, but at lower densities than desirable using either the Marquis and others (1992) or Bowersox and others (1998) standards. The Marquis and others standard requires a minimum of 3,330 seedling-sized or 1,334 sapling-sized stems per hectare to have adequate stocking on 70 percent of the area. The standard that Bowersox and others are using in a model being tested in mixed-oak stands of central Pennsylvania requires a minimum of 25,950 tree seedling-sized or 2,595 tree sapling-sized stems per hectare to recommend clearcutting. Based on these two years of data, it is clear that clearcutting the present stands, without augmentation and possibly vegetation control, will not produce a fully stocked forest stand.

Artificial Regeneration

Green ash, quaking aspen, northern red oak, and pin oak were selected to enhance natural regeneration because of their potential value to wildlife and their capability to respond to the additional water and nutrients. Green ash is a fast-growing species that grows best on fertile, moist, well-drained soils (Kennedy 1990). Quaking aspen, another fast-growing species grows best on well-drained, loamy soils that are high in organic matter, calcium, magnesium, potassium, and nitrogen (Perala 1990). Northern red oak is moderate- to fast-growing and achieves its best growth on deep, well-drained loam to silty-clay loam soils (Sanders 1990). Pin oak is found on poorly-drained alluvial floodplains and river bottoms, and it is also fast-growing (McQuilkin 1990).

The first-year survival of the four planted tree species was excellent (Table 5). Northern red oak had the lowest overall average survival of 91 percent, while green ash had 100 percent survival. Green ash and quaking aspen did not have significantly different survival among the four harvest treatments. There were significant differences among treatments for the survival of both oak species.

Height growth and total height for the first growing season varied among species, and within species among treatments (Table 5). Green ash planted seedlings had good height growth during the first growing season. Overall, green ash seedlings grew 45 centimeters and were 107 centimeters tall at the end of the first growing season (Table 5). Both the Summer/Leave and Summer/Remove treatments had an average height growth of 47 centimeters and were significantly different from the Winter/Leave (41 centimeters) and the Winter/Remove (43 centimeters) treatments. The overall average total height (107 centimeters) of the green ash seedlings was not significantly different among treatments.

Quaking aspen had the greatest average height growth and average total height of all four species planted. Overall, quaking aspen planted seedlings grew 105 centimeters and were 134 centimeters tall after the first growing season. The Summer/Remove treatment had the greatest average height growth (131 centimeters) and total height (158 centimeters) and was significantly different from the other three treatments.

Northern red oak and pin oak did not attain the height growth in the first growing season that the quaking aspen and the green ash did. Height growth was significantly different among treatments for both oak species (Table 5).

Table 5—Average[a] first and second year survival, height growth and total height for planted green ash, quaking aspen, northern red oak and pin oak seedlings, by harvesting treatment

Species	First year Survival	First year Height growth	First year Total height	Second year Survival	Second year Height growth	Second year Total height
	Pct	-------- *Cm* --------		Pct	-------- *Cm* -------	
Green ash						
Summer/leave	100a	47a	109a	99a	44b	165a
Summer/remove	100a	47a	110a	99a	60a	152ab
Winter/leave	100a	41b	106a	99a	56a	144b
Winter/remove	100a	43b	104a	97a	45b	159ab
Overall avg.	100	45	107	98	51	156
Quaking aspen						
Summer/leave	100a	100b	125b	89ab	104b	279a
Summer/remove	98a	131a	158a	92a	128a	215b
Winter/leave	100a	99b	133b	87b	98b	175c
Winter/remove	99a	88b	121b	85b	80c	218b
Overall avg.	99	105	134	88	104	222
Northern red oak						
Summer/leave	93a	13ab	47a	53a	1a	29a
Summer/remove	93a	15a	48a	56a	1a	28a
Winter/leave	87b	12b	50a	41a	-1a	21a
Winter/remove	92a	11b	48a	42a	1a	20a
Overall avg.	91	13	48	48	1	25
Pin oak						
Summer/leave	100a	10a	32a	85a	0a	24b
Summer/remove	99a	10a	29b	71b	2a	28a
Winter/leave	98a	9b	27c	77ab	1a	24b
Winter/remove	96b	8b	28bc	79a	1a	22b
Overall avg.	98	9	29	78	1	25

[a] There were 288 bare-root seedlings per species planted in each of the four harvest treatments. Harvest treatment values within each species and year with the same letter were not significantly different at the 0.05 level.

Northern red oak did not significantly differ in average total height among treatments and the overall average total height was 48 centimeters. Pin oak significantly differed in total height among treatments, but the amount of growth was very low in all treatments, averaging 9 centimeters (Table 5).

At the end of the first growing season, all four species had better height growth (Table 5) in the Summer harvest treatments than in the Winter harvest treatments. There did not seem to be any consistent patterns in height growth between residue treatments (Remove vs. Leave) or in total height among season and residue treatments.

After the second growing season the green ash and quaking aspen continued to have high survival, but there was a dramatic decrease in the survival of the two oak species (Table 5). Overall, green ash had 98 percent survival and quaking aspen had 88 percent survival. Northern red oak had the greatest decline in survival from 91 percent to 48 percent (Table 5). Pin oak survival also declined, from 98 percent to 78 percent at the end of the second growing season.

Green ash and quaking aspen continued to respond with vigorous height growth during the second growing season. Over all harvest treatments, green ash grew an average of 51 centimeters (Table 5). Green ash had significantly different height growth among treatments with the Summer/Remove (60 centimeters) and Winter/Leave (56 centimeters) treatments being significantly different from the Summer/Leave (44 centimeters) and Winter/Remove (45 centimeters) treatments. The overall average total height was 156 centimeters and there were significant differences among treatments.

In the second growing season, quaking aspen had the greatest height growth and obtained the tallest total height of all four species. Overall, quaking aspen grew an average of 104 centimeters and averaged 222 centimeters in height (Table 5). Summer/Leave (104 centimeters) and Winter/Leave (98 centimeters) treatments were significantly different from the Winter/Remove (80 centimeters) treatment, but not from each other. All three of these treatments were significantly different from the Summer/Remove treatment, which had the greatest average height growth of 128 centimeters.

Second-year average height growth and average total height for the two oak species were lower than the averages obtained in the first year of the study (Table 5). Overall average height growth of northern red oak and pin oak was 1 centimeter each, and the overall total height of both species averaged 25 centimeters. Overall, both the northern red oak and pin oak failed to demonstrate acceptable growth in these wastewater conditions. Neither of the oaks would be able to surpass the canopy of the competing forb, grass and vine plants.

At the end of two growing seasons, it was determined that the quaking aspen and green ash seedlings had acceptable height growth and could be used to regenerate the irrigated clearcuts. There does not seem to be any consistent evidence to suggest which treatment may be the best for artificially regenerating the irrigated forest.

White-tailed deer browsing in the first growing and dormant season was less than 10 percent of all planted seedlings. In the spring of the second growing season, before and after bud break, browsing became a frequent occurrence in the fenced and unfenced plots. The number and average two-year total height of the browsed and not-browsed planted seedlings are presented in Table 6. There was a significant difference in the species-specific heights between the seedlings that were not browsed and those that were browsed. The average height of the non-browsed quaking aspen seedlings was 257 centimeters. This was more than 11 times greater than the average height of quaking aspen seedlings that were browsed (22 centimeters). Green ash, northern red oak, and pin oak seedlings that were not browsed had average heights 1.5, 5, and 3 times greater than seedlings that were browsed. The number of browsed green ash and quaking aspen stems were lower than the number of browsed northern red oak and pin oak stems. We believe that the oaks were more frequently browsed mainly because their height made them more accessible to the white-tailed deer.

Based on the silvics of each of the four species, all had the potential for responding favorably to the additional water and nutrients that are being added by the wastewater effluent. However, only the green ash and quaking aspen responded as expected to the effluent treatment applied to the clearcut areas. It is possible that the northern red oak species could not tolerate the additional 5 centimeters of water per week during the growing season. Pin oak, however, should have been able to adjust to the additional water. One explanation for why the oaks did not respond was that they could not adapt to the soil chemistry found at the site. Competition for light was controlled in the first growing season by herbicides and hand-weeding of the plots. Light competition did not seem to be the reason why the two oak species did not exhibit similar growth to the quaking aspen and the green ash during the first growing season. During the second growing season, however, it was possible that the green ash and quaking aspen may have contributed to the decline of the two oak species. By the end of the first growing season, the quaking aspen and the green ash started to overtop the oaks. Another explanation may be that the oaks were more frequently browsed because they were preferred by white-tailed deer, or their heights made them more accessible for browsing.

Table 6—Number and average[a] two-year old total height of green ash, quaking aspen, northern red oak and pin oak seedlings browsed and not browsed during the second growing season

	Browsed		Not-browsed	
Species	Number of seedlings	2-year total height	Number of seedlings	2-year total height
		Cm		*Cm*
Green ash	78	93b	1,057	161a
Quaking aspen	31	22b	979	257a
Northern red oak	203	11b	353	55a
Pin oak	172	11b	723	33a

[a] Total height values within each species with the same letter were not significantly different at the 0.05 level.

SUMMARY AND CONCLUSIONS

Pennsylvania State University has been studying over-land irrigation of treated wastewater as an alternative to discharging into local streams since the 1970's. In the summer of 1995 and winter of 1996 three replications, of four clearcut treatments were administered to assess the potential for the naturally occurring seedling-sized and sapling-sized shrubs and trees to develop young, well-stocked forest stands. An initial inventory of tree and shrub species densities found that natural regeneration was insufficient and that artificial regeneration would be necessary. The treatments units were used to determine the effects of season of harvest (Summer or Winter) and residue (Remove or Leave) on the ground cover, naturally occurring shrubs and trees and artificial regeneration. Two years after the clearcut treatments, forb and vine plants averaged > 90 percent of the existing ground cover. Naturally occurring seedling- and sapling-sized stems of shrub and tree species were insufficient for stocking a healthy forested ecosystem. Two years after clearcutting, the average densities of shrub and tree seedling-sized stems were 3,056 and 1,337 stems per hectare, respectively. The average densities of shrub and tree sapling-sized stems were 1,209 and 846 stems per hectare, respectively. There were some significant differences in densities among treatments, however, there was no consistently better treatment two years after clearcutting. Overall, the densities of tree seedling- and sapling-sized stems ere insufficient to stock a forest stand capable of renovating the wastewater.

Green ash, quaking aspen, northern red oak and pin oak were planted to evaluate their potential survival and growth in the wastewater conditions. At the end of two growing seasons, overall average survival for green ash was 98 percent and quaking aspen was 88 percent. These two species also had overall average total heights of 156 centimeters and 222 centimeters, respectively, after two growing seasons, demonstrating their ability to respond to the additional nutrients and water. The two oak species did not survive and grow as well as the green ash and quaking aspen. After two growing seasons, the oaks were smaller than when originally planted. We believe that green ash and quaking aspen can be planted to regenerate the irrigated clearcuts, augmenting the existing natural regeneration in renewing the forest stands.

Future research studies will focus on the effects of forbs and vines on survival and growth of planted seedlings and securing natural regeneration before removing the overstory.

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REFERENCES

Bowersox, T.W.; Ahlswede, L.M.; Smith, J.L. [In press]. A model to inventory reproduction and predict regeneration success in mixed oak stands of central Pennsylvania. University Park, PA: Agricultural Research Station, Pennsylvania State University.

Brister, G.H.; Schultz, R.C. 1981. The response of a Southern Appalachian forest to waste water irrigation. Journal of Environmental Quality. 10: 148-153.

Epstein, C.M.; Sawhill, G. 1977. Changes in the ground cover of a New Jersey pine barrens woodland resulting from spray irrigation of sewage effluent. Bulletin New Jersey Acad. Sci. 22: 6-11.

Kennedy, Jr., H.E. 1990. *Fraxinus pennsylvanica* Marsh. Green ash. In: Silvics of North America. Agric. Handb. 654. U.S. Department of Agriculture, Forest Service: 348-354. Vol. 2.

Larrick, D.S.; Bowersox, T.W. 1999. Long-term effects of wastewater irrigation on forested ecosystems at Pennsylvania State Game Lands 176. In: Proceedings, 12th central hardwood conference. Gen. Tech. Rep. SRS-24. U.S. Department of Agriculture, Forest Service: 90-97.

Lull, H.W. 1968. A forest atlas of the northeast. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 46 p.

Marquis, D.A.; Ernst, R.L; Stout, S.L. 1992. Prescribing silvicultural treatments in hardwood stands of the Alleghenies. Revised. Gen. Tech. Rep. NE-96. U.S. Department of Agriculture, Forest Service. 102 p.

McQuilkin, R.A. 1990. *Quercus palustris* Muenchh. Pin oak. In: Silvics of North America. Agric. Handb. 654. U.S. Department of Agriculture, Forest Service: 709-714. Vol. 2.

Perala, D.A. 1990. *Populus tremuloides* Michx. Quaking aspen. In: Silvics of North America. U.S. Department of Agriculture, Forest Service: 555-569. Vol. 2.

Sanders, I.L. 1990. *Quercus rubra* L. Northern red oak. In: Silvics of North America. Agric. Handb. 654. U.S. Department of Agriculture, Forest Service: 727-733. Vol. 2.

Sopper, W.E.; Kardos L.T., eds. 1973. Recycling treated municipal wastewater and sludge through forest and cropland. University Park, PA: The Pennsylvania State University Press. 479 p.

Storm, G.L. 1995. Wildlife investigations. In: 1995 forecast of loading report. University Park, PA: Office of Physical Plants, the Pennsylvania State University: 501-529.

NATIVE HIGH VALUE TREE RECLAMATION ON SURFACE MINED SPOILS IN EASTERN KENTUCKY

William R. Thomas, Matthew H. Pelkki, and James M. Ringe[1]

Abstract—Grading standards on reclaimed coal mines as outlined in PL 95-87, the Surface Mining Control and Reclamation Act of 1977, have resulted in spoil compaction that severely limits tree root penetration. This leads to high tree seedling mortality. A field study was established on surface mined lands in eastern Kentucky to study the effects of two levels of compaction (no compaction and compacted) and two soil amendments (bark mulch and barn straw mulch) on three native high value tree species: white ash (*Fraxinus americana*), yellow-poplar (*Liriodendron tulipifera*), and northern red oak (*Quercus rubra*). Bark mulch seemed to have little effect on seedling vigor and survivorship and the barn straw seemed to lower both vigor and survivorship. Bulk densities for each compaction level were significantly different. Results are based on one year of data, in the future we hope to answer the question, will the uncompacted plots result in higher vigor and survivorship than the compacted plots, and that is our hypothesis.

INTRODUCTION

Coal mining has disturbed over 2.4 million ha in the United States since 1930 and the majority of that land was originally forested in Appalachia (Zeleznik and Skousen 1996). Often the best post mining land use is to return the lands to forests as quickly as possible. There is much interest in reclaiming mined lands with high value trees, especially if economical methods can be found. Reclamation costs could be significantly reduced if a method is developed to successfully establish trees and grasses (Larson and Vimmerstedt 1990).

In order to achieve productive forests on post mined lands, reclamation techniques that ensure good tree survivability and growth must be adopted (Campbell 1997). One of the key problems in establishing a productive forest on previously surfaced mined lands is excessive compaction, which leads to high tree seedling mortality. Chaney et al. (1995) believe that the initial condition of mined sites reconstructed under specifications of PL 95-87 result in a growth media that is less than optimal for high-value species of hardwood trees.

Reclamation research in the midwest has shown that pre-federal law mining sites resulted in some of the most productive areas in the region for the growth of tree seedlings (Ashby et al., 1978). A study by Burger and Torbert (1992) has shown that when mine spoil is loosely dumped without grading and planted with various species of trees, survival is higher than that on graded mine spoil. The problem of soil compaction has been approached in two ways: (1) prevent or minimize occurrence of the problem and (2) ameliorate the problem once it has happened (Sweigard, 1990). The former is preferable to the later because it conserves resources.

The two objectives of this study were to 1) investigate the effects of two levels of compaction on seedling survival of three native high value tree species, and 2) investigate the influence of two soil amendments on seedling survival of three native high value tree species. This paper reports first-year results and serves as baseline data for an ongoing study.

METHODS AND MATERIALS

Study Area

The study was conducted on a surface mine in eastern Perry County, Kentucky (37° 24' N, 83° 8' W). This mine is located in Kentucky's eastern coalfield in the Cumberland Plateau physiographic region. Climate is temperate humid continental with average annual precipitation of 117 cm, and an average monthly precipitation of 10 cm, which ranges from 6-12 cm. Average temperature is 13°, with a mean daily maximum and minimum of 31° and 18° in July and 8° and -4° in January (Hill 1976). The mine is within the Hazard Coal Reserve District as delineated by the U.S. Geological survey (Huddle et al. 1963).

Plot Construction

Six enclosed 1 ha plots, approximately 70 m wide and 155 m long, were constructed (Figure 1). Two of the plots (#7 and #8) were constructed in time for the 1996 planting season. Due to construction delays the remaining four plots were not ready until the 1997 planting season. Bulldozers were used to create the borders of the plots, which were required for a hydrologic study, earth moving equipment was then used to loose dump the spoil material in consecutive piles until the entire plot was filled. Three of the plots (#2,#3,#4) were left in the loose condition and represented the no compaction treatment. The other three plots (#7,#8,#9) were driven over repeatedly with a bulldozer and represented the compacted treatment, these compacted plots were constructed to industry compaction levels. When reestablishing forests in this region it is important to ensure adequate ground cover for erosion

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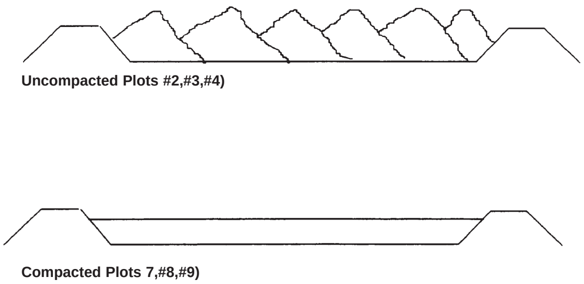

Figure 1—Cross-sectional diagram of plots. Plots are 70 m wide and 155 m long, borders of plots are 2-3 m high.

control; however if inappropriate herbaceous species are chosen, tree seedling survival will be reduced due to excessive competition. All plots were seeded with a slow establishing non-aggressive mixture of grasses and legumes. The grass and legume mixture consisted of annual rye (*Secale cereale*), perennial rye (*Lolium perenne*), orchard grass (*Dactylis glomerata*), birdsfoot trefoil (*Lotus corniculatus*), and Appalow lespedeza (*Serecia lespedeza*, var. Appalow) at the following rates of application 33.61 kg/ha for the annual rye and 5.61 kg/ha for each of the other species. Bark mulch was applied to two plots (#3,#8) at the rate of 125 ton/ha. Barn straw mulch was applied to two plots (#4,#7) at the rate of 125 ton/ha. The remaining plots (#2,#9) received no soil amendments.

Tree Species Selection

Tree species selection for our project was based on species that are native to the area, had documented performance in land reclamation, and a potential commercial value. White ash was chosen because it is a high value native tree species that has shown high survivability on surfaced mined lands (Zeleznik and Skousen 1996, Medvick 1969). Yellow-poplar was chosen because it is native to the area, very fast growing, and there is a rapidly growing market for this species. Northern red oak was chosen because it is a native species that has a high value.

Planting and Measuring of Tree Seedlings

Twenty-one subplots were established inside each of the plots. These subplots were 20 m x 20 m and one corner was permanently marked with rebar and metal tags identifying subplot number and species planted within the subplot. Each of the three species were randomly allotted to three subplots within each plot. The remaining twelve subplots in each plot were allotted to four other tree species not included in this study. Tree seedlings (1-0) were purchased from the Kentucky Division of Forestry's tree nursery in Morgan County, Kentucky. Tree seedlings were planted on a 1.8 m x 1.8 m spacing, yielding 121 trees per subplot. Two of the plots (#7,#8) were planted in the spring of 1996 and the remaining plots were planted in the spring of 1997. All tree seedlings were planted by professional tree planters. Annual measurements were made from the end of July through mid August. Total height of each tree was recorded to the nearest cm. In addition to total height, each tree was assigned a vigor class. Vigor classes were developed by the authors and consisted of five classes: 0-tree was dead, 1-tree was nearly dead, 2-intermediate/ stressed, 3-vigorous with one or two signs of stress, and 4-vigorous with no signs of stress. Signs of stress included chlorotic, withered, or misshapen foliage. Tree survival percentages were calculated by dividing the number of live tree seedlings measured in each plot by the number of trees originally planted in the plots. In addition to survivorship percentages, vigor rankings were characterized by the mode and mean of each plot.

Bulk Density

Bulk density is defined as the mass of a unit volume of soil (Brady 1990). There is a direct relation between bulk density and compaction level of the mine spoil. Bulk density was measured by a technique developed by Muller and Hamilton (1992). This technique consists of excavating a hole (approx. 1000 cm^3) in the mine spoil and carefully saving and weighing all material. The hole is then filled with expanding polyurethane foam, which can be obtained from most home improvement or building stores, cardboard is then secured on top of the hole to ensure that the foam fills all crevices in the hole. Curing of the foam takes 8 hrs, and after curing the foam is removed from the hole. Loose debris is then washed from the foam, and any foam that was not actually in the hole is removed. Spoil material was sieved using a 2mm sieve, weights of the sieved spoil material and the spoil material that would not pass through the 2mm sieve were recorded. This reveals the rock fragment content of the spoil material. Volume of the hole is then determined by displacing water with the foam. Seven subplots were randomly selected within each plot and sampled for their bulk density. To test for significant differences among compaction levels, a general linear model procedure with least significant difference t-tests of the SAS System (1996) was used.

Spoil Analysis

Characterization of the spoil materials chemical properties were obtained through spoil analysis. A composite sample was taken from five systematic sub-samples on each subplot. Care was taken to ensure that no soil amendments were included in the spoil analysis. Samples were submitted for analysis of ten parameters, which included organic matter, phosphorus, potassium, calcium, magnesium, pH, nitrate nitrogen, soluble salts, water holding capacity , and total nitrogen. There were a total of nine composite samples from each plot; results of spoil analysis were averaged for each plot.

RESULTS AND DISCUSSION

Because tree seedlings age varied, analysis was limited to reporting percentages and averages. Northern red oak preformed well on all plots (Table 1) except for plot #8 which was compacted and had barn straw added, this plot is also one year older. White ash preformed equally well on the uncompacted and compacted spoils; however yellow-poplar did better on the plots that were one year old as compared to the two year old plots. Mortality was highest on plots #7 and #8, where the seedlings were 1 year older than in the other plots. The big question is, how will the

Table 1—Tree survival on surface mined lands in eastern Kentucky, treated with two levels of compaction and two soil amendments

Plot	White ash	Yellow-poplar	Northern red oak
	Pct survival		
Measured in 1st growing season only			
2 Uncompacted/no mulch	89	90	93
3 Uncompacted/bark mulch	92	95	100
4 Uncompacted/barn straw	99	94	97
9 Compacted/no mulch	95	98	100
Measured in 2nd growing season only			
7 Compacted/barn straw	82	15	52
8 Compacted/bark mulch	97	41	96

one year old tree seedlings do next year, and will the compacted plots look similar to the uncompacted plots?

Compaction levels as characterized by bulk densities were significantly different at p=.05 (Table 2). Overall bulk density was 1.4 Mg/m^3 on the uncompacted spoils and 2.3 Mg/m^3 on the compacted spoils. Bulk density within the uncompacted plots was not different at the p=.05 level of significance, within the compacted level. Plots #7 and #8 were not different from each other. Rock fragment content ranged from 50.3 percent to 54.5 percent on the uncompacted plots and from 55.9 percent to 71.1 percent on the compacted plots (Table 3).

Nine composite samples were submitted for analysis of spoil chemical properties of each plot. Composite samples results were then averaged for the entire plot. Table 4 reports the average spoil characteristics of each plot.

Plot #7 had the lowest mean vigor (1.18) of all the plots; however it is evident that vigor was lowest on the 2 year old seedlings as compared to the 1 year old seedlings. White ash preformed better on the one year old plot (Table 5). Yellow-poplar performed much better on the one year old plots than on the two year old plots (Table 6). The same pattern held true for northern red oak (Table 7) with performance better on the one year old plots as compared to the two year old plots. Overall vigor of the tree seedlings was highest on the one year old plots (Table 8).

Plots that were treated with soil amendments had lower overall vigor for all species than those plots left untreated. Plots treated with bark mulch fared better than those treated with the barn straw mulch. The barn straw mulch plots seem to support a much hardier ground cover than the bark mulch plots. This could have caused increased competition on the tree seedlings from the ground cover species.

Table 2—Bulk density (Mg/m^3) values on uncompacted and compacted plots on surface mined lands in eastern Kentucky

Plot	Bulk density
	*Mg/m^3**
Uncompacted	
2	1.5^a
3	1.3^a
4	1.4^a
Overall mean**	1.4^1
Compacted	
7	2.1^a
8	2.0^a
9	2.7^b
Overall mean**	2.3^2

*Bulk density values within a compaction level followed by the same letter are not significantly different at P=.05.
**Overall means for a compaction level followed by the same number are not significantly different at P=.05.

Table 3—Average rock fragment content of spoil material on surface mined lands in eastern Kentucky

Plot	Avg. weight of spoil material <2mm	Avg. weight of spoil material >2mm	Average rock fragment
	g		*Pct*
2	852.92	864.22	50.3
3	1194	1431.3	54.5
4	1088	1267.4	53.8
7	890.09	1884.64	67.9
8	655.3	1613	71.1
9	707.92	898.85	55.9

Table 4—Average spoil characteristics of plots on surface mined lands in eastern Kentucky

	Plot					
	Uncompacted			Compacted		
Parameter	2	3	4	7	8	9
Organic matter (%)	1.8	2.0	3.2	4.0	4.7	2.8
Phosphorus (ppm)	0.0	0.0	0.0	0.3	0.3	0.3
Potassium (ppm)	4	6	13	14	11	11
Calcium (ppm)	56	46	79	48	60	80
Magnesium (ppm)	23	19	30	25	26	48
pH	5.9	5.2	5.8	6.9	7.5	6.9
Nitrate nitrogen (ppm)	25	4	7	11	35	38
Soluble salts (mmhos/cm)	0.4	0.4	0.6	0.4	0.5	0.7
Water holding capacity(%)	9.8	9.5	10.6	13.6	13.8	12.7
Total nitrogen (lb/acre)	624	653	1166	2034	1574	1107

Table 5—Vigor of white ash seedlings planted on surface mined lands in eastern Kentucky

Plot	N	Mode	Mean
Measured in 1st growing season only			
2 Uncompacted/no mulch	323	2	2.39
3 Uncompacted/bark mulch	333	2	2.24
4 Uncompacted/barn straw	375	2	2.24
9 Compacted/no mulch	345	3	2.49
Measured in 2nd growing season only			
7 Compacted/barn straw	364	2	1.69
8 Compacted/bark mulch	357	2	2.09

Note: vigor consisted of five classes: 0-dead, 1-nearly dead, 2-intermediate/stressed, 3-vigorous, slight stress present, 4-vigorous with no signs of stress.

Table 6—Vigor of yellow-poplar seedlings planted on surface mined lands in eastern Kentucky

Plot	N	Mode	Mean
Measured in 1st growing season only			
2 Uncompacted/no mulch	326	2	2.24
3 Uncompacted/bark mulch	346	2	2.14
4 Uncompacted/barn straw	340	2	2.13
9 Compacted/no mulch	355	2	2.25
Measured in 2nd growing season only			
7 Compacted/barn straw	363	0	0.32
8 Compacted/bark mulch	363	2	1.38

Note: vigor consisted of five classes: 0-dead, 1-nearly dead, 2-intermediate/stressed, 3-vigorous, slight stress present, 4-vigorous with no signs of stress.

CONCLUSION

This study reports first year data on establishment of native high-value tree seedlings on surface mined lands in eastern Kentucky. Current law requires mine operators to grade the surface to the approximate original contour which results in the soil being compacted to the point where seedling survival is severely hindered. Due to the fact that tree seedlings age varied we are unable to compare the compacted plots and uncompacted plots treated with soil amendments, and the two plots that had no soil amendments (compacted & uncompacted) had similar vigor and survivorship. The real question is will, once data has been collected for several years, the uncompacted plots result in higher vigor and survivorship than the compacted plots, and that is our hypothesis. If forestry is the post mining land use then bond release does not occur until five years after the last reclamation activity. Bark mulch seemed to have little effect on seedling vigor and survivorship and the barn straw seemed to lower both vigor and survivorship. In the future we hope to demonstrate that seedling survival will be least successful on compacted plots suggesting that a viable alternative to current reclamation practices exists; avoiding compaction by

Table 7—Vigor of northern red oak seedlings planted on surface mined lands in eastern Kentucky

Plot	N	Mode	Mean
Measured in 1st growing season only			
2 Uncompacted/no mulch	338	2	2.32
3 Uncompacted/bark mulch	363	2	2.23
4 Uncompacted/barn straw	353	2	2.22
9 Compacted/no mulch	363	2	2.08
Measured in 2nd growing season only			
7 Compacted/barn straw	363	0	1.05
8 Compacted/bark mulch	362	2	1.93

Note: vigor consisted of five classes: 0-dead, 1-nearly dead, 2-intermediate/stressed, 3-vigorous, slight stress present, 4-vigorous with no signs of stress.

Table 8—Vigor of white ash, yellow-poplar, and northern red oak tree seedlings planted on surface mined lands in eastern Kentucky

Plot	N	Mode	Mean
Measured in 1st growing season only			
2 Uncompacted/no mulch	987	2	2.32
3 Uncompacted/bark mulch	1042	2	2.20
4 Uncompacted/barn straw	1068	2	2.20
9 Compacted/no mulch	1063	2	2.27
Measured in 2nd growing season only			
7 Compacted/barn straw	1090	0	1.18
8 Compacted/bark mulch	1082	2	1.80

Note: vigor consisted of five classes: 0-dead, 1-nearly dead, 2-intermediate/stressed, 3-vigorous, slight stress present, 4-vigorous with no signs of stress.

minimizing grading. This will result in reclamation that is not only cheaper but will allow quicker establishment of productive forest lands. A limitation of this type of reclamation is that it requires relatively flat land, and while appropriate for mountaintop removal is of limited use on contour mines.

REFERENCES

Ashby, C.W.; Kolar, C.A.; Guerke, M.L. [and others]. 1978. Our reclamation future with trees, coal extraction and utilization. Research Center, Southern Illinois at Carbondale. 101 p.

Brady, N.C. 1990. The nature and property of soils. New York: Macmillan. 103 p.

Burger, J.A.; Torbert, J.L. 1992. Restoring forests on surface-mined lands. Publ.: 460-123. Virginia Cooperative Extension Service; Virginia Polytechnic Institute and State University. 16 p.

Campbell, C. 1997. Reforestation initiative. Reclam. Advis. Memo. 124. Frankfort, KY: Kentucky Department for Surface Mining Reclamation and Enforcement. 11 p.

Chaney, W.R.; Pope, P.E.; Byrnes, W.R. 1995. Tree survival and growth on land reclaimed in accord with Public Law 95-87. Journal of Environmental Quality 24: 630.

Hill, J.D. 1976. Climate of Kentucky. Kentucky Agric. Exper. Stn. Prog. Rep. 221.

Huddle, J.W.; Lyons, F.J.; Smith, H.L.; Ferm, J.C. 1963. Coal reserves of eastern Kentucky. U.S. Geol. Surv. Bull. 1120. 147 p.

Larson, M.M.; Vimmerstedt, J.P. 1990. Effects of mine soil and seeded herbaceous species on survival of planted trees. Internat. J. of Sur. Min. and Recla. 4: 53.

Medvick, C. 1969. Selecting plant species for revegetating surface coal mined lands in Indiana, a forty-year record. Mid-West Coal Producers Institute. [As presented in Papers on Reclamation Hutnik]. 1440: 65.

Muller, R.N.; Hamilton, M.E. 1992. A simple, effective method for determining the bulk density of stony soils. Commun. Soil Sci. Plant Anal. 23(3&4): 313-319.

SAS Institute Inc. 1996. The SAS system for windows. Version 6.12. Cary, NC.

Sweigard, R.J. 1990. Reclamation of surface-mined lands for agricultural use. Internat. J. of Sur. Min. and Recla. 4: 131.

Zeleznik, J.D.; Skousen, J.G. 1996. Survival of three tree species on old reclaimed surface mines in Ohio. Journal of Environmental Quality. 25: 1429.

GROWTH OF WHITE AND RED OAK SEEDLINGS AND SEED ON MINED UNGRADED CAST OVERBURDEN

W. Clark Ashby[1]

Abstract—Tree seedlings and/or seed of white oak (*Quercus alba* L.) and red oak (*Q. rubra* L.) were planted spring and fall from 1978 to 1982 on ungraded cast overburden (also termed spoil banks) in southern Illinois. Each planting season there were 50 seedlings, or seed spots, of each species per row in replicated plots. Some seasons no seedlings or seed were available for planting. The 20 plots had 1800 white oak and 1700 red oak planting spots total. The oak seedling and seed rows were randomly assigned with other species in a plot so that nearest neighbor tree rows differed from plot to plot. Soil bulk density averaged 1.3 grams per cubic centimeter, pH 7.5, and plant nutrients were low by agricultural standards. The ground cover was old-field weeds that were sprayed with herbicide in later planting years.

Trees in each row were counted 3 years after planting to determine establishment, that varied widely. In 1993-1994 all trees were measured for height and diameter breast height (DBH). Ages then ranged from 11 to 16 years. Survival after establishment was generally good, somewhat greater for red than white oak, for spring- than fall-planted trees of each species, and for trees planted as seedlings rather than seed. Heights of older spring-planted seedlings averaged more than 6 meters with DBHs, not here reported, more than 8 centimeters. Red oak tended to be taller than white oak. DBHs were more variable. Heights and DBHs of trees planted as seedlings tended to be greater than those planted as seed. Variations in survival and growth were not clearly attributable to quality of planting stock, use of herbicides, or differences in randomized nearest-row neighbors. Holding white oak seedlings in a cold room from spring over summer for fall planting gave reduced survival and equivalent growth.

INTRODUCTION

White and northern red oak are major components of forests in Illinois and other midwestern states (Burns and Honkala 1990). The natural importance of white oak is greater on mid to upper slopes, and of northern red, here called red, oak is greater in ravines and on lower slopes, related to aspect. These two high-quality species seem logical choices to use in reforestation of minesoils in the lower midwest. In turn, their growth on a rooting medium that does not occur naturally in the local climate can add to understanding of their ecological life requirements.

Within the past 50 years thousands of hectares in the midwest have been surface mined for coal and planted to millions of trees. In a pre-law period roughly from 1930 to 1960 in Illinois, 8.7 million hardwoods and 7.2 million conifers were voluntarily planted by coal companies and coal associations in cooperation with the Illinois Department of Conservation, the USDA Forest Service, and a few universities (Ashby and others 1978). Although scant records exist of the performance of these plantings other than the Forest Service and university research plots, overall tree performance was considered to be good to excellent, with failures of non-adapted species planted on isolated acidic areas.

A quarter million of the 8.7 million hardwoods in Illinois were white or bur (*Q. macrocarpa* Michx.) oak, and half a million were red or black (*Q. velutina* Lam.) oak. A 50-year-old white oak stand growing on ungraded cast overburden in southern Illinois had excellent survival, height of 26±3 meters, and a DBH of 23±1 centimeters (Ashby 1996). A near-by stand of red (intermixed with *Q. shumardii Buckley*) oak 55 years old also had excellent survival, average height of 33±3 meters, and DBH of 35±1 centimeters.

Many experienced reclamation, and an increasing number of regulatory, personnel have been concerned that trees planted in the 20 years since passage of a federal 1977 reclamation law have not survived and grown well compared to those planted pre-law (Ashby 1991). Another problem is that species with high 5-year survival rates selected for bond release commonly are not ecologically or economically desirable for long-term forestry plantings.

Viewing tree planting on post-law minesoils as a lost cause has unfortunately become a common misconception. State and federal regulatory authorities are now recognizing the need for re-evaluating out-dated regulations, with increased interest in successful reforestation. Even the well-documented effects of soil compaction from mandatory grading after mining (Josiah 1986) need not detract from the utility of studies of tree growth on minesoils. The adverse ecological and economic consequences of grading to parking-lot standards have been demonstrated. Mined lands can be reclaimed with minimal compaction using new grading methods and understanding. A great opportunity for restoring mined lands to productive forestry should not be neglected.

My research group in 1978 set up studies to document carefully the survival and growth of 25 tree species planted on ungraded cast overburden, including many of the hardwood species grown earlier. This paper reports

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performance of white and red oak, planted as seedlings, and as seed, if available. Plots replicated once were established each spring and fall from 1978 to 1982. Several survival counts and growth measurements were taken in later years. The goals of this paper include:

1. To evaluate and compare relative performance of white and red oak planted as seedlings or seed during fall and spring for 5 years on ungraded cast overburden.
2. To compare tree performance when planted fall or spring.
3. To compare tree performance when planted as seedlings or seed.
4. To evaluate white oak performance using other kinds of growing stock.

METHODS AND MATERIALS

My study area is on the Sahara Coal Company, Inc. surface mine No. 6 just west of Harrisburg in southeastern Illinois. Annual temperature averages 13 degrees Centigrade, with July 25 degrees and January 0 degrees Centigrade, and annual precipitation 1100 millimeters. Figure 1 has the growing season precipitation from April through September 1978 to 1984, years of tree establishment and early growth. Monthly temperatures for this period were tabulated and are not here listed.

Rooting Medium

The predominant regional pre-mining soil was Hosmer silt loam, Typic Fragiudalfs Alfisols (Miles and Weiss 1978). A geological column taken on the permit area in 1981 prior to mining was described in an unpublished report by W.C. Hood. He reported 4 meters of unconsolidated material overlying numerous intermixed layers of sandy shale (9 meters total), limestone (3 meters), and sandstone (1 meter) above a 1.5 meter-thick seam of No. 6 coal. The rooting medium of the spoil banks cast by a Marion 5761 power shovel was thus a mixture of soil fines and coarse fragments from shattered rock layers above the coal. The slopes of the bank ridges trending southeast-northwest ranged from 21 to 34 degrees and averaged 29 degrees. The ridges were spaced from 8 to 30 meters apart with an average distance of 20 meters. Bank heights ranged from 3 to 10 meters with an average height of 6 meters. Flooded areas between the banks were not found.

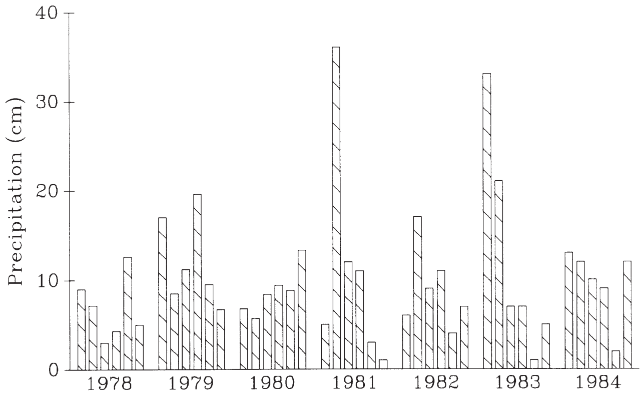

Figure 1—Growing-season (April-September) monthly precipitation from 1978 to 1984 recorded at Harrisburg, IL.

The bulk density of the cast overburden corrected for rock fragments ranged from 1.22-1.35 grams per cubic centimeter and the percent coarse fragments larger than 2 millimeters from 38-52 percent. Stones of varying sizes exposed on the surface functioned as an erosion pavement and provided numerous niches or micro-habitats for plant establishment and growth. Numerous excavations of cast overburden in later years showed deep rooting to 4 meters by various species with extensive branching of root systems. Roots also penetrated shale and sandstone fragments as they broke apart when exposed to near-surface weathering agents. The minesoil likely had a more favorable moisture regime for growth of deep-rooted plants than the fine-textured, fragipan native soils.

The post-mining rooting medium based on soil fines was classified as a silty clay loam to clay loam, probably related to the well-developed B horizon of the pre-mining fragiudalfs and to rapid weathering of the predominant shales of the overburden in the years between mining and tree planting. The rooting medium was mildly alkaline (pH 7.5) with average Bray-1 phosphorus 5 parts per million, Bray-2 phosphorus 24 parts per million, potassium 154 parts per million, magnesium 175 parts per million, and calcium 2313 parts per million (Josiah 1986). Variations among the individual measurements of these properties on the total 14-hectare plot area likely came from differential mixing of overburden materials in the mining process. The ground cover at planting ranged from sparse goldenrod (*Solidago L.* spp.) or broomsedge (*Andropogon virginicus* L.) and other old-field invaders to relatively dense patches of trumpet creeper (*Campsis radicans* (L.) Seemann.) or Japanese honeysuckle (*Lonicera japonica* Thunb.).

Planting Stock and Plot Layout

All seedlings were graded in the laboratory by basal caliper and the smallest ones rejected. Depending on stock quality and numbers available, the largest seedlings and those with carrot-type roots were also rejected. Numbers of lateral roots were not counted. Roots were trimmed to fit in a planting hole without distortion. Representative-sized seedlings for each row were bundled, 52 each, with roots packed in moist sphagnum moss in a plastic bag tied off around the lower stems. Acorns when collected were floated in water for approximately an hour and those undamaged that sank were similarly placed in moist sphagnum moss as recommended by Bonner (1993). Seedlings and seed were stored in a 5 degrees Centigrade cold room until planted.

Seedlings were planted with planting bars (dibbles). Seed was planted with a mattock, three acorns in a hole three times their diameter in depth. Each planting spot was marked with a color-coded pot label that aided in following a row for early survival counts on rough terrain and occasional thick vegetation.

Each replicated tree plot of this study had 20 randomly-assigned, single-species rows that included rows of white and red oak. Thus a red oak seedling row in one randomized plot was planted between a row of walnut (*Juglans nigra* L.) and of pin oak (*Q. palustris* Muenchh.), and in the paired plot was planted between a row of baldcypress (*Taxodium distichum* (L). Rich.) and of red oak from seed. An oak row was planted all to seedlings or all to acorns. Each row had 50 planting spots with 2.5 meters spacing between trees and rows. The replicated plots had 100 of each available oak species/planting stock planted each season. The rows ran primarily up and down across the banks rather than along the length of a bank.

White oak—The sum of the areas of the rows of white oak seedlings or seed spots planted both spring and fall from 1978 to 1982 was 1.12 hectares with 1800 trees. Quality of white oak nursery stock as received each year was rated as good to excellent except for root and stem damage noted for the 1980 seedlings planted fall 1980 and spring 1981. Limitations to growth of these seedlings were not evident in the 1993 measurements. Seedlings were donated from the Illinois Union County state tree nursery 100 kilometers southwest of the study site, except for fall 1981. The only seedlings planted fall 1981 were donated from the Licking, MO state nursery 200 kilometers further west. The Missouri seedlings were planted again for comparison with Illinois seedlings spring 1982. In other comparisons white oak seedlings were held in cold storage over a summer each for planting in fall 1979, and in fall 1980. Roots of the seedlings held over were dipped in a 1 percent Captan solution before placing in cold storage.

White oak seed was planted only in the fall each year. We were not successful in holding over a sufficient number of white oak acorns for spring planting without germination and growth of radicles that were broken in handling. The amount of local seed we collected each fall varied greatly; some years only one tree we found had a good crop and other years many trees had acorns. All the white oak seed planted was rated excellent. It was not unusual to have acorns already germinating at fall planting.

Red oak—The sum of the areas of the rows of red oak seedlings or seed spots was 1.06 hectares planted to 1700 trees. Seedlings were donated by the Illinois state nursery. Quality of red oak nursery stock was rated fair for fall 1979, the season with the lowest later field survival, and otherwise good to excellent. We dug at the nursery the fall-lifted red oak seedlings for the test plantings. In fall 1978, 1979, 1980, and 1982 many seedlings when dug were still in leaf or even actively growing.

Red oak annual acorn crops varied substantially. If available they were planted both fall and spring. Missing seasons were fall 1978 and 1979, and spring 1980. Acorns were treated the same as white oak acorns, and for spring planting were stratified over winter in cold storage. The red oak seed planted was rated good to excellent. Germination after stratification was typically good, and ranged from 50 percent to over 90 percent in several seasons.

Herbicide Applications and Tree Counts and Measurements

Starting in spring 1981 all seedling and seed planting spots that had been planted either the previous fall or in the spring were sprayed in a 1.5 meter circle before tree growth began using a backpack sprayer with a mixture of glyphosate (Roundup®) and simazine (Princep®). Herbicide control was effective in the first year, and often not observable by the fourth year after planting. No further cultural practices were carried out.

All trees were counted for establishment the first summer after planting. If there was more than one seedling per seed spot, those of lesser vigor were clipped off at ground level. All trees were also counted and measured for height and basal caliper in year 3. As an example those planted in fall 1978 and spring 1979 were both measured in summer 1981. The number of trees established after 3 years was used to calculate the subsequent percent survival in 1993 shown in table 1. All trees were measured for height in 1988, and for height and diameter breast height (DBH), if that tall, in 1993 (Ashby and others 1995). Statistical methods were analysis of variance for heights and DBHs of several groupings of trees. Acorn production, animal damage if any, and other observations were recorded at each measurement and occasionally at other times.

RESULTS AND DISCUSSION

Relative Performance of White and Red Oak

Number of trees in 1993 of the 11- to 16-year-old Illinois white oak was 411 of 1800 planted (23 percent) and of red oak was 728 of 1700 planted (43 percent). Most of the losses of trees by 1993 took place in the first years after planting. Mortality from year 3 to 1993 was much lower, especially for red oak (table 1). Mortality was commonly several trees in a gap and canopy closure was found where trees were contiguous. If these white or red oaks had been planted solidly in a stand, a full forest canopy as found in pre-law plantings could be expected, even though higher planting rates would be needed for bond release.

Table 1—Percent of fall- and spring-planted white and red oak established at age 3 that survived to age 11-16 years when planted as seedlings or as seed

Season planted	White oak	Red oak
	------------- %	----------------
Seedlings		
Fall	85	98
Spring	79	95
Seed		
Fall	62	89
Spring	-	88

Tree form was good where trees were contiguous. Red oak would be the better choice for planting based on survival after establishment.

Irregularities in survival from year to year were found for both species (fig. 2). Some of the variability likely was related to undetected variation in planting-stock quality. Planting was carefully supervised with the planting crew supervisors and many of the planters the same from year to year. An important time-related factor that likely affected survival was weather conditions among the several planting seasons. April, May, and total precipitation were low in 1980, with only September at the end of the growing season relatively wet (figure 1). July, August, and September were so hot and dry that many corn fields were not harvested. The white oak seedlings that established in spring 1980 had relatively low survival at age 14. White oak growth was, however, not evidently suppressed (fig. 3), and the 14-year-old red oak trees were essentially equal in height to the 15- and 16-year-old trees (fig. 4).

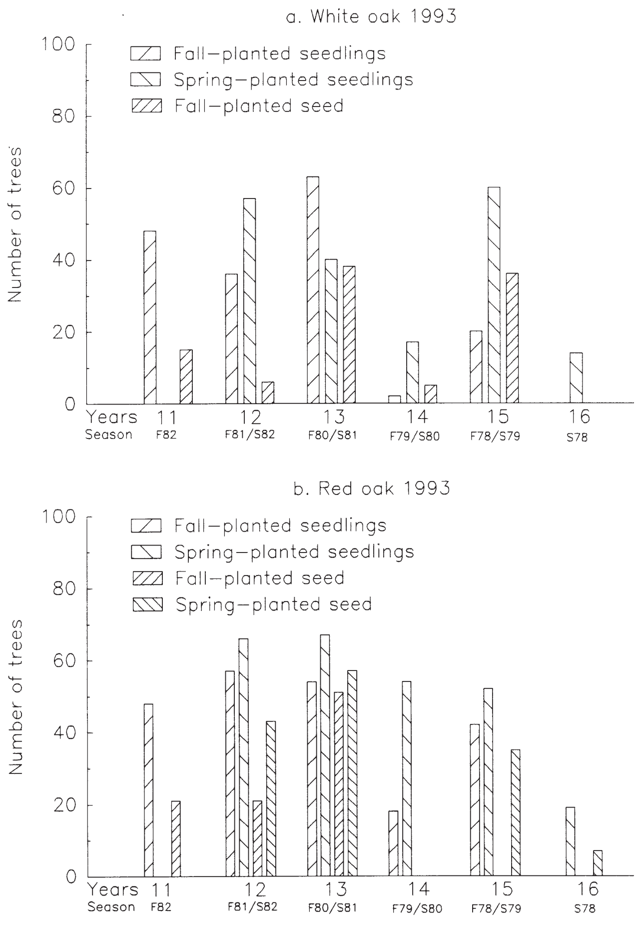

Figure 2—a. Average number of survivors in 1993 of 11- to 16-year-old white oak planted in fall or spring as seedlings or as seed. Each column is based on 100 seedlings or seed spots in 2 rows of 50 each planted per season.
b. Average number of survivors in 1993 of 11- to 16-year-old red oak planted in fall and spring as seedlings and/or as seed.

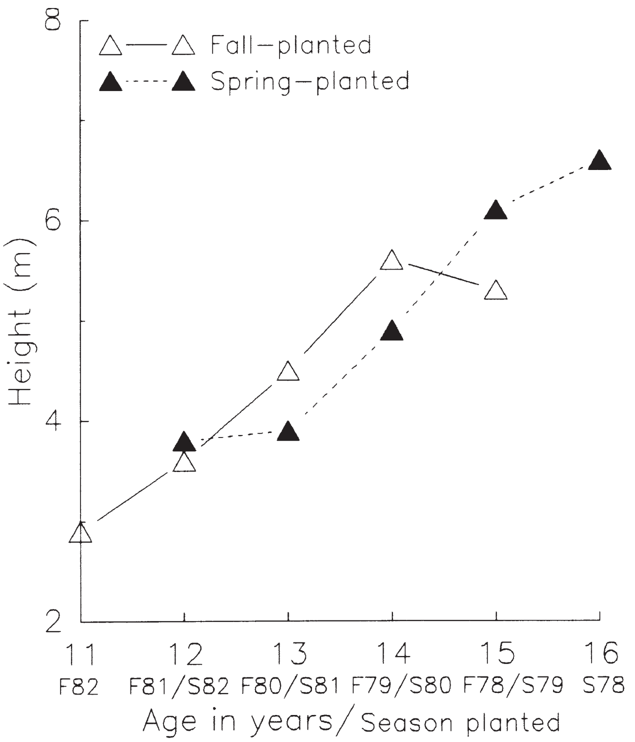

Figure 3—Average height in 1993 of 11- to 16-year-old white oak planted in fall or spring as seedlings or as seed.

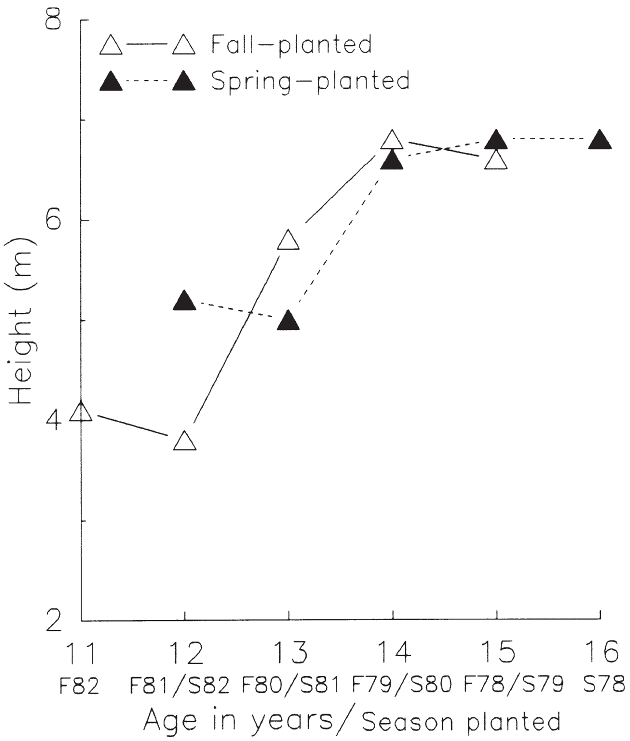

Figure 4—Average height in 1993 of 11- to 16-year-old red oak planted in fall or spring as seedlings or as seed.

A management change to application of herbicides happened only once. If no herbicide applications for trees ages 14, 15, or 16 years versus herbicide applications for trees ages 11, 12, and 13 years increased survival, it corresponded with greater expected survival of younger trees and did not seem to have done away with year-to-year variability.

Possible changes on a seasonal or yearly basis of spatial variation in properties of the successive plot areas planted on cast overburden for 5 years, or in competition from randomly assigned neighbors in the several plots, cannot be ruled out. Overall variation in topography or in observed spoil characteristics at the scale of an 0.7 hectare plot was not evidently synchronous with seasonal differences in survival and growth. The volunteer vegetation present seemed similarly heterogenous throughout the study area.

All the nearest neighbor species to each white and each red oak row were plotted. Neighbors were compared for oaks that had unusually low survival or growth performance in a given season. Any neighbor species consistently in common were, however, not found. For the seven lowest survival plantings, having one-fifth or fewer surviving trees in 1993 (fig. 2), most-common neighbors were four times black walnut planted as seedling or seed, three times each sycamore (*Platanus occidentalis* L.) or baldcypress, and two times or once nine other species. No neighbor species were in common for oak rows having unusually low height or DBH values. If there was a neighbor effect, how many years it would take to suppress growth of adjacent white or red oak is not known.

Relative height growth of the oaks was similar to survival. When all trees combined of each species are compared, white oak was significantly shorter (4.4 versus 5.4 meters) and differences in DBH (6.1 versus 6.0 centimeters) were not statistically significant. White oak planted as seedlings was, however, actively growing in years 15 and 16, while red oak had stagnated (figures 3 and 4). Reasons for the apparently lower heights of the 15- versus 14-year-old trees of both species from seed were not apparent. If growth criteria were of importance, choice of red oak with greater height may be offset by its apparent stagnation after age 14. Older trees of red oak produced acorns and could have enhanced forest succession if the mast had not been eaten or trampled by deer. Not much other animal damage was noted. Red oak typically has been much more damaged by deer than has white oak in stripmine plantings.

Tree Performance When Fall- versus Spring-Planted

Tree survival was highly variable with little consistency related to season. For example, white oak seedlings planted fall 1980 had the greatest number of trees and those planted fall 1979 the lowest (fig. 2). Red oak had the greatest number of trees when planted spring 1981 and the lowest when planted fall 1979 (fig. 2). Mortality after establishment was relatively similar for fall versus spring planting (table 1).

Average heights were lesser for fall- than spring-planted white and red oak seedlings and red oak planted as seed (table 2). DBH was greatest for spring-planted seedlings of both species, though not for spring-planted red oak seed. These averages mask year-to-year variability in relative heights, and DBHs not shown, of fall- versus spring-planted white or red oak (figs. 3 and 4).

Tree Performance When Planted as Seedlings versus Seed

Seedling versus seed differences were variable. Average percent survival tended to be lower for white versus red oak and greater for seedlings versus seed of both species (table 1). There was substantial variability. Number of surviving fall-planted white oak seedlings was greater than from seed at ages 11, 12 and 13 years, and vice versa at ages 14 and 15 years. Reasons for these differences were not evident. The few remaining white oak at age 14 had established in the dry, hot 1980 growing season. For the seasons when both were planted, red oak tended to have greater survival as seedlings than as seed.

The growth of either species from seedlings or seed was also irregular from year to year. White oak trees planted as seedlings were not significantly taller, or greater in DBH, than the averages of those planted as seed (table 3). Red oak seedlings tended to outgrow trees planted as seed when either fall- or spring-planted, and averaged significantly greater DBH when spring-planted.

In a recent survey of seven large coal companies, no local mine planted seedlings in the fall. Although seeding large-seeded tree species has been successful in the midwest, use of seed is unusual on local mines at present.

Table 2—Average height and dbh at age 11-16 years of white and red oak when fall- or spring-planted

Season planted	White oak Height	White oak D.b.h.	Red oak Height	Red oak D.b.h.
	m	*Cm*	*m*	*Cm*
Seedlings				
Fall	4.1[a]	5.5[a]	5.1[a]	5.5[a]
Spring	4.9	7.1	5.9	6.8
Seed				
Fall	3.8	5.0	4.8[a]	5.0[b]
Spring	No seed planted		5.5	5.7
Seedlings and seed				
Fall	4.0	5.3	5.0[a]	5.4[a]
Spring	No seed planted		5.8	6.4

[a] Differences in paired means in a column were statistically significant at $p \leq 0.01$.
[b] Differences in paired means in a column were not statistically significant at $p > 0.05$.

Table 3—Average height and dbh at age 11-16 years of white and red oak when planted as seedlings or seed

Season planted	White oak		Red oak	
	Height	D.b.h.	Height	D.b.h.
	m	*Cm*	*m*	*Cm*
Fall-planted only				
Seedlings	4.1[a]	5.5[a]	5.1[a]	5.5[a]
Seed	4.9	7.1	5.9	6.8
Spring-planted only				
Seedlings	3.8	5.0	4.8[a]	5.0[b]
Seed	No seed planted		5.5	5.7
Fall- and spring-planted				
Seedlings	4.0	5.3	5.0[c]	5.4[b]
Seed	No seed planted		5.8	6.4

[a] Differences in paired means in a column were not statistically significant at $p>0.05$.
[b] Differences in paired means in a column were statistically significant at $p\leq0.01$.
[c] Differences in paired means in a column were statistically significant at $p\leq0.05$.

White Oak Performance with Alternative Growing Stock

Spring-lifted white oak seedlings held in cold storage over summer 1979, or summer 1980, and fall-planted had heights and DBHs similar to those regularly planted in the fall. Growth of Illinois and Missouri seedlings both planted in spring 1982 did not differ. These planting options offer important gains for successful tree planting and bond release on mined lands by extending the planting season. Stuart Miller, Missouri Department of Natural Resources, recently outlined his method for extending to spring the seed planting season with white oak, based on successful over-winter storage of acorns collected in the autumn before or as falling from the trees.

CONCLUSIONS

1. White oak seemed to be inherently less vigorous than red oak in survival and height, and not so in DBH.
2. Trees of both species fall-planted as seedlings or seed had lesser survival, height, and DBH than if spring-planted.
3. Trees of both species tended to have greater survival, height, and DBH when planted as seedlings than as seed.
4. Fall planting of seedlings and fall and spring planting of seed usefully extended our white and red oak reforestation program compared to usual reclamation practice.
5. White oak seedlings held from spring over summer in cold storage for fall planting can have growth equivalent to those regularly fall-planted.
6. Availability of either white or red oak seedlings and seed varied greatly from season to season.
7. Red oak on mined cast overburden produced acorns by year 15.

ACKNOWLEDGMENTS

The work of Clay Kolar and Gary Philo, former researchers in the Department of Botany, Southern Illinois University at Carbondale, contributed substantially to this paper. The Sahara Coal Company, Inc. funded much of the research, the Illinois Department of Conservation supplied tree seedlings, and the USDI Bureau of Mines funded the 1993 measurements.

REFERENCES

Ashby, W.C. 1991. Surface mine tree planting in the midwest pre- and post-public law 95-87. In: Proceedings of the 1991 national meeting of the American society for surface mining and reclamation; Durango, CO. Princeton, WV: American Society for Surface Mining and Reclamation: 617-623.

Ashby, W.C. 1996. Growth of hardwoods and conifers after 47 years on coal mine soils in southern Illinois. Tree Planters' Notes. 47(1): 24-29.

Ashby, W.C.; Kolar, C.A.; Guerke, M.L. [and others]. 1978. Our reclamation future with trees. Carbondale, IL: Southern Illinois University, Coal Extraction and Utilization Research Center. 100 p.

Ashby, W.C.; Kolar, C.A.; Philo, G.R.; Huetteman, C.A. 1995. Database of reclamation with trees in southern Illinois, 1978-1993. Carbondale, IL: Southern Illinois University, Coal Research Center. 47 p. Hard copy of a database on disk.

Bonner, F.T. 1993. Collection and care of acorns. In: Loftis, D.L.; McGee, C.E., eds. Oak regeneration—serious problems, practical recommendations: Proceedings of a symposium; 1992 September 8-10; Knoxville, TN. Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 290-297.

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

Josiah, S.J. 1986. The effects of minesoil construction techniques and ripping on the long term survival and growth of black walnut. In: Proceedings American society for surface mining and reclamation national meeting; Jackson, MS. Princeton, WV: American Society for Surface Mining and Reclamation: 183-193.

Miles, C.; Weiss B. 1978. Soil survey of Saline County, Illinois. Washington, DC: U.S. Department of Agriculture Soil Conservation Service and Forest Service. 94 p. + maps.

LONG-TERM EFFECTS OF WASTEWATER IRRIGATION ON FORESTED ECOSYSTEMS AT PENNSYLVANIA STATE GAME LANDS 176

David S. Larrick and Todd W. Bowersox[1]

Abstract—In 1983, the Pennsylvania State University engaged in a spray irrigation system for the disposal of treated wastewater on 209 hectares of farm and forest land in State Game Lands 176. The system was designed to distribute 5 centimeters of wastewater 52 weeks per year. Differences in species composition and structure have been observed between the irrigated and non-irrigated. In 1997, to monitor the long-term effects of wastewater irrigation on plant community and soil conditions in forested ecosystems, 20 pairs of irrigation and non-irrigation plots were established, and density and species composition of tree, shrub, and herbaceous plants were inventoried. Soil samples of A and B horizons were also taken for each 20 x 20 meters plot. At each plot, plants components included (1) overstory stems on 20 x 20 meters plot, (2) sapling-sized stems on four 10 x 10 meters subplots, (3) seedling-sized stems on four 2 x 2 meters subplots, (4) shrub species coverage on four 10 x 10 meters subplots, and (5) herbaceous plants coverage on sixteen 1 x 1 meters subplots. Initial results indicate striking differences in species composition, in densities of trees and shrubs, and in shrub and herbaceous coverage values, between the irrigated and non-irrigated forested communities. Total numbers of combined tree and shrub species in the non-irrigated overstory, sapling-sized, and seedling-sized components were 15, 47, and 33 species, respectively, compared to 17, 29, and 10 species, respectively, for the irrigated components. Overstory, sapling-sized, and seedling-sized stem densities were 536, 2,823, and 107,316 stems/hectares, respectively, for the non-irrigated and 361, 706, and 1,127 stems/hectares, respectively, for the irrigated components. Shrub and herbaceous coverage values were 17 and 80 percent, respectively, in the irrigated area, compared to 61 and 30 percent, respectively, in the non-irrigated area. The fertility levels in soil A and B horizons were also very different for the two areas. For both A and B soil horizons, the irrigated area had higher pH, higher phosphorus, magnesium, and calcium contents, and lower exchangeable acidity than the non-irrigated area. These data suggest that modifications to either the plant community species base or the methods of wastewater distribution will be needed to maintain forested ecosystems in the wastewater irrigation area of State Game Lands 176.

INTRODUCTION

Disposal of sewage effluent has become a problem for many municipalities due to population increases and stricter regulatory controls. Typically, wastewater from sewage treatment plants is discharged into rivers or lakes. However, this disposal method has often lead to environmental problems through the eutrophication of aquatic systems. An alternative method, disposal on land, has the potential advantages of reducing stream pollution and recharging ground water reserves (Sopper and Kardos 1973). Effluent disposal experiments conducted in forested areas have shown that spray irrigation with chlorinated sewage effluent caused changes in the structure and species composition of vegetation. Increased rate of tree growth, increased plant biomass, altered species composition of ground-level plants, and decreased production and survival of native shrub and tree species have been brought about by the addition of nutrients and water to soil and by ice damage during winter (Brister and Schultz 1981, Epstein and Sawhill 1977, Sopper and Kardos 1973).

In 1983, the Pennsylvania State University engaged a spray irrigation system for 209 hectares of farm and forest land managed by the Pennsylvania Game Commission (State Game Lands 176). The system is designed to distribute 5 centimeters of wastewater 52 weeks per year. About one-half of the current forest land was always forest land, whereas the other half had been used for agriculture. No actual inventories of the shrub and tree species composition and structure have been conducted, but there has been an observed difference between irrigated and non-irrigated areas. Observations suggested there has been increased mortality of the trees in the overstory of irrigated areas as compared to non-irrigated areas. It has also been observed that the understory of the irrigated forests has more herbaceous plants and fewer shrub and tree species stems than the nearby non-irrigated forests. Actual changes that have occurred to the areas that would be different from normal stand development are unknown. However, it is possible to compare the present conditions in similar irrigated and non-irrigated forests and to measure future changes in these two conditions. A project has been started to determine the long-term effects of wastewater irrigation on forested ecosystems by establishing and maintaining a program to monitor plant community and soil conditions in irrigated and non-irrigated forested areas. Results from the initial inventories are presented in this paper.

METHODS

The study area was located in central Pennsylvania, 2 kilometers north of State College. The gently rolling terrain was in the Ridge and Valley region (Lull 1968) on lands managed by the Pennsylvania Game Commission. The soil was Morrison sandy loam (Ultric, Hapludalf; fine, loamy,

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mixed, mesic). Site quality was about average, and if occupied by evenaged mixed oak stands, the site index would be about 65.

Ideally the sampling procedures and design would have been prepared prior to treatment but the effects of wastewater irrigation on forested ecosystems was not a concern in 1983. In July 1997, 20 pairs of irrigation and non-irrigation long-term monitor plots were established to inventory density and species composition of the tree, shrub and herbaceous plants in forested areas. The plots were paired on the basis of having similar physical site conditions, history of use, and perceived species conditions in 1983 when the irrigation system was engaged. Reasons for some forested areas to be irrigated and others not irrigated were only due to the design features of the irrigation network. Plots were established and inventoried according to the Storm and Ross (1992) protocol for monitoring vegetation on public lands. A total of 40, 20 x 20 meters permanent plots were used to inventory the overstory trees and shrubs. Species, diameter (at 1.3 meters above ground in centimeters), health class (healthy, dead, or injured), and type of injury (branch breakage, stem breakage, or stem decay) of all overstory stems (≥11 centimeters in diameter) were recorded. Each 20 x 20 meters plot was subdivided into four 10 x 10 meters subplots to acquire species and diameter of all sapling-sized stems and shrub coverage. Nested within each 20 x 20 meters plot were four 2 x 2 meters subplots, each located 5 meters from plot center along cardinal directions, to inventory abundance of seedling-sized stems. Each 2 x 2 meters subplot was further subdivided into four 1 x 1 meters subplots to inventory coverage of herbaceous groups (grass, broadleaf, moss, vine). A randomly selected soil sample was also taken from near the center of each plot. Soil from the A (0-4 cm below humus layer) and B (10-25 cm below humus layer) horizons was analyzed for pH, extractable phosphorus (kilograms/hectare), exchangeable calcium, magnesium, potassium and cation exchange capacity.

Analysis of variance was used to test for significant differences between irrigated and non-irrigated areas in mean (1) number of species, density, basal area and stocking of overstory stems,(2) number of species and density of sapling-sized stems, (3) number of species and density of seedling-sized stems, (4) shrub species and coverage, (5) herbaceous coverage, and (6) soil chemical properties. Significance at the 0.05 level was used in all cases. Using analysis of variance to test for significance between sample plots selected after treatment violates basic design requirements. However, the analyses were conducted to provide an expression of variability about mean values.

RESULTS AND DISCUSSION

Overstory Stems

A total of 17 tree species were inventoried on the 20 paired permanent overstory plots, with the irrigated and non-irrigated areas containing 15 and 13 species, respectively. There was no significant difference in number of tree species per plot between irrigated (4 species/plot) and non-irrigated (5 species/plot) areas (Table 1). Tree density (Table 1) in the non-irrigated area (531 stems/hectare) was significantly different from the irrigated area (355 stems/hectare). The non-irrigated area had more aspen (see Appendix 1 for scientific names), black cherry, and pine stems than the irrigated area (Table 2). Oak, red maple, and other trees were equally abundant in both areas. The irrigated area had more ash and hickory stems than the non-irrigated area, but the numbers of ash and hickory were low for both areas. The irrigated area (Table 3) had a lower proportion of healthy trees (64 percent) and a higher proportion of injured trees (29 percent) than the non-irrigated area (84 percent healthy and 9 percent injured). These data indicate poor health in the overstory of the irrigated area. Basal area (27 meters^2/hectare) and stocking (100 percent) were significantly greater in the non-irrigated area (Table 1) than in the irrigated area (19 meters^2/hectare basal area and 73 percent stocking).

Overstories in both the irrigated and non-irrigated areas had low numbers of shrub species (2 in both areas) and low shrub densities (6 and 5 stems/hectare, respectively). There were no significant differences between irrigated and non-irrigated areas in terms of shrub species, density, basal area, healthy, dead, or injured stems (Tables 1 and 3).

It appears that the forested areas in the irrigated area are developing an open, park-like overstory that is comprised of less healthy trees than adjacent non-irrigated areas. There seems to be a change in the species composition from the non-irrigated mixed hardwood-pine to red maple-pine-mixed hardwood in the irrigated areas. Overstory density of oak and red maple trees did not appear to be affected by irrigation.

Table 1—Mean[a] overstory (≥11 cm in diameter) number of species per 0.04 ha plot, density, basal area, and stocking for tree, shrub, and total species in irrigated and non-irrigated areas

Species group area	Number of species	Density	Basal area	Stocking
	Per plot	*Stems/ha*	*m²/ha*	*Percent*
Tree				
Irrigated	4 a	355 b	19 b	73 b
Non-irrigated	5 a	531 a	27 a	100 a
Shrub				
Irrigated	<1 a	6 a	<1 a	-
Non-irrigated	<1 a	5 a	<1 a	-
Total				
Irrigated	5 a	361 b	20 b	73 b
Non-irrigated	5 a	536 a	27 a	100 a

[a] Means with the same letter are not significantly different (α = 0.05).

Table 2—Mean density of overstory (≥11 cm in diameter) trees and shrubs by species group[a] in irrigated and non-irrigated areas

Species group	Irrigated	Non-irrigated
	Stems/ha	
Trees		
Ash	11	1
Aspen	23	119
Black cherry	28	56
Hickory	14	1
Oak	74	70
Pine	82	161
Red maple	120	120
Other tree	3	3
Total tree	355	531
Shrubs		
Flowering dogwood	0	1
Sassafras	5	4
Striped maple	1	0
Total shrubs	6	5

[a] Hickory included mockernut and pignut; oak included black, chestnut, scarlet, and white; pine included pitch, Scotch, and eastern white; other tree included American beech, blackgum, slippery elm, and sweet birch (see Appendix 1 for scientific names).

Table 3—Mean[a] overstory (≥11 cm in diameter) tree, shrub, and total healthy, dead, and injured stems in irrigated and non-irrigated areas

Species group area	Healthy	Dead	Injured
		Pct	
Tree			
Irrigated	64 b	7 a	29 a
Non-irrigated	84 a	7 a	9 b
Shrub			
Irrigated	56 a	33 a	11 a
Non-irrigated	0 a	25 a	75 a
Total			
Irrigated	64 b	7 a	29 a
Non-irrigated	83 a	7 a	9 b

[a] Means with the same letter are not significantly different ($\alpha = 0.05$).

Sapling-Sized Stems

A total of 51 sapling-sized tree and shrub species were inventoried, with the irrigated and non-irrigated areas containing 29 and 47 total species, respectively. Number of species per plot (Table 4) was significantly greater in the non-irrigated area (7 species/plot) than in the irrigated area (3 species/plot). Total density also was significantly different between the non-irrigated (2,823 stems/hectare) and irrigated (706 stems/hectare) areas (Table 4).

Of the 21 tree species inventoried, the irrigated area had 13 species, and the non-irrigated area had 19 species. There was a significant difference in number of tree species between irrigated (1 species/plot) and non-irrigated (3 species/plot) areas (Table 4). The 1,585 tree species stems/hectare in the non-irrigated area was significantly greater than the 309 stems/hectare in the irrigated area (Table 4). All tree species, except ash, had substantially lower numbers of sapling-sized stems in the irrigated area as compared to the non-irrigated area (Table 5). There were 118 ash stems/hectare in the irrigated area and 98 stems/hectare in the non-irrigated area. Birch, hickory, oak, and pine sapling-sized stems were present in the non-irrigated area but were nearly absent from the irrigated area (Table 5).

The total number of shrub species was 30. There were 16 and 28 shrub species inventoried in the irrigated and non-irrigated areas, respectively. Numbers of shrub species and shrub densities (Table 4) were significantly greater in the non-irrigated area (3 species/plot and 1,238 stems/hectare) than the irrigated area (1 species/plot and 397 stems/hectare). The non-irrigated area had an abundance of sapling-sized stems for a variety of shrub species, with Tatarian honeysuckle and sassafras being the most prevalent. Hophornbeam, Tatarian honeysuckle, and spicebush were the most common sapling-sized shrub species in the irrigated area. Autumn olive, blackhaw, blueberry, dogwood, privet, and sassafras were less common in the irrigated area than in the non-irrigated area (Table 5).

Table 4—Mean[a] number of species and density of sapling-sized (≥151 cm in height and <11 cm in diameter) tree, shrub, and total species in irrigated and non-irrigated areas

Species group area	Number of species	Density
	0.01/ha	Stems/ha
Tree		
Irrigated	1 b	309 b
Non-irrigated	3 a	1,585 a
Shrub		
Irrigated	1 b	397 b
Non-irrigated	3 a	1,238 a
Total		
Irrigated	3 b	706 b
Non-irrigated	7 a	2,823 a

[a] Means with the same letter are not significantly different ($\alpha = 0.05$).

Table 5—Mean density of sapling-sized (≥151 cm in height and <11 cm in diameter) trees and shrubs by species group[a] in irrigated and non-irrigated areas

	Area	
Species group	Irrigated	Non-irrigated
Trees		
Ash	118	98
Birch	1	24
Black cherry	55	371
Hickory	4	13
Maple	113	850
Oak	7	171
Pine	1	44
Other tree	10	14
Total tree	309	1,585
Shrubs		
Autumn olive	3	56
Raspberry	9	89
Blackhaw	0	23
Blueberry	0	34
Dogwood	1	76
Hophornbeam	11	18
Privet	1	78
Rose	9	101
Sassafras	0	330
Spicebush	14	6
Tatarian honeysuckle	330	386
Other shrubs	19	41
Total shrubs	397	1,238

[a] Birch included paper and sweet; hickory included bitternut, mockernut, and pignut; maple included red and sugar; oak included black, chestnut, northern red, scarlet and white; other tree included American elm, aspen, basswood, black walnut, blackgum, and slippery elm. Dogwood included flowering and gray; other shrub included American chestnut, American hazel, apple, boxelder, buckthorn, common elderberry, cranberry, dwarf oak, hawthorn, Hercules'-club, Japanese barberry, mapleleaf viburnum, musclewood, red elderberry, red mulberry, striped maple, viburnum, and witch-hazel.

Seedling-Sized Stems

Of the 33 seedling-sized tree and shrub species inventoried, only 10 species occurred in the irrigated area, while all 33 species were inventoried in the non-irrigated area. There were significant differences in total species per plot and total density per hectare between the irrigated and non-irrigated areas (Table 6). The irrigated area contained less than 1 species/plot and had a density of 1,127 stems/hectare, whereas the non-irrigated area had 5 species/plot and 107,316 stems/hectare.

The non-irrigated area contained 13 tree species, compared to only 6 species in the irrigated area. On a per plot basis, there were 3 tree species inventoried in the non-irrigated area, versus less than 1 species in the irrigated area (Table 6). Tree density in the irrigated area of 501 seedling-sized stems/hectare was essentially non-existent compared to the 77,970 stems/hectare in the non-irrigated area (Table 6). Seedling-sized stems in the non-irrigated area were dominated by maple, with abundant black cherry and oak (Table 7). Ash, black cherry, and maple were the most abundant species in the irrigated area. However, seedling-sized densities were much lower in the irrigated area for all of the tree species groups, with aspen, paper birch, and eastern white pine not recorded in the irrigated area.

Table 6—Mean[a] number of species and density of seedling-sized (<151 cm in height) tree, shrub, and total in irrigated and non-irrigated areas

Species group area	Number of species	Density
	0.0004/ha	*Stems/ha*
Tree		
Irrigated	<1 b	501 b
Non-irrigated	3 a	77,970 a
Shrub		
Irrigated	<1 b	626 b
Non-irrigated	2 a	29,346 a
Total		
Irrigated	<1 b	1,127 b
Non-irrigated	5 a	107,316 a

[a] Means with the same letter are not significantly different ($\alpha = 0.05$).

Striking differences in the number of shrub species and densities of shrub species stems also occurred between irrigated and non-irrigated areas. Shrub species totaled 20 in the non-irrigated area and 4 in the irrigated area. Both the number of species/plot (2) and stems/hectare (29,346) in the non-irrigated area were significantly greater than the number of species/plot (<1) and stems/hectare (626) in the irrigated area (Table 6). There were 12 seedling-sized shrub species groups in the non-irrigated area, with good densities of blackberry/raspberry, blueberry, rose, sassafras, and Tatarian honeysuckle (Table 7). In the irrigated area, there were only four seedling-sized shrub species groups, and only Tatarian honeysuckle seemed to be somewhat tolerant of the conditions in the irrigated area.

Shrub Coverage

Over all size classes, the non-irrigated area contained 33 shrub species, compared to 20 shrub species in the irrigated area. There was a significant difference in number of shrub species per plot between irrigated (2 species/plot) and non-irrigated (6 species/plot) areas (Table 8). Shrub coverage in the non-irrigated area (61 percent) was significantly greater than in the irrigated area (17 percent) (Table 8). Tatarian honeysuckle had equal coverage values (13 percent, each) on both the irrigated

Table 7—Mean density of seedling-sized (<151 cm in height) trees and shrubs by species group[a] in irrigated and non-irrigated areas

Species group	Area: Irrigated	Area: Non-irrigated
	------------- *Stems/ha* -------------	
Trees		
Ash	94	938
Aspen	0	250
Black cherry	219	21,750
Hickory	31	750
Maple	94	49,781
Oak	63	4,407
Paper birch	0	63
White pine	0	31
Total tree	501	77,970
Shrubs		
Autumn olive	0	563
Raspberry	0	15,531
Blueberry	0	3,406
Common elderberry	0	219
Cranberry	0	188
Dogwood	0	750
Japanese barberry	31	188
Rose	94	2,156
Sassafras	0	1,906
Tatarian honeysuckle	438	3,313
Viburnum	0	688
Other shrubs	63	438
Total shrubs	626	29,346

[a] Hickory included mockernut and pignut; maple included red and sugar; oak included black, chestnut, northern red, and white. Dogwood included flowering and gray; viburnum included mapleleaf and viburnum; other shrub included American chestnut, apple, blackhaw, hawthorn, hophornbeam, spicebush, and striped maple.

and non-irrigated areas, whereas the coverage values of blackberry/raspberry, rose, and sassafras were considerably lower in the irrigated area as compared to the non-irrigated area (Table 8). Coverage values of the other species were too low to evaluate any irrigation effects.

Herbaceous Coverage

Total herbaceous coverage (Table 9) in the irrigated area (80 percent) was much greater than in the non-irrigated area (30 percent), primarily due to a significant difference in broadleaf coverage (63 and 17 percent coverage in irrigated and non-irrigated areas, respectively). Vine and moss coverage values were slightly higher in the irrigated area (15 percent vine and 2 percent moss coverage) than the non-irrigated area (11 percent vine and 1 percent moss coverage). Each area contained less than 1 percent grass coverage.

Table 8—Combined mean coverage of overstory, sapling- and seedling-sized shrubs by species group[a] in irrigated and non-irrigated areas

Species group	Area: Irrigated	Area: Non-irrigated
	------------ % coverage -----------	
Autumn olive	<1	2
Raspberry	1	19
Blueberry	<1	3
Dogwood	<1	3
Hophornbeam	1	<1
Japanese barberry	<1	1
Privet	<1	2
Rose	<1	7
Sassafras	0	9
Spicebush	<1	<1
Tatarian honeysuckle	13	13
Other shrubs	1	2
Total shrubs	17	61

[a] Dogwood included flowering and gray; other shrub included American chestnut, American hazel, apple, blackhaw, boxelder, buckthorn, choke cherry, common elderberry, cranberry, currant/gooseberry, dwarf oak, hawthorn, Hercules'-club, mapleleaf viburnum, musclewood, red elderberry, red mulberry, serviceberry, striped maple, sumac, viburnum, winterberry holly, and witch-hazel.

Table 9—Mean[a] broadleaf, grass, moss, vine, and total herbaceous coverage in irrigated and non-irrigated areas

Species group area	Coverage
	Percent
Broadleaf	
Irrigated	63 a
Non-irrigated	17 b
Grass	
Irrigated	<1 a
Non-irrigated	<1 a
Moss	
Irrigated	2 a
Non-irrigated	1 b
Vine	
Irrigated	15 a
Non-irrigated	11 b
Total	
Irrigated	80 a
Non-irrigated	30 b

[a] Means with the same letter are not significantly different (a = 0.05).

Soils

There were substantial differences between the irrigated and non-irrigated areas in the chemistry of soil A horizon (Table 10). The A horizon of the irrigated area had significantly higher soil pH (7.1), phosphorus (345 kilograms/hectare), magnesium (3.16 milliequivalent/100 grams), and calcium (9.78 milliequivalent/100 grams) than the non-irrigated area (4.8 soil pH, 48 kilograms/hectare phosphorus, 0.39 milliequivalent/100 grams magnesium, and 2.42 milliequivalent/100 grams calcium). The non-irrigated area had significantly higher exchangeable acidity (10.32 milliequivalent/100 grams) than the irrigated area (0.10 milliequivalent/ 100 grams). Potassium (0.19 milliequivalent/100 grams) and cation exchange capacity (13.22 milliequivalent/100 grams) for the irrigated area were not significantly different from the non-irrigated area (0.18 and 13.03 milliequivalent/100 grams potassium and cation exchange capacity, respectively).

For the B horizon (Table 10), the irrigated area had significantly higher soil pH (7.2), phosphorus (295 kilograms/hectare), potassium (0.18 milliequivalent/100 grams), magnesium (0.95 milliequivalent/100 grams), and calcium (2.41 milliequivalent/100 grams) than the non-irrigated area (5.0 soil pH, 45 kilograms/hectare phosphorus, 0.09 milliequivalent/100 grams potassium, 0.18 milliequivalent/100 grams magnesium, and 0.55 milliequivalent/100 grams calcium). The non-irrigated area had significantly higher exchangeable acidity (4.86 milliequivalent /100 grams) and cation exchange capacity (5.61 milliequivalent/100 grams) than the irrigated area (0.00 milliequivalent/100 grams exchangeable acidity and 3.52 milliequivalent/100 grams cation exchange capacity).

Differences between irrigated and non-irrigated areas were a direct reflection of the chemical composition of the wastewater additions, which had greater levels of phosphorus, magnesium and calcium than what would be found in rain water. The soil chemistry in the non-irrigated area was what would be expected for a soil that developed from acid sandstone. With the exception of potassium in the A horizon, soil chemical levels in the irrigated area were what would expected for a soil that developed from a calcareous parent material. Water for the Pennsylvania State University is from wells supplied from primarily limestone bedrock.

CONCLUSIONS

Initial results of the vegetation monitoring program at Pennsylvania State Game Lands 176 indicate substantial differences in species composition and densities of trees and shrubs between the wastewater irrigated and non-irrigated forested areas. Total numbers of combined tree and shrub species in the non-irrigated overstory, sapling-sized, and seedling-sized components were 15, 47, and 33 species, respectively, compared to 17, 29, and 10 species, respectively, for the irrigated components. Overstory, sapling-sized, and seedling-sized stem densities were 536, 2,823, and 107,316 stems/hectare, respectively, for the non-irrigated and 361, 706, and 1,127 stems/hectare, respectively, for the irrigated components. It appears that the forested areas in the irrigated area are developing an open, park-like overstory that is comprised of less healthy trees than adjacent non-irrigated areas. Shrub and herbaceous coverage values were 17 and 80 percent in the irrigated area, compared to 61 and 30 percent in the non-irrigated area. For both A and B soil horizons, the irrigated area had higher pH, higher phosphorus, magnesium, and calcium contents, and lower exchangeable acidity than the non-irrigated area.

Since there were no pre-wastewater irrigation inventories, we can not be certain that the differences after 15 years between the irrigated and non-irrigated areas were due to the application of wastewater. However, the evidence strongly supports that the differences are related to the year-long irrigation with 5 cm of wastewater per week. The reasons for these differences in the plant communities are not known. It is possible that the indigenous herbaceous, shrub, and tree species can not adapt to the enhanced nutrients and water from the irrigation system. There may be other herbaceous, shrub, and tree species that are

Table 10—Mean[a] chemical properties of soil A and B horizons[b] in irrigated and non-irrigated areas

Horizon area	pH	P	Acidity	K	Mg	Ca	CEC
		Kg/ha	------- Exchangeable cations (*meq/100 g*) -------				
Horizon A							
Irrigated	7.1a	345a	0.10b	0.19a	3.16a	9.78a	13.22a
Non-irrigated	4.8b	48b	10.32a	0.18a	0.39b	2.42b	13.03a
Horizon B							
Irrigated	7.2a	295a	0.00b	0.18a	0.95a	2.41a	3.52b
Non-irrigated	5.0b	45b	4.86a	0.09b	0.18b	0.55b	5.61a

[a] Means with the same letter are not significantly different ($\alpha = 0.05$).
[b] A horizon was 0-4 cm below humus layer; B horizon was 10-25 cm below humus layer.

capable of creating and maintaining an acceptable forest community in these amended conditions. It is also possible that the mechanics of the overhead spray of water may be causing physical damage to the plants or creating environments that are favorable for the development of undesirable insects or pathogens. Finally, it is also possible that the bending and breaking on the seedling-sized and sapling-sized stems from heavy ice loading may be selectively removing most of the tree and shrub species from the communities. Based on the data collected in this study, it is clear that modifications to either the plant community species base or the methods of wastewater distribution will be needed to maintain forested ecosystems in the wastewater irrigation area of State Game Lands 176.

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REFERENCES

Brister, G.H.; Schultz, R.C. 1981. The response of a Southern Appalachian forest to waste water irrigation. Journal of Environmental Quality. 10: 148-153.

Epstein, C.M.; Sawhill, G. 1977. Changes in the ground cover of a New Jersey pine barrens woodland resulting from spray irrigation of sewage effluent. Bulletin New Jersey Acad. Sci. 22: 6-11.

Lull, H.W. 1968. A forest atlas of the Northeast. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 46 p.

Sopper, W.E.; Kardos, L.T., eds. 1973. Recycling treated municipal wastewater and sludge through forest and cropland. University Park, PA: The Pennsylvania State University Press. 479 p.

Storm, G.L.; Ross A.S. 1992. Manual for monitoring vegetation on public lands in mid-Atlantic United States. University Park, PA: Pennsylvania Cooperative Fish and Wildlife Unit; The Pennsylvania State University. 89 p.

Appendix 1—List of total tree and shrub species

Tree Species
•American beech (*Fagus grandifolia* Ehrhart)
•American elm (*Ulmus americana* L.)
•ash (*Fraxinus* spp.)
•aspen (*Populus* spp.)
•basswood (*Tilia americana* L.)
•bitternut hickory (*Carya cordiformis* (Wangenheim) K. Koch)
•black cherry (*Prunus serotina* Ehrh.)
•black oak (*Quercus velutina* Lam.)
•black walnut (*Juglans nigra* L.)
•blackgum (*Nyssa sylvatica* Marshall)
•chestnut oak (*Quercus prinus* L.)
•eastern white pine (*Pinus strobus* L.)
•mockernut hickory (*Carya tomentosa* (Lam. ex Poir.) Nutt.)
•northern red oak (*Quercus rubra* L.)
•paper birch (*Betula papyrifera* Marshall)
•pignut hickory (*Carya glabra* (P.Mill.) Sweet)
•pitch pine (*Pinus rigida* P.Mill.)
•red maple (*Acer rubrum* L.)
•scarlet oak (*Quercus coccinea* Muenchh.)
•Scotch pine (*Pinus sylvestris* L.)
•slippery elm (*Ulmus rubra* Muhl.)
•sugar maple (*Acer saccharum* Marshall)
•sweet birch (*Betula lenta* L.)
•white oak (*Quercus alba* L.)

Shrub Species
•American chestnut (*Castanea dentata* (Marshall) Borkh.
•American hazel (*Corylus americana* Walt.)
•apple (*Malus* spp.)
•autumn olive (*Elaeagnus umbellata* Thunb.)
•blackberry/raspberry (*Rubus* spp.)
•blackhaw (*Viburnum prunifolium* L.)
•blueberry (*Vaccinium* spp.)
•boxelder (*Acer negundo* L.)
•buckthorn (*Rhamnus cathartica* L.)
•choke cherry (*Prunus virginiana* L.)
•common elderberry (*Sambucus canadensis* L.)
•cranberry (*Viburnum trilobum* Marshall)
•currant/gooseberry (*Ribes* spp.)
•dwarf oak (*Quercus prinoides* Willd.)
•flowering dogwood (*Cornus florida* L.)
•gray dogwood (*Cornus racemosa* Lam.)
•hawthorn (*Crataegus* spp.)
•Hercules'-club (*Aralia spinosa* L.)
•hophornbeam (*Ostrya virginiana* (P.Mill.) K.Koch
•Japanese barberry (*Berberis thunbergii* DC.)
•mapleleaf viburnum (*Viburnum acerifolium* L.)
•musclewood (*Carpinus caroliniana* Walter)
•privet (*Ligustrum obtusifolium* Sieb. & Zucc.)
•red elderberry (*Sambucus pubens* Michx. PFGBW)
•red mulberry (*Morus rubra* L.)
•rose (*Rosa* spp.)
•sassafras (*Sassafras albidum* (Nutt.) Nees)
•serviceberry (*Amelanchier arborea* (Michx.f.) Fern.)
•spicebush (*Lindera benzoin* (L.) Blume)
•striped maple (*Acer pennsylvanicum* L.)
•sumac (*Rhus* spp.)
•Tatarian honeysuckle (*Lonicera tatarica* L.)
•viburnum (*Viburnum dilatatum* L.)
•winterberry holly (*Ilex verticillata* (L.) A.Gray)
•witch-hazel (*Hamamelis virginiana* L.)

SURVIVORSHIP AND GROWTH OF NATURAL NORTHERN RED OAK (*QUERCUS RUBRA* L.) SEEDLINGS IN RESPONSE TO SELECTED TREATMENTS ON AN EXTREMELY ACIDIC FOREST SOIL

Michael C. Demchik and William E. Sharpe[1]

Abstract—Natural seedlings of northern red oak (*Quercus rubra* L.) growing on extremely acidic soils are subjected to stresses resulting from the relatively inhospitable soil chemical environment, competition from other plants and herbivory by white-tailed deer (*Odocoileus virginianus* Boddaert). The objective of this study was to determine if lime and fertilizer, fencing and vegetation control could increase the growth and survivorship of natural northern red oak seedlings. Three replicate plots for each possible combination of liming and fertilization (6600 kg/ha dolomitic lime, 110 kg/ha K_20 equivalents and 220 kg/ha P_2O_5 equivalents), vegetation control (removal of fern fronds) and fencing (woven wire fencing) were established in the spring of 1995. Two years after treatment, the area was subjected to a partial cut. Without any other site treatment, vegetation control increased mortality. Lime and fertilizer treatment increased height growth before and after cutting. Vegetation control increased height growth after cutting but had no effect on height growth prior to cutting. Fencing increased seedling height growth prior to and after cutting. Seedlings that were not fenced actually lost height in the year after cutting. While seedlings were responsive to treatment, the magnitude of the growth response was low.

INTRODUCTION

Northern red oak (*Quercus rubra* L.) is a major component of mature forest stands in southwestern Pennsylvania. Oak is a valuable species for both timber and wildlife; therefore, inclusion of a large component of oak in forest regeneration after logging is an important goal of forest management in these forests. However, on many sites in Pennsylvania, northern red oak is often only a minor component of seedling regeneration (McWilliams and others 1995). Historically, 75 percent of the regeneration that developed into the current oak overstory was seedling origin (McIntyre 1936) but regeneration following recent harvests is often less than desirable. Several factors have been hypothesized to explain this lack of regeneration success. These include competition with hayscented ferns (*Dennstaedtia punctilobula* (Michx.) Moore); (Horsley 1993; Lyon 1996), browsing by deer (*Odocoileus virginianus* Boddaert); (Tilghman 1989; Marquis and Brenneman 1981) and acidic, infertile soil (Demchik and Sharpe 1996; Lyon 1996; Cronan and Grigal 1995; Joslin and Wolfe 1989).

Competition by hayscented fern is capable of impeding success of forest regeneration. Horsley (1993) has documented that although black cherry (*Prunus serotina* Ehrh.) seeds germinate after cutting, they are out-competed by hayscented fern. One factor in this competition is light. Light competition has been suggested as an important factor by Hippensteel and Bowersox (1995). They showed inhibition of white ash (*Fraxinus americana* L.) seedlings by altered light quality; however, they also reported the existence of a non-specific below ground competition. Lyon and Sharpe (1996) also found that hayscented fern limited northern red oak seedling growth as a consequence of below ground competition.

Deer browsing also constitutes a serious limitation to regeneration of oaks (Tilghman 1989; Marquis and Brenneman 1981). Both oak stump sprouts and seedlings are extensively browsed. However, even when deer browsing is limited by electric fencing, regeneration of oaks may not be successful (Demchik and Sharpe 1996; Lyon and Sharpe 1995).

Low levels of available soil bases and high levels of available aluminum may reduce the growth rates of northern red oak (DeWald and others 1990; Joslin and Wolfe 1989). Extremely acidic forest soils often have quite low levels of available calcium and high levels of available aluminum. The potential for aluminum toxicity exists when this occurs (Cronan and Grigal 1995). Indeed, Joslin and Wolfe (1989) found a reduction of growth (at ambient aluminum levels) due to increases in levels of aluminum in their test soils.

Attempts to increase growth of northern red oak seedlings with addition of lime and fertilizer have been largely successful in greenhouse trials, but field results have been mixed (Hart 1995; Hart and Sharpe 1997; Sharpe and others 1993; Pickens 1995). While field results have varied, amelioration of soil base nutrient deficiency and aluminum toxicity remains a promising technique for improving oak seedling growth.

Our study site was located in the Laurel Hill region of southwestern Pennsylvania. This region contains soil horizons with relatively high plant available Al and low Ca (Lyon and Sharpe in press; Demchik and Sharpe 1996).

The objective of this study was to determine if lime and fertilizer, fencing and vegetation control could increase

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growth and survivorship of natural northern red oak seedlings.

METHODS

The Site

An 80-85 year old stand of hardwoods was selected on the Laurel Hill in southwestern Pennsylvania. The stand was composed of 67 percent northern red oak (average diameter of 31.5 cm), 22 percent red maple (*Acer rubrum*; average diameter of 15.9 cm), 7 percent black cherry (average diameter of 30.4 cm) and 2 percent other species. There was an average of 291 trees/ha. The land was mildly sloping (0-20 percent), northeast facing ridgetop. The area is part of Pennsylvania's State Forest system managed by the Pennsylvania Bureau of Forestry. The soils are Typic Dystrochrepts of the Dekalb very stony loam series. Results from 10 hand excavated soil pits on the site revealed a mean organic layer thickness of 4.7±1.4 cm and A-horizon thickness of 7.1±3.6 cm. Site index (50 year) for oak was 19.5 - 22.5 meters. Data from an adjacent site showed soil pH values of the mineral horizon averaged 4.2 and base saturation was 3.1 percent of total exchange sites.

The site had received a light shelterwood cut 7 years prior to the beginning of this experiment, which reduced the basal area to approximately 18 m^2/ha. In addition, a salvage cut was made 3 years prior to this experiment to remove dead and dying trees. The cause of this mortality was unknown. Even though the understory cover was dominated by hayscented fern, northern red oak seedlings averaged 56,000±1000 /ha.

The Design

Twenty-four 1.25 m^2 plots each containing a minimum of 10 natural northern red oak seedlings each were established. All seedlings on the site were small (none taller than 25 cm). Ten randomly selected seedlings were tagged in each plot. These seedlings were measured for height and number of leaves immediately prior to treatment for use as a covariate. Three replicate plots of all possible combinations of the binary treatments of soil amendment, fencing and vegetation control were created. The soil amendment treatment consisted of an application of 6600 kg/ha dolomitic lime (53.5 percent calcium carbonate; 42.0 percent magnesium carbonate; 105.6 percent total calcium carbonate equivalents), 110 kg/ha K_2O equivalents and 220 kg/ha P_2O_5 equivalents applied in mid-May (shortly after bud break) or a control with no amendment. On an adjacent site, this application increased A-horizon soil pH from 4.1±0.04 to 4.5±0.04 and increased base saturation from 3.2±0.3 percent to 15.8±2.5 percent (Demchik 1998). Fencing consisted of encircling the plots with 1.25 meter tall 5 cm woven wire secured to iron reinforcement bars or a control with no fence. Vegetation control consisted of removal of all fern fronds within the plots as they appeared (approximately 3 times annually) or no vegetation control. Measurement of height, number of leaves and total leaf area of each plant was conducted in mid-August for three growing seasons. After 19 months (mid January 1997), the surrounding stand was cut to a residual of 13 m^2/ha basal area. Bolewood was removed and slash was left on site. Care was taken not to damage any of the seedlings. Nested ANCOVA was used for analysis of seedling height in a nested factorial design. Survivorship was tested with Chi-square test.

RESULTS AND DISCUSSION

Mortality

During the first growing season, no mortality occurred from the time of treatment until mid-August. During the second growing season, overall mortality was approximately 7.5 percent. No significant difference existed for mortality between treatments (Table 1). For the third growing season (first season after partial cutting), the highest mortality occurred in plots that were not limed and fertilized or fenced but received vegetation control (65 percent) (Table 1). This treatment combination consistently had the greatest mortality. Fenced plots had lower mortality (23 percent) than unfenced plots (39 percent). This response was primarily driven by the high rate of mortality for the plots that only received vegetation control (65 percent). With exclusion of these plots, unfenced plots had 30 percent overall mortality. This rate of mortality was similar to the rate for fenced plots. Plots that were limed and fertilized had less mortality (26 percent) than those that were not limed and fertilized (36 percent). When the treatment combination of no lime and fertilizer, no fencing but vegetation control was excluded rates of mortality were similar (both 26 percent).

The treatment of only vegetation control resulted in the greatest mortality. When vegetation control was applied with fencing, the rate of mortality was similar to all other treatments. Exposure of seedlings to deer browsing was therefore suggested as a contributing factor in the increased mortality on the vegetation control plots, perhaps because seedlings were more visible to the deer. This is consistent with the observations of other researchers (Tilghman 1989; Marquis and Brenneman 1981). Annual seedling mortality, even in control plots, was generally less than 15 percent before cutting and less than 40 percent after cutting, excluding the vegetation control treatment plots. This suggested that northern red oak seedlings could persist under the adverse conditions present on this site. Seedling red oaks are moderately shade tolerant (Phares 1971) and can remain viable under moderate light conditions for 8-10 years in the understory (McQuilkin 1983). However, in order for seedlings to be successful, they must grow at suitable rates.

Height Growth

Prior to partial overstory cutting, unfenced seedlings grew more slowly (annual growth of 0.8±0.04 cm) than fenced seedlings (annual growth of 1.3±0.05 cm); ($p<0.05$) (Table 2). After cutting, unfenced seedlings lost 1.1±0.06 cm of height and fenced seedlings gained 3.5±1.0 cm of height ($P<0.0001$) (Table 2). Deer browsing is well known to reduce success of forest regeneration (Tilghman 1989; Marquis and Brenneman 1981). The slow growth of fenced seedlings was surprising. When free from deer browsing, these seedlings only grew an average of 3.5 cm. This

Table 1—The effect of all possible combinations of lime and fertilizer (L&F), vegetation control (weed) and fencing on annual percentage mortality of northern red oak seedlings for two years prior to partial overstory cutting (1995 and 1996) and 1 year after partial overstory cutting (1997). Mortality is based on number of seedlings alive at the previous census

			Annual mortality		
L&F	Weed	Fencing	1995	1996	1997
			-----------*Percent*-----------		
No	No	No	0	7	23
No	No	Yes	0	13	27
No	Yes	No	0	25	65
No	Yes	Yes	0	0	27
Yes	No	No	0	7	37
Yes	No	Yes	0	0	15
Yes	Yes	No	0	0	30
Yes	Yes	Yes	0	10	23

Table 2—The effect of all possible combinations of lime and fertilizer (L&F), vegetation control (weed) and fencing on height growth of northern red oak seedlings for two years prior to partial overstory cutting (1995 and 1996) and 1 year after partial overstory cutting (1997). Annual growth is included for comparison because initial seedlings height was variable

			Total height				Annual growth		
L&F	Weed	Fence	Initial	1995	1996	1997	1995	1996	1997
			---------------*Cm*---------------						
No	No	No	14.0	14.9	15.7	13.9	0.9	0.8	-1.8
No	No	Yes	12.2	13.1	13.8	16.2	0.9	0.7	2.4
No	Yes	No	12.8	13.8	13.0	12.0	1.0	-0.8	-1.0
No	Yes	Yes	14.1	15.1	17.7	21.5	1.0	2.6	3.8
Yes	No	No	12.6	13.8	15.9	15.4	1.2	2.1	-0.5
Yes	No	Yes	12.0	12.2	14.2	18.1	0.2	2.0	3.9
Yes	Yes	No	14.1	14.1	15.5	14.5	0.0	1.4	-1.0
Yes	Yes	Yes	14.0	14.8	17.1	20.8	0.8	2.3	3.7

result suggested that deer browsing was not the only mechanism responsible for slow growth of these seedlings.

Vegetation control had no effect on height growth prior to cutting (1.1±0.06 cm for both). After cutting, seedlings that received vegetation control added more height growth (1.4±0.1 cm) than those that did not receive vegetation control (1.0±0.07 cm); ($P<0.05$) (Table 2). While these seedlings were responsive after cutting to release from aboveground competition by ferns, the magnitude of the response was small. Possibly, below ground competition was responsible for the limited response of these seedlings. Hippensteel and Bowersox (1995) found a non-specific competition between white ash (*Fraxinus americana* L.) and hayscented fern even when the fronds were removed. Likewise, Lyon and Sharpe (1996) reported below ground competition between northern red oak seedlings and hayscented ferns. While below ground competition was not tested, herbicide treatment as opposed to the frond removal treatment used in this study may generate a different response by eliminating below ground fern competition.

A vegetation control by fencing interaction was found for seedlings in 1996 and 1997. The seedlings in unfenced plots that received vegetation control grew the most slowly overall. These seedlings also experienced the highest rates of mortality. We believe that these slow growth rates and high rates of mortality are due to increased deer browsing.

Seedlings that were limed and fertilized grew more quickly prior to cutting (1.3±0.05 cm) and after cutting (1.5±0.06 cm) than those that were not limed or fertilized (0.9±0.06 cm and 0.9±0.05 cm, respectively); ($P<0.05$) (Table 2). While a response to soil amendments was found for these seedlings, its magnitude was small. These results are puzzling, since, in a companion study with planted 2-0 northern red oak nursery stock, similar treatment produced mean total 3 year growth of 61.8 cm (Sharpe and others 1998). Additional research will be necessary to ascertain why these differences were observed.

Overall, while these seedlings were responsive to all treatments, even the best possible combinations of treatments resulted in less than 4 cm of annual height growth. This rate is far from adequate. All of the factors that we addressed increased growth; however, it was apparent that additional requirements must be met to achieve a suitable rate of height growth.

Leaf Area

During the entire experiment, fenced seedlings increased in leaf area (18±6 cm^2; 64±10 cm^2, respectively) and unfenced seedlings lost leaf area (-135±20 cm^2; -180±25 cm^2, respectively) (Table 3). The factors of vegetation control and of lime and fertilizer were of limited importance to leaf area. Since deer browsing was responsible for negative height growth, reduction in leaf area would also be expected.

CONCLUSIONS

Without any other site treatment, vegetation control increased rates of mortality of natural northern red oak seedlings. Lime and fertilizer treatment increased height growth before and after cutting. Vegetation control increased height growth after cutting but had no effect on height growth prior to cutting. Fencing increased seedling height growth prior to and after cutting. Seedlings that were not fenced actually lost height in the year after cutting. While this investigation revealed statistically significant height growth gains in response to the treatments employed, the magnitude of these responses was low. Since the best growth rates were still less than 4 cm per year, other factors must have also limited height growth of these seedlings. These factors may have included but were not limited to light, soil moisture, other limiting nutrients and insect herbivory.

REFERENCES

Cronan, C.S.; Grigal, D.F. 1995. Use of calcium/aluminum ratios as indicators of stress in forest ecosystems. Journal of Environmental Quality. 24: 209-226.

Demchik, M.C. 1998. The influence of soil nutrient conditions in northern red oak stands on litter decomposition, tree growth and tree mortality and the effects of soil nutrient amendments on tree growth and stump sprouting on Pennsylvania's Laurel Hill. PA: The Pennsylvania State University. 158 p. Ph.D. dissertation.

Demchik, M.C.; Sharpe, W.E. 1996. *Quercus rubra* decline in southwestern Pennsylvania. In: Proceedings 77th meeting of the Ecological Society of America; August 10-14; Providence, RI: 41.

DeWald, L.E.; Sucoff, E.I.; Ohno, T.; Buschena, C.A. 1990. Response of northern red oak (*Quercus rubra* L.) seedlings to soil solution aluminum. Canadian Journal of Forest Resources. 20: 331-336.

Hart, D.H. 1995. Response of potassium deficient red maple seedlings and stump sprouts to fertilization and the effect of soil acidity on northern red oak regeneration success. PA: The Pennsylvania State University. 179 p. M.S. thesis.

Hart, D.H.; Sharpe, W.E. 1997. Response of potted northern red oak and hay-scented fern to additions of calcium, magnesium, potassium and phosphorus. In: Proceedings 11th central hardwood conference; Columbia, MO: 304-312.

Table 3—The effect on total leaf area (cm^2) of all possible combinations of lime and fertilizer (L&F), vegetation control (weed) and fencing on leaf area growth of northern red oak seedlings for two years prior to partial overstory cutting (1995 and 1996) and 1 year after partial overstory cutting (1997). Annual growth is included for comparison because initial seedlings height was variable

			Total leaf area			Annual growth	
L&F	Weed	Fence	1995	1996	1997	1996	1997
			Cm^2				
No	No	No	227	162	82	-65	-80
No	No	Yes	99	99	172	0	73
No	Yes	No	113	109	76	-4	-33
No	Yes	Yes	160	161	288	1	127
Yes	No	No	150	129	108	-21	-21
Yes	No	Yes	144	146	189	2	43
Yes	Yes	No	175	130	84	-45	-46
Yes	Yes	Yes	142	209	223	67	14

Hippensteel, T.E.; Bowersox, T.W. 1995. Effects of hayscented fern density on light and white ash seedling growth. In: Proceedings, 10th central hardwood conference; Morgantown, WV: 256-270.

Horsley, S.B. 1993. Mechanisms of interference between hay-scented fern and black cherry. Canadian Journal of Forest Resources. 23: 2059-2069.

Joslin, J.D.; Wolfe, M.H. 1989. Aluminum effects on northern red oak seedling growth in six forest soil horizons. Soil Sciences Society of America Journal. 53: 274-281.

Lyon, J. 1996. The impacts of soils, herbaceous competition and soil liming on the growth and nutrition of northern red oak (*Quercus rubra* L.) seedlings. PA: The Pennsylvania State University. 169 p. Ph.D. dissertation.

Lyon, J.; Sharpe, W.E. [In press]. An assessment of Ca/Al ratios of selected Pennsylvania forest soils. Water, Air and Soil Pollution.

Lyon, J.; Sharpe, W.E. 1995. Impacts of electric deer exclusion fencing and soils on plant species abundance, richness and diversity following clearcutting in Pennsylvania. In: Proceedings, 10th central hardwood forest conference; Morgantown, WV: 47-59.

Lyon, J.; Sharpe, W.E. 1996. Hay-scented fern (*Dennstaedtia punctilobula* (Michx.) Moore) interference with growth of northern red oak (*Quercus rubra* L.) seedlings. Tree Physiology. 16: 923-932.

Marquis, D.A.; Brenneman. R. 1981. The impact of deer on forest vegetation in Pennsylvania. Gen. Tech. Rep. NE-65. U.S. Forest Service. 7 p.

McIntyre, A.C. 1936. Sprout groups and their relation to the oak forest of Pennsylvania. Journal of Forestry. 34: 1054-1058.

McQuilkin, R.A. 1983. Silvical factors affecting the regeneration of oaks and associated species in Pennsylvania. In: Proceedings regenerating hardwood stands. University Park, PA: Pennsylvania State University.

McWilliams, W.H.; Bowersox, T.W.; Gansner, D.A. [and others]. 1995. Landscape-level regeneration adequacy for native hardwood forest of Pennsylvania. In: Proceedings, 10th central hardwoods conference; Morgantown, WV: 196-203.

Phares, R.E. 1971. Growth of red oak (*Quercus rubra* L.) seedlings in relation to light and nutrients. Ecology. 52: 669-672.

Pickens, C.J. 1995. Early indicators of acidification: a whole watershed approach to studying forest response to additions of nitrogen and sulfur. PA: The Pennsylvania State University. 109 p. Ph.D. dissertation.

Sharpe, W.E.; Swistock, B.R.; DeWalle, D.R. 1993. A greenhouse study of northern red oak seedling growth on two forest soils at different stages of acidification. Water, Air and Soil Pollution. 66: 121-133.

Sharpe, W.E.; Swistock, B.R.; Mecum, K.A.; Demchik, M.C. 1998. Greenhouse and field growth of northern red oak seedlings inside different types of tree shelters. Submitted to Northern Journal of Applied Forestry.

Tilghman, N.G. 1989. Impacts of white-tailed deer on forest regeneration in northwestern Pennsylvania. Journal of Wildlife Management. 53: 524-532.

JAPANESE AND GIANT KNOTWEED SEED REPRODUCTIVE ECOLOGY

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Abstract—Japanese knotweed (*Polygonum cuspidatum* Sieb and Zucc.) and Giant knotweed (*Polygonum sachalinensis* F. Schmidt ex Maxim) are invasive, exotic perennials. They were introduced from eastern Asia as ornamental plants. These species were found throughout Pennsylvania especially along riparian, roadway, and railway corridors, and in waste areas. Japanese and Giant knotweed propagate asexually by rhizomes and basal buds. Researchers in the United Kingdom reported no viable seed set on Japanese knotweed due to the absence of male-fertile plants. In New Jersey, seedlings were found under Japanese knotweed. This research project was conducted to determine if the prolific seed produced within Pennsylvania was viable and capable of germinating in nature.

The reproductive ecology of Japanese and Giant knotweed seed was studied in eighteen populations from three river drainages across Pennsylvania. Field investigations showed that Japanese and Giant knotweed plants produce millions of seed annually. Two-year-old seedlings were discovered at four study sites in 1997 and 1998. Apparent viability assessments of seeds were made using seed coat and endosperm conditions. Seed coat was not a consistently good indicator of seed viability. Germination requirements were determined by testing the effects of moisture, temperature, and wing treatments on four seed sources over 30, 75, and 120 day storage periods. The temperature and moisture treatment combinations were all sufficient for germination. The overall mean germination rates were above 84 percent. Significantly different results were achieved with cold (3 degrees Celsius) moist conditions, yielding 95 percent mean germination over all storage times. Seedbank studies were conducted to determine the existence of naturally occurring knotweed seedbeds at four sites. The range of seedlings per soil core was 0-120. This equates to greater than 5,000 potential seedlings per square meter at the Rothrock and Lumber Road sites. Japanese and Giant knotweed were capable of reproducing sexually in these experiments.

INTRODUCTION

Japanese and Giant knotweed have established populations in a variety of habitats across Pennsylvania. They grew especially well in riparian zones, on strip-mine spoils, and on roadway and railway fill material. Their capabilities to out-compete and to displace other vegetation were obvious. Long stretches of the Susquehanna and Delaware River corridors were covered by both species. Large stands existed in the Allegheny Portage Railroad National Historic Site Staple Bend Tunnel Unit and Delaware Water Gap National Recreation Area. Japanese and Giant knotweed propagated vegetatively by extensive root systems. Researchers in the United Kingdom reported no viable seed on Japanese knotweed due to the absence of male-fertile plants. The small quantities of seed that were found were the result of hybridization with other members of *Polygonaceae*. A researcher in New Jersey reported seedlings under stands of Japanese knotweed (Locandro 1973). The observed prolific seed set on populations in the Northeastern United States warranted further investigation into the potential for reproduction from seed.

Japanese and Giant knotweed are rhizomatous perennials that form dense clumps and reach heights of 2.5 and 4 meters, respectively. Plants develop woody stocks with thick, bamboo-like, hollow stems that are erect and branched at the top. Perennating buds form at the base of the stem and on woody rhizomes between autumn and winter, and emerge as vertical shoots the following spring (Beerling and others 1994). Populations usually consist of parent plants called ramets. The vegetative shoots that grow from the ramets are called genets (Richards 1986). Stocks can produce prolific, creeping rhizomes within one year. Japanese knotweed leaves are truncate at the base, 5-15 centimeters long and 2-10 centimeters wide (Mitchell and Dean 1978). Giant knotweed leaves are cordate based, up to 40 centimeters long, and 22 centimeters wide. Japanese knotweed is functionally dioecious in the United States. It is gynodioecious (male-sterile and hermaphroditic individuals) in the United Kingdom (Beerling and others 1994). Giant knotweed is dioecious, possibly polygamous when hermaphroditic flowers are produced (Bailey 1994). Achenes are dark, glossy brown, 2-4 millimeters long and 2 millimeters wide. The hybrid between Japanese and Giant knotweed, *Fallopia x bohemica* (Chrtek & Chrtková)J.Bailey, is intermediate in growth habit and leaves (Bailey and Stace 1991).

Germination research on Japanese knotweed seed has been conducted in the United States. Germination of 84-100 percent was obtained after storage in air-dryness at room temperature for five months, and at 2-4 degrees Celsius in water, in moist peat, and between layers of moist cotton for three months (Justice 1941). Light versus dark tests resulted in higher percentages of seeds germinating in lighted conditions (61 percent) as opposed to dark conditions (13 percent) (Seiger 1993). However, Locandro (1973) found no significant differences due to

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light and temperature conditions in Japanese knotweed seed germination. Germination trials were conducted without light and exhibited high percentages of germination.

Seiger (1993) had higher germination rates with two-year-old seeds (54 percent) than with two-week-old seeds (3 percent). Locandro (1973) found a decline in percent germination after 270 days in storage. Significant differences were noted due to the different storage and cold treatment periods, prior to germination trials. Locandro (1973) reported that germination continued to increase with longer periods of cold treatment. Germination was poor after one year of seed storage at 18 degrees Celsius and 32 percent humidity. Cold treated seed reacted differently than non-cold treated seed after 30, 60, and 90 days of storage. Germination increased with pre-chill treatments of 60 and 90 days (Locandro 1973).

It was not known if Japanese and Giant knotweed growing within Pennsylvania was capable of producing viable seed, and if the potential seedlings were capable of becoming established under natural conditions. The specific objectives of this study were: to determine the potential for seed reproduction from Japanese and Giant knotweed populations, and to evaluate the potential for the natural establishment of knotweed seedlings beneath or adjacent to seed producing populations in Pennsylvania.

METHODS

Field Sites

Eighteen field sites containing populations of Japanese and Giant knotweed were chosen across Pennsylvania. Figure 1 shows the location of the sites and the watershed designations. The majority of locations were within or near these National Park Service properties: Allegheny Portage Railroad National Historic Site, Delaware Water Gap National Recreation Area, and Valley Forge National Historic Site.

The sites were in the Delaware, Ohio, and Susquehanna River watersheds. Descriptions of the habitat and morphology of the knotweed plants were recorded over two years. Four of the seven sites in the Susquehanna River drainage were within 30 meters of water. The Delaware River drainage sites were within 5 meters of water, except Bushkill Church which was on abandoned fill. The Ohio drainage populations were in mixed settings. Johnstown and Saint Michael were within 10 meters of a stream, while Staple Bend was approximately 150 meters from a river. Bitumen and Lumber Road were in very remote forest locations on roadway material (gravel and soil mixed).

Flower development, sex, and the general location of male and female plants were recorded in July and August at each site. At that time, differences in leaf size and shape, and stem height were recorded to aid in species

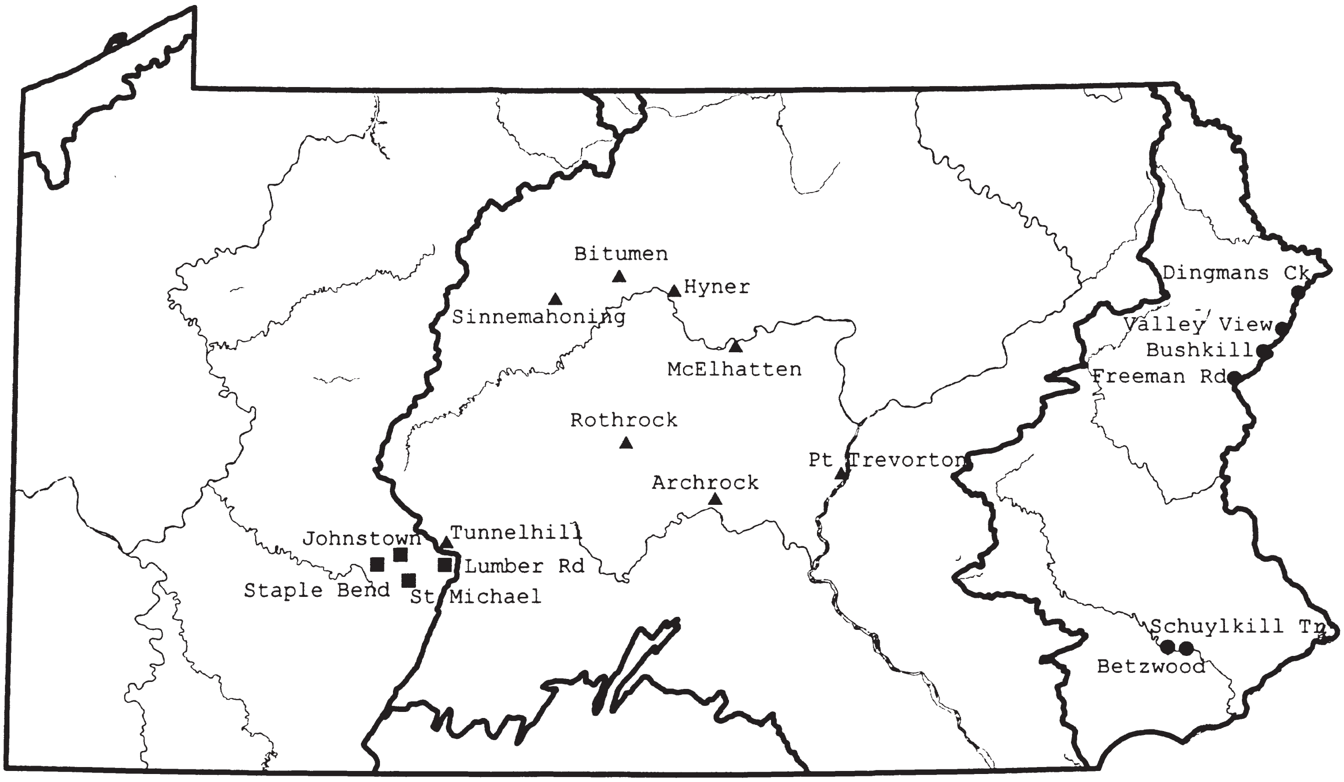

Figure 1—Locations of field sites in the Ohio River (squares), Delaware River (circles), and Susquehanna River (triangles) watersheds.

identification. The most reliable characteristics for the identification of the species and the hybrid were leaf shape and size. Only mature basal leaves were used because immature and upper leaves were extremely variable (Bailey 1988b). Botanical keys for the identification of *Polygonum* species differentiated between Japanese and Giant knotweed by truncate and cordate leaf bases (Mitchell and Dean 1978, Wofford 1989). Knotweed seeds were collected in September and October of 1996 and 1997. The amount of seed produced at each site varied widely. Single stems produced 50,000-150,000 seeds annually. Millions of seeds were produced over a 10 square meter area at locations such as Lumber Road, Staple Bend, and Rothrock. Seed collections were random samples of whole inflorescences from different plants. Seeds were air dried, and stored with either the wing intact (natural seed) or dewinged (achene only).

Apparent Viability

Visual assessments of seed viability were based on seed coat and endosperm conditions. Estimates of viability were made on random samples of 100 seeds from each source collected in 1996. The seeds were inspected under a dissecting scope. The conditions of the seed coat and endosperm were evaluated. Seed coats were described as normal (hard), soft, or shrunken. Endosperm content was estimated as full, half, or absent, and the color was classified as cream or white. Similar seed viability assessments were made on the 1997 seeds using 25 seeds per source. Percent viability was based on the number of seeds having normal seed coats and full endosperm. Tetrazolium chloride tests on endosperm viability were not used because Locandro (1973) reported poor results with the method.

Germination Studies

Four seed sources (Rothrock, Staple Bend, Port Trevorton, and Bitumen) were used to test different seed storage regimes. The seeds used in the germination trials were stored at room temperature for 100 days after collection. To test the effect of wing inhibition, seeds were either left natural (wing intact) or dewinged (achene only) by means of friction. The presence of the papery wing could inhibit both water absorption by the seed coat and radicle emergence. The seeds were stored for 30, 75, and 120 days under three different environmental conditions: warm-dry, cold-dry, and cold-moist (18, 3, and 3 degrees Celsius, respectively) (Table 1). Warm-moist, 18 degrees Celsius, conditions resulted in immediate germination of seeds. Refrigerated seeds were placed in plastic bags containing sand moistened with distilled water. All bags were then placed in glass jars and sealed. Sterile petri dishes, 9 centimeters in diameter, were lined with 9 centimeter diameter, grade 4 filter paper. Random samples of 40 seeds were placed into 96 dishes and covered with another filter paper. Three milliliters of distilled water was poured over the filter paper to provide moisture. Each dish was placed in a quart size plastic bag with a wet paper towel as a supplemental moisture source. Twenty four dishes were arranged in randomized block designs on 4 trays (4 replications). The trays were placed in a growth chamber at 27 degrees Celsius.

Light was not provided in the growth chamber. The basis for determining if a seed had germinated was radicle and cotyledon emergence. The number of germinates was recorded in each dish every three days over a fourteen day period. The seeds were exposed to fluorescent light during the counts. The number of seeds used in each environmental condition was 3,840, for a total of 11,520.

The percent germination values were log transformed. Analysis of Variance (alpha=.05) was run to test the effects of seed source, seed condition, environmental condition, and all of the interactions on seed germination. Tukey’s HSD mean separation procedures were used to determine significant differences (alpha=.05).

Viability Germination Trials

The reliability of using seed coat condition as an indicator of seed quality was tested on seeds collected in 1997. All seeds had been stored under dry, room temperature conditions for 60 days prior to testing. Random samples of 50 seeds were taken from each source. The seed coats were determined to be either normal or shrunken. The seeds were separated as such and put between layers of wetted filter paper in petri dishes. The growth chamber conditions were 27 degrees Celsius and 66 percent humidity. The number of germinates was recorded every four days for a sixteen day period. Chi Square 2-way Test of Independence (Little 1978) was used to evaluate the ability to determine seed viability based on seed coat condition.

Table 1—Experimental design of seed germination trials conducted in 1997 on Giant and Japanese knotweed seed collected in 1996

Variables	Number	Treatments
Seed sources	4	Bitumen, Staple Bend, Rothrock, Port Trevorton
Seed conditions	2	Whole seed, achene only
Environmental conditions	3	Warm-dry, cold-dry, cold-moist
Storage times	3	30, 75, 120 days
Replications	4	

Soil Core Studies

Two hundred soil cores from four study sites, known to have produced viable seed in the Fall of 1996, were collected in the Spring of 1997. Staple Bend, Rothrock, Lumber Road, and Bitumen sites were used in this study. Two hundred soil cores (15 centimeters diameter, 10 centimeters deep, 182 square centimeters) were transported to a greenhouse and placed into commercial potting media to ensure adequate moisture. The cores were collected with intact (minimally disturbed) soil profiles. The randomized block design had five replications of ten cores per soil source. The number of knotweed seedlings that emerged was recorded from May 25 to August 15. The June 20 data was used for analysis, since there were no net changes in the number of seedlings per core after that date. The total number of seeds within the soil was not counted.

RESULTS AND DISCUSSION

Field Sites

The natural variation that occurs within a plant community, phenotypic plasticity, made identification of species difficult. Mature basal leaves were reliable indicators of species and were used to identify Japanese and Giant knotweed and the hybrid. Differences in stem height, leaf shape, and leaf size were recorded at each site.

Table 2 summarizes the locations and morphological characteristics of the plants found at the sites. There were 10 Japanese knotweed sites, 7 Giant knotweed sites, and 1 hybrid site. The Japanese knotweed populations had maximum heights of 3 meters. The tallest Giant knotweed plants were located at Staple Bend, Sinnemahoning, and Bitumen, all of which had maximum heights of 4 meters.

An abundance of seeds were produced when both female and male- fertile plants were present at a site. The intermingling of the two plants resulted in higher quantities of seed than if the plants were spatially separated. The amount of seed produced is expressed as seed crop values based on an average of 1,000 seeds per raceme (Table 2). Plants at every study sites produced seeds during the first year of research. No viable seeds were found when male-fertile plants were absent from the site. No viable seeds were found at the Valley View and Archrock sites in 1996, all of the seeds were empty. No seeds were found at the Schuylkill Trail site in 1997.

Table 2—Species, colony height, sex ratio, seed crop, and Chi Square values for Giant and Japanese knotweed study sites in the Ohio, Susquehanna, and Delaware River watersheds

Watershed location	Knotweed species	Colony height	Seed crop[a]	Viability[b] 1996	Viability[b] 1997	Sex ratio	Chi square[c]
		m	*Percent*	*Percent*	*Percent*		
Ohio River							
Johnstown	Giant	3.0	90	95	90	F=M	—-
Lumber Road	Giant	4.0	90	100	85	F=M	0.72
St. Michael	Giant	3.0	90	95	75	F=M	7.53*
Staple Bend	Giant	4.0	90	95	90	F<M	0.23
Tunnel Hill	Giant	3.5	90	100	90	F<M	1.89
Susquehanna River							
Archrock	Japanese	3.5	1	0	40	F	14.00*
Bitumen	Giant	4.0	90	90	90	F<M	1.98
Hyner[d]	Japanese	2.7	70	60	80	F<M	7.53*
McElhatten[de]	Hybrid	4.0	90	80	-	F>M	—-
Port Trevorton[e]	Japanese	4.0	90	85	95	F>M	—-
Rothrock	Japanese	3.5	90	85	100	F=M	—-
Sinnemahoning	Giant	4.0	90	-	95	F=M	2.62
Delaware River							
Betzwood Park	Japanese	2.5	1	80	85	F>M	6.97*
Bushkill Church	Japanese	3.3	5	95	85	F>M	3.25
Dingmans Creek	Japanese	3.5	75	90	95	F>M	—-
Freeman Road	Japanese	3.3	1	90	95	F	—-
Schuylkill Tr.	Japanese	2.0	1	65	-	F	—-
Valley View	Japanese	2.5	1	2	10	F	8.20*

[a] Based on a fully developed raceme with about 1,000 seeds. Seed crop values were the same in 1996 and 1997.
[b] Viability was based on visual assessments of seed coat and endosperm.
[c] X^2 cv= 3.84, df= 1. Seed coat condition used as a predictor of viability. Significance is designated by the * symbol.
[d] Mostly the indicated species but the other species or a hybrid of the two species was observed to be in the general vicinity.
[e] Maybe the hybrid of the two species, *F. bohemica*.

The majority of populations in this study had two types of flowers. There were male-fertile (male) and female-fertile (female) flowers on separate plants. Male-fertile flowers were white, had 7-8 long stamens, and a vestigial, small, yellow ovary. The ovaries on male-fertile flowers were presumed nonfunctional because no seeds were produced. Male inflorescence stems pointed upwards. Male flowers were fragrant and had a feathery appearance. Male flowers were bright white, while female flowers were yellow-green. The female flowers had a larger, yellowish ovary with feathery stigmas, and short stamens around the base. The vestigial stamens in female-fertile flowers were presumed non-functional, due to the absence of seed set when male-fertile flowering plants were absent. The female plants were not able to fertilize each other. The inflorescences were arrayed outward or down. Female plants drooped with the weight of abundant, mature seed. Viability tests were not conducted on the two types of flowers to determine if the vestigial (shorter) sex organs were functional. Flower maturation and pollination were not simultaneous in all populations. The sites that were higher in elevation had earlier development of flowers and seeds. The third flower type, hermaphroditic, was found at a few locations. The pistil and stamens were proportionate in size with the stigmas and anthers equal in height. There were occasional breakdowns in dioecy, because seeds were found on male-fertile plants at Sinnemahoning and Staple Bend.

In the field there were instances of achene separation from the perianth prior to dispersal. It was not known if the seed naturally fell out or if it was consumed. Predation by birds may be a factor in rarity of seedling germination and establishment in the wild (Beerling and others 1994). The abscission zone in the pedicel may be prematurely inactivated by the first hard frost, preempting dispersal by wind (Beerling and others 1994). For this reason, the dewinged seed condition was tested in our seed germination trials.

The flowers were pollinated by bees, ants, butterflies, and beetles. Insect damage was seen in some inflorescences where the seed coat was partially eaten and the endosperm gone. It was believed that damage was done by invertebrates. We observed slugs eating Giant knotweed seedling cotyledons in another study (Niewinski 1998).

Tens to thousands of newly germinated seedlings were seen at Bitumen, Dingmans Creek, Lumber Road, Port Trevorton, Rothrock, St. Michael, Sinnemahoning, and Staple Bend. Differences in germination and survival of seedlings were found between these two areas: the ground underneath the clone, and the area beyond the perimeter of the clone. Those seedlings that had germinated under the canopy of the parents and in the leaf litter did not survive. Thousands of seedlings survived to the end of the first growing season. Some were more than 10 centimeters tall. Second-year seedlings of Giant knotweed were discovered at Bitumen and Sinnemahoning in 1997 and at Bitumen, Port Trevorton, and Lumber Road in 1998. These seedlings had completely independent root systems, and had remnant stems from the previous year's growth.

Apparent Viability Assessments

There was little variation in color and size of seeds among populations over the two years. Giant knotweed had slightly larger seeds, about 1 millimeter longer than Japanese knotweed seeds. Seed conditions ranged from firm seed coat with slightly convex sides and a cream endosperm to shrunken seed coats with a depleted or absent endosperm. Like the seed coat, the endosperm was generally convex trigonal. The outer cells were either green or tan. As the endosperm desiccated it turned chalky white. Some seeds were empty shells with brown crystals inside. The cream colored endosperm may hold more favorable seed reserves than the white color. The white colored endosperm appeared to be desiccating, resulting in a slight separation of the embryo from the endosperm. We were not able to test the differences between colors. Seed production in a population was related to the number and proximity of male and female-fertile plants. Endosperm absence was found when no male-fertile plants grew near the study sites.

Viability values for each site, based on the seed coat and endosperm conditions, are given in Table 2. In 1996 and 1997, Betzwood Park, Bitumen, Bushkill Church, Dingmans Creek, Freeman Road, Johnstown, Lumber Road, Port Trevorton, Rothrock, Staple Bend, and Tunnel Hill sites had viability values above 80 percent. Seed sources with the lowest viability (0-40 percent) were Archrock and Valley View. These sites produced endosperm that was only partially developed or absent. Seed coat condition was not the most accurate method of determining potential viability. There may be better methods of assessing endosperm, like water floatation or X-ray analysis.

Germination Trials

The four seed sources chosen had shown high potential viability in the visual assessments. Cold stratification was not necessary to induce germination. Seed samples stored at room temperature germinated within five days when given water.

The overall potential of seeds to germinate was 91 percent. Analysis of Variance (alpha=.05) indicated significant differences in germination due to seed source, seed conditions, and environmental conditions over time. Tukey's HSD mean separation procedures indicated significant differences among treatments. After 30 days storage, Bitumen and Port Trevorton seeds had significantly different germination percentages (96 and 93 percent germination, respectively) than Rothrock and Staple Bend (88 and 86 percent, respectively) (Table 3). After 75 and 120 days, Bitumen seeds germinated at significantly different rates (97 and 98 percent, respectively) than Rothrock (88 and 90 percent) and Staple Bend (88 and 87 percent). Average germination among seed sources and over all seed and environmental conditions were significantly different. The results were Staple Bend 88 percent, Rothrock 89 percent, Port Trevorton 92 percent, and Bitumen 96 percent. The range

Table 3—Mean[a] germination percentages for seed source, seed condition, and environmental condition factors after 30,75, and 120 days storage on Giant and Japanese knotweed seed. N= 11,520 seeds used total

Factors	Storage time (days) 30	75	120	Overall avg.
	------------------------ *Percent*	------------------------		
Seed source (SS)				
Staple Bend (SB)	86.0b	87.4b	86.9c	86.8c
Rothrock (RR)	88.1b	87.6b	90.0bc	88.6c
Port Trevorton (PT)	92.6a	91.8ab	92.8ab	92.4b
Bitumen (BT)	95.7a	96.4a	97.9a	96.7a
Seed condition (SC)				
Natural	89.9a	88.9b	90.0b	89.6b
Dewinged	91.3a	92.7a	93.8a	92.6a
Environmental condition (EC)				
Warm-dry	89.8b	86.7b	89.9b	88.8b
Cold-dry	87.4b	90.5b	89.8b	89.2b
Cold-moist	94.6a	95.2a	96.1a	95.3a
SS * EC				
SB * warm-dry	83.6bc	84.7b	83.8b	84.1c
SB * cold-dry	82.8c	86.6b	82.8b	84.1c
SB * cold-moist	91.6ab	90.9ab	94.1ab	92.2b
RR * warm-dry	85.3bc	80.0b	84.1b	83.1c
RR * cold-dry	80.9c	85.3b	88.8b	85.0c
RR * cold-moist	98.1a	97.5ab	97.2ab	97.6a
PT * warm-dry	95.3ab	88.1b	93.8ab	92.4b
PT * cold-dry	90.3b	94.1ab	90.3ab	91.6b
PT * cold-moist	92.2ab	93.1ab	94.4ab	93.2ab
BT * warm-dry	95.0ab	94.1ab	97.5a	95.5ab
BT * cold-dry	95.6ab	95.9ab	97.5a	96.4ab
BT * cold-moist	96.6ab	99.1a	98.8a	98.1a

[a] Means within a factor (e.g. seed source) and under a storage time with the same lower case letter are not significantly different at the .05 level.

of germination inclusive of sources and different conditions was 60-100 percent.

The dewinged seed condition had significantly different germination percentages after 75 and 120 days storage (93 and 94 percent, respectively) than the natural seed condition (89 and 90 percent) (Table 3). Initially we thought that this might be important, however we found the dewinged seed condition was not a critical factor in nature, because seed rarely dispersed without the wing attached.

The three environmental conditions tested, warm-dry, cold-dry, and cold-moist, were all sufficient for germination (Table 3). Germination increased with time and reached 98 percent for the cold-moist treated seeds after 120 days storage. The cold-moist storage condition simulated Mid-Atlantic winter weather. The overall mean germination percentages for the environmental conditions over storage times were: warm-dry 88 percent, cold dry 90 percent, and cold-moist 95 percent (Table 4). Over the three storage times, germination percentages for seeds stored under cold-moist conditions were significantly different than seeds stored under the warm-dry and cold-dry conditions.

Some of the variability in seed germination was due to a significant interaction between seed source and environmental condition (Figure 2). Staple Bend and Rothrock had one pattern of response, while Port Trevorton and Bitumen had another pattern. The Staple Bend-Rothrock pattern showed statistically significant germination differences between cold-moist and warm-dry/cold-dry conditions. For the Port Trevorton-Bitumen pattern the differences among treatments could not be separated out. The patterns were not attributed to species differences because seeds collected from Staple Bend and Bitumen were Giant knotweed, while those from Rothrock

Table 4—Overall mean[a] percent seed germination for storage times and environmental conditions among all seed sources and seed conditions. N = 3,840 seeds used per storage time

Environmental conditions	Storage time (days) 30	75	120	Overall
	------------------	Percent	-----------------	
Warm dry	88b	88b	90b	88b
Cold dry	88b	90b	90b	90b
Cold moist	95a	95a	98a	95a

[a] Values within each storage time with the same lower case letter were not significantly different at the .05 level.

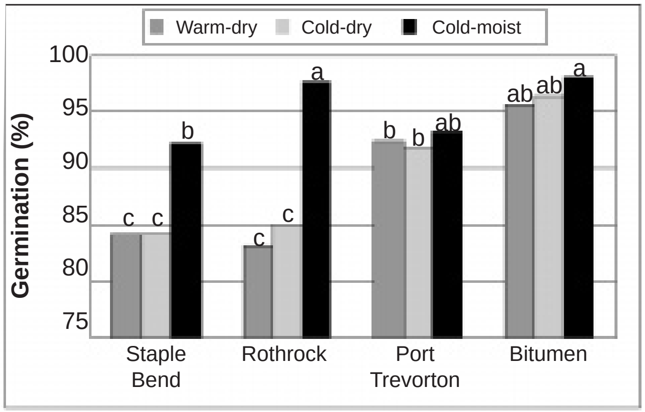

Figure 2—Overall mean percent germination for Giant and Japanese knotweed seed under three environmental conditions. N= 960 seeds per source under each environmental condition. Values among all seed sources and storage times with the same lower case letter are not significantly different at the .05 level.

and Port Trevorton were Japanese knotweed. Since collection was done in October, all seed had similar maturity. The plants growing at Bitumen produced seed with the highest percent germination under the cold-moist storage conditions.

Although germination varied due to a few factors, overall percentages were high. The different combinations we tested provided valuable evidence that Japanese and Giant knotweed were capable of reproducing sexually in the United States.

Viability Assessment Germination Trials

Seeds with shrunken seed coats were believed to be those in which the endosperm had not been fully developed. The null hypothesis was that seed coat condition, normal or shrunken, made no difference in germination. Chi Square analysis (alpha=.05, df=1, critical value = 3.84) indicated significant differences in germination percentages for these five sources: St. Michael, Archrock, Hyner, Betzwood Park, and Valley View (Table 2). The null hypothesis was rejected for these seed sources, because seed coat condition did make a difference in germination. Seeds with shrunken seed coats had significantly lower germination percentages. Germination percentages for Lumber Road, Staple Bend, Tunnelhill, Bitumen, Sinnemahoning, and Bushkill Church were not significantly different between normal and shrunken seed coat conditions. The null hypothesis was not rejected for these six seed sources.

Half of the seeds performed as expected, in that seeds with the shrunken coat condition did not germinate. However, some shrunken seeds germinated at high percentages. The explanation is probably linked to the population structure and the relative proportion of plants of different sexes. A common factor among the sources that performed as expected was the ratio of female:male plants. Japanese knotweed plants at Hyner, Valley View, Archrock, and Betzwood Park exhibited shrunken seed coats and all had mostly female or only female parent plants. Bushkill Church had more female Japanese knotweed plants than male. The remainder of the sites had Giant knotweed, of these, Staple Bend, Tunnelhill, and Bitumen, had a greater proportion of male plants. Lumber Road and Sinnemahoning had equal numbers of male and female plants. The abundance of male plants was integral, because of increased chances of pollination. Seed coat condition was not a consistently good descriptor of seed quality, and other methods should be explored.

Soil Core Study

Soil cores were placed into a greenhouse on May 20. Germination of the knotweed seeds began May 27 and culminated on July 2. The mean number of seedlings per core by June 21 were Bitumen 4, Lumber Road 24, Rothrock 32, and Staple Bend 5. The mean number of potential seedlings per square meter were extrapolated from this data (Figure 3). The range of seedlings in the cores was 0-120, or equivalent to greater than 6,500 seedlings per square meter). Analysis of Variance (alpha=.05) showed that the mean number of seedlings from soil samples taken at Rothrock and Lumber were significantly different than those from Staple Bend and

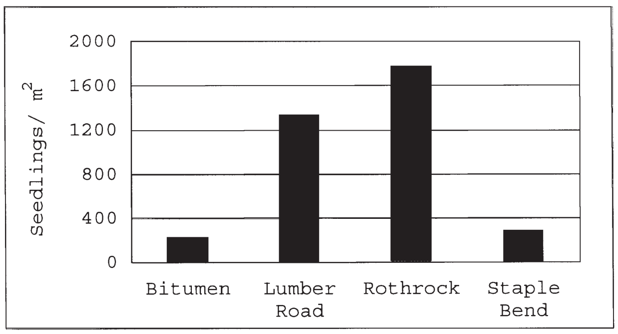

Figure 3—Mean number of potential seedlings/ m^2 at the four study sites, extrapolated from the number of seedlings per 182 cm^2 soil core. N= 50 cores per soil source.

Bitumen. There were no significant differences between Staple Bend and Bitumen or between Rothrock and Lumber Road (Table 5).

The majority of seedlings emerged from the sides of the core, where the soil profile had been slightly disturbed during collection and placement. The soil was not mixed to promote germination.

The results of Bitumen cores were lower than expected, based on the viability assessments and germination data. This may be attributed to the difficulty of collecting an intact soil core. Gravel in the strip-mine spoil made it nearly impossible to get an intact soil core close to the knotweed. The low number of seedlings could be due to a reduced amount of soil collected, or to increased distance from parent plants.

Rothrock soil cores had significantly different numbers of seedlings than those from Staple Bend, although there were no significant difference in germination potentials for the two sources of seed. Soil quality and abundance of seed-eating invertebrates may be contributing factors to the differences in number of seedlings emerging from the soil cores. Staple Bend and Rothrock soil cores had thick humus layers, while Bitumen and Lumber Road cores had little humus and came from gravel road areas.

There was a naturally occurring, viable seedbank at these four locations. Seedling mortality was expected because of the reported absence of seedlings in nature. However, the soil core study showed that seedlings were capable of surviving and producing flowers and seeds within one growing season.

CONCLUSIONS

Japanese and Giant knotweed were found throughout three river drainages in Pennsylvania. The populations examined were the dominant vegetation on the sites. Millions of seed were produced annually by Japanese and Giant knotweed at these locations. Independent, multiple year seedlings were discovered at four locations in the summer of 1996 and 1997.

The use of physical characteristics to determine viability was not reliable. Seed coat condition was not a consistently good indicator of seed quality. The potential of Japanese and Giant knotweed seed to germinate depended on the viability of the endosperm. Endosperm content can be visually assessed for a rough estimate of viability. A germination test is the best method for determining the sexual reproduction of a population.

The germination trials showed that Mid-Atlantic winter weather conditions were the best for seed germination. Cold-moist storage of seeds over 30, 75, and 120 days yielded the highest percent germination, averaging 95 percent overall. The mean percent germination for the four seed sources over all storage times, seed conditions and environmental conditions was 91 percent.

Seedbanks were found to be naturally occurring at four of the study sites and had the capacity to germinate in the spring and summer. Seeds in soil can go into induced dormancy and germinate under more favorable conditions, like when exposed to air after soil disturbance. Seedlings have the capacity to survive and grow into vegetative adults in one growing season. Observations during the seedbank study suggested that soil disturbance might lead to higher incidences of seedling emergence.

Japanese and Giant knotweed pose serious threats to our natural ecosystems and should be managed carefully. Seeds germinate in the spring and summer. Areas should be monitored for seedlings, especially in riparian zones and on paths where bare soil is exposed.

Table 5—Mean[a] number of Giant and Japanese knotweed seedlings/ core, range of seedlings/core, and number of potential seedlings per m^2 from four soil sources. N = 50 cores per soil source

Soil source	Seedlings	Range	Seedlings
	-------- *Per core* --------		*Per m^2*
Bitumen	4b	0-37	220
Lumber Road	24a	0-120	1319
Rothrock	32a	0-106	1758
Staple Bend	5b	0-28	275

[a] Values of seedlings per core among the soil sources with the same lower case letter were not significantly different at the .05 level.

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REFERENCES

Bailey, J.P. 1994. Reproductive biology and fertility of *Fallopia japonica* (Japanese knotweed) and its hybrids in the British Isles. In: de Waal; Child; Wade; Brock, eds. Ecology and management of invasive riverside plants. John Wiley, Ltd.: 141-158.

Bailey, J.P.; Stace, C.A. 1991. Chromosome number, morphology, pairing, and DNA values of species and hybrids in the genus *Fallopia* (*Polygonaceae*). Plant Systematics and Evolution. 180: 29-52.

Beerling, D.J.; Bailey, J.P.; Conolly, A.P. 1994. *Fallopia japonica* (Houtt.) Ronse Decraene. Biological flora of the British Isles. Journal of Ecology. 82: 959-979.

Justice, O.L. 1941. A study of dormancy in seeds of *Polygonum*. Ithaca, NY: Cornell University Agricultural Experimental Station Memoirs; 235: 1-43.

Little, T.M.; Hills, F.J. 1978. Agricultural experimentation, design and analysis. New York: John Wiley. Appendix table A.6: 309.

Locandro, Roger R. 1973. Reproductive ecology of *Polygonum cuspidatum*. Rutgers University: 17-91. Ph.D. dissertation.

Mitchell, R.S.; Dean, J.K. 1978. *Polygonaceae* (Buckwheat family) of New York State. In: Contributors to the flora of New York State. NY State Museum, Bulletin. 431: 25.

Niewinski, A.T. 1998. The reproductive ecology of Japanese knotweed (*Polygonum cuspidatum*) and giant knotweed (*Polygonum sachalinensis*) seed. The Pennsylvania State University: 31-48. M.S. thesis.

Richards, A.J. 1986. Plant breeding systems. Boston: Allen & Unwin Ltd.: 463-477.

Seiger, L.A. 1993. The ecology and control of *Reynoutria japonica* (*Polygonum cuspidatum*). George Washington University: 16-107. Ph.D. dissertation.

Wofford, B.G. 1989. Guide to the vascular plants of the Blue Ridge. Athens, GA: University of Georgia Press: 262-264.

Harvesting

SOIL DISTURBANCE AND PRODUCTIVITY FROM WIDE-TIRED SKIDDER TRIALS IN MINNESOTA ASPEN HARVESTS

Mathew F. Smidt and Charles R. Blinn[1]

Abstract—The soil disturbance and productivity of wide-tired and narrow-tired skidders was compared on aspen harvests in east-central Minnesota. Split plot comparisons showed that wide-tired skidders (73-44.00x32 tires) had cycle times 16 percent greater than the narrow-tired skidder (30.5Lx32 tires) for an average travel distance of 160 meters. The increased total cycle time for the wide-tired skidder came partly from significantly greater load building time. Since adequate traction and flotation were available throughout the harvest trials, wide-tired skidders had no advantage in any of the cycle elements. No significant difference was detected for production (tonnes/productive machine hour) partly due to small sample size. The mineral soil upland sites used in the study did not present any difficulty in operability for either skidder. The area in shallow depressions and ruts ranged from 2 to 23 percent (mean = 12 percent) with no significant difference due to tire size. During the study the application of the wide-tired skidder was affected by 1) poor truck access over haul roads and landings which restricted activity when soils were wet and 2) other production bottlenecks such as trucking limitations and volume quotas which limited operation productivity.

INTRODUCTION

In the Lake States, harvesting operations are often restricted during and following the spring snow melt (often referred to as spring break-up) because of the potential of soil damage and low harvesting productivity. Operators can be idle for as long as three months in this period. In an effort to recover productive time during this period and decrease soil impact throughout the growing season loggers are experimenting with wide tires.

A number of production trials comparing skidders with wide tires to those with conventional tires reported increased skidder productivity for wide-tired skidders on wet, level terrain (Hassan and Gupta 1988, Heidersdorf and Ryans 1986, Mellgren and Heidersdorf 1984, Novak 1988). In addition to productivity benefits, the increased mobility of skidders equipped with wide tires was also an important production benefit (Meek 1994, Mellgren and Heidersdorf 1984, Novak 1988).

Comparison of rutting in operational trials on swampy sites showed that while wide tire application reduced rut depth, the total coverage of ruts was only slightly lower for wide-tired skidders (Groot 1987, Heidersdorf and Ryans 1986, Novak 1988). Although wide-tired skidder use reduced soil compaction and maintained higher infiltration when compared to the narrow-tired skidder use, Aust and others (1991) cautioned that operation of skidders on wet sites made possible by wider tires could produce greater soil impact due to the increased site access not possible with narrow tires.

While the primary motive of wide tire use may be to decrease site disturbance, production comparisons may provide loggers and foresters with decision criteria for wide tire application. The objectives of this research were to compare a) the production of wide-tired and narrow-tired skidders on upland aspen sites and b) the soil disturbance resulting from the trials.

SITE DESCRIPTION

Research sites were established in northern Kanabec and Mille Lacs counties of east-central Minnesota. Each of the five sites were between six and eight hectares in size. The forest cover was generally aspen and mixed hardwood forest that varied slightly in composition, topography, and drainage. Tree volume, volume per acre, and stand basal area varied slightly among the sites (table 1).

The gently sloping sandy loam soils on the research sites were derived from glacial till. The soils were generally classified as having moderate drainage, but the dense till layers about 50 to 100 centimeters below the surface restricted drainage. Each site included small wetland or low areas with organic surface horizons. Site 1 had slightly better drainage than sites 2, 3, and 4. Site 5 had the poorest drainage and approximately 40 percent of the site was covered by poorly drained soils with organic surface horizons. Sites 1 and 4 had the most topographic relief (5-20 percent slope), sites 2 and 3 were gently rolling (5-10 percent slope), and site 5 was nearly flat (0-5 percent slope).

METHODS

The sites were divided into approximately equal halves or harvest areas for application of either the wide-tired or narrow-tired skidder. The harvest of sites 1 and 2 began in mid July and was completed at the end of August in 1991. In 1992 harvesting commenced on site 3 on May 25 and all sites were harvested by July 17. Efforts were made to begin harvesting as early as possible, but the poor condition of site access roads delayed harvesting in both years to the dates listed.

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Table 1—Stand variables for each harvested site

Stand variable	Site 1	2	3	4	5
Volume/tree (m^3)[a]	0.20	0.29	0.21	0.30	0.27
Merchantable trees (#/ha)[a]	449	480	380	395	296
Volume/hectare (m^3/ha)[a]	88	138	80	119	80
Basal area (m^2/ha)[b]	24.7	32.5	22.8	31.1	24.4

[a] Northern red oak was excluded. Merchantability standards were dbh > 15 cm, top diameter >10 cm (dob), and merchantable height > 5 m.
[b] Included all trees with dbh > 2.5 cm.

In 1991 the logging firm had two skidder operators, one operator for each skidder-tire combination. In 1992 a single operator was used for both skidders. All operators had extensive experience (> 5 years) in skidder operation and at least one year of experience with the wide-tired skidder.

Trees were skidded to the landing in tree-length form on all sites except site 5 where it was full-tree skidded and chipped at the landing. A John Deere (JD) 640D grapple skidder was employed for most of the study with limited use of a JD 648D grapple skidder (site 1 and 4). The narrow-tired skidder was equipped with 30.5Lx32 tires, and the wide-tired skidder was equipped with 73-44.00x32 tires. The JD 648D was used only as the narrow-tired skidder. In 1991 felling was accomplished with a JD 643 equipped with 64-34.00x26 tires in wide-tired configuration and 28Lx26 tires in the narrow-tired configuration (sites 1 and 2). In 1992 a Case tracked excavator with a feller-buncher head was used for felling all harvests (sites 3, 4 and 5).

Soil Conditions

Forty-four permanent plots were located randomly throughout each site prior to harvest for collection of soil information. On the same day that the area around the plot was skidded, soil samples were collected with a probe from undisturbed soil near the permanent plot markers from 0 to 20 centimeter and 20 to 40 centimeter depths. The soil samples were dried for 24 hours at 105° C and the gravimetric soil moisture was calculated.

The soil disturbance sampling technique was based on that described by Howes and others (1983). The permanent plot markers were used as starting points for the transects. Ten points, 3 meters apart, were located along the transect which was oriented at a random azimuth. The area below each point was categorized as undisturbed, slash covered, rutted, mineral soil exposure, water covered, rocks, or stumps. The points with possible soil compaction were categorized as shallow depressions (< 5 centimeters), moderate ruts (≥ 5 and < 15 centimeters), and deep ruts (≥ 15 centimeters). Mineral soil exposure included conditions ranging from mixed mineral soil and organic matter to complete removal of the surface organic layer.

The proportion of area compacted (shallow depressions and all ruts), moderate and deep ruts combined (≥ 5 centimeters), no disturbance, and all disturbance (all compacted classes plus mineral soil exposure) were the dependent variables modeled with a transformed logistic model (Neter and others 1985). The model included site, treatment (wide-tired or narrow-tired skidder harvest), and the interaction of site and treatment. The surface soil moisture (0 to 20 centimeters) for each sampling point was included as a covariate. The F ratio for the treatment was formed by the ratio of treatment mean square to the interaction mean square appropriate for nested designs.

Continuous Timing and Production

During the narrow- and wide-tired skidder harvests, 50 to 100 entire skidding cycles were timed midway though each harvest. The major skidding elements timed were unloaded travel, loaded travel, load building, delay, piling, and trail building. Load building included all maneuvering and hookup activities to build the load. Location of landmarks and permanent plot locations were used to estimate the loaded travel distance or travel distance(from the landing to the first bunch collected). Travel distance was collected for each cycle regardless of continuous timing. Missing data resulted in different sample size for elements. During continuous timing load size was measured only as trees per load.

The cumulative production data were collected over 2 to 5 days of production from each harvest area. To make up for the small number of sites, additional cumulative production samples were created from each site when the operation stopped completely due to circumstances like serious breakdown or poor weather conditions. A total of 15 production samples were collected. Data collected included total production (stick or weight scaled in tonnes), total productive time, number of cycles, and fuel use, and the travel distance for each cycle. From those data total cycle time, average load size, average travel distance, production per hour, and fuel consumption were calculated.

The dependent variables from continuous timing and cumulative production data were modeled using the GLM procedure (SAS Institute 1988) with the expanded model: site, treatment (tire: narrow or wide), travel distance (covariate), and the interactions tire*site and tire*travel distance. If the F value (type III sums of squares) for the interaction, tire*travel distance, was not significant ($P < 0.10$), it was deleted from the model and a reduced model was used.

RESULTS AND DISCUSSION

Soil Disturbance

Soil moisture conditions on sites 1, 3, 4, and the narrow-tired area of 5 had similar average soil moisture at both soil depths (table 2). Organic areas within the wide-tired area of site 5 had increased average soil moisture, and the August harvest date of site 2 resulted in considerably drier soil during the harvest.

Tire size was not a significant factor in any of the disturbance measures ($P < 0.10$), and site was largely the only significant term in the models. Even with moist soil conditions throughout the sites, soils provided generally adequate support for the equipment as indicated by the low coverage of deep ruts across all sites (table 3). The low level of soil disturbance on site 2 could be related to the dry soil. The low soil disturbance on site 5 was produced in part by the concentrated skidding pattern present. Relatively high levels of disturbance, especially on sites 1 and 4, resulted from the high skid trail density produced by unplanned skid trails. The change from wheeled (sites 1 and 2) to tracked feller buncher (sites 3, 4, and 5) was not obvious from the site totals, but was correlated with generally drier (sites 1 and 2) and generally (sites 3, 4, and 5) wetter soil conditions in the successive harvest years.

Continuous Timing and Production

Model P values were generally lower for the cumulative production data than for continuous timing data due in part to the smaller sample size and the increased error

Table 2—Completion date and average soil moisture data for the harvest areas harvested by the wide-tired (W) or narrow-tired (N) skidder

Site	Harvest date	Harvest area	Gravimetric soil moisture	
			0 to 20 cm	*20 to 40 cm*
1	7/91	W	0.38	0.22
		N	0.38	0.22
2	8/91	W	0.18	0.12
		N	0.20	0.14
3	6/92	W	0.39	0.26
		N	0.33	0.19
4	6/92	W	0.39	0.26
		N	0.39	0.24
5	7/92	W	0.69	0.36
		N	0.36	0.25

Table 3—Soil disturbance categories for the narrow-tired (N) and wide-tired (W) skidder harvest areas for each site. Totals do not equal 100 percent since coverage of natural water filled depressions, stumps, and rocks was omitted

	Site									
	1		2		3		4		5	
Disturbance class	W	N	W	N	W	N	W	N	W	N
1) Shallow depressions (<5 cm)	18	10	5	7	8	8	14	12	7	2
2) Moderate ruts (> 5 cm,< 15 cm)	5	2	2	1	2	5	3	6	0	0
3) Deep ruts (> 15 cm)	0	1	1	0	1	1	1	1	0	0
Total depressions (TD)(1 + 2 + 3)	23	13	8	8	11	14	18	19	7	2
Slash cover	17	20	20	20	14	17	15	22	15	21
Mineral soil (MS)	27	40	33	41	36	38	36	30	25	29
Total disturbed (MS + TD)	50	53	41	49	47	52	54	49	32	31
Total undisturbed	18	26	37	27	36	26	30	30	49	43

explained by the covariate, travel distance, in the continuous timing models (table 4). In the continuous timing data each cycle had a travel distance estimate whereas in cumulative production data the total cycle time and travel distance where averages for that cell. In the continuous timing models, significant ($P < 0.10$) differences for tire size were found for load building time and for travel distance and tire interactions for total cycle time and delay. In the cumulative production data a significant difference was found for only total cycle time.

With the continuous timing data the difference between wide-tired and narrow-tired skidder total cycle time increased with increasing travel distance. This was indicated by the significant interaction, tire*travel distance. The total cycle time for the wide-tired skidder increased with travel distance about 22 percent faster than the narrow-tired skidder total cycle time (figure 1). The delay model also had a significant interaction, tire*travel distance, and produced results similar to the total cycle time model with respect to the wide-tired skidder. Although the delay model is significant, the R^2 is only 0.055 compared to over 0.30 for all other element models. The interaction term was not significant for any other element, but the model predicted load building times 0.38 minutes/cycle greater for the wide-tired skidder across all travel distances.

In trials on wet terrain the cycle time advantage for wide-tires was produced during the travel elements of the cycle (Hassan and Gupta 1988, Heidersdorf and Ryans 1986, Mellgren and Heidersdorf 1984, Novak 1988). The similarity of unloaded and loaded travel times in this trial might indicate that the wide-tires were not able to produce improvements in traction or reduction in slip because of the relatively firm soil conditions present throughout the harvests. Similar or smaller increases in load building time for wide-tired skidders over narrow-tired skidders have been reported in the previous trials. The delay time

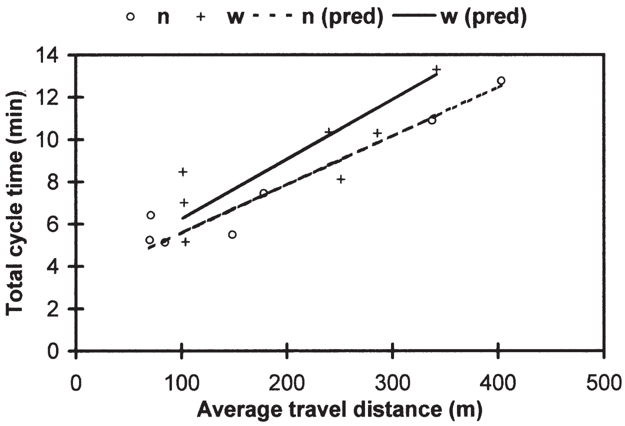

Figure 1—Average observed total cycle time from the harvest areas and predicted total cycle time from the continuous timing model for wide-tired (+) and narrow-tired (o) skidders versus average travel distance.

difference is difficult to explain since reasons for the delays in this study were not completely specified. The significant differences might result partially from confounding among site, delay time, and travel distance. Travel distance and site are somewhat confounded since a few sites contribute the majority of very long travel distances. The confounding problem is greater with the continuous timing data since most of the data for a site is taken over the more limited range of travel distances traversed in one or two days.

The high R^2 in the cumulative production models were mainly the result of limited replication within the harvest areas on each site. Only one model, total cycle time was significant at even a high P value of 0.10. The estimate of total cycle time was calculated as the total productive time divided by the number of cycles in the sample period (figure 2). These estimates were larger for a given travel

Table 4—Model R^2, N, and P value and tire, travel distance (TD), and tire*travel distance interaction P values for continuous timing and cumulative production data. If the tire*travel distance interaction was not significant ($P < 0.10$), model, tire, and travel distance interaction P values are from the reduced model without that interaction term

	Model			P values				Model			P values		
Continuous timing	P value	N	R^2	Tire	TD	Tire*TD	Cumulative production	P value	N	R^2	Tire	TD	Tire*TD
Unloaded travel	0.0001	737	0.85	0.422	0.0001	0.549	Total cycle time	0.006	15	0.96	0.093	0.003	0.737
Load building	0.0001	760	0.30	0.020	0.0001	0.631	Production	0.246	15	0.79	0.128	0.191	0.359
Loaded travel	0.0001	776	0.85	0.565	0.0001	0.400	Load size	0.448	15	0.70	0.693	0.365	0.388
Delay	0.0001	894	0.06	0.186	0.0001	0.001	Fuel use (l/t)	0.310	13	0.87	0.359	0.065	0.812
Total cycle time	0.0001	703	0.50	0.370	0.0001	0.001	Fuel use (l/pmh)	0.626	13	0.74	0.271	0.412	0.506

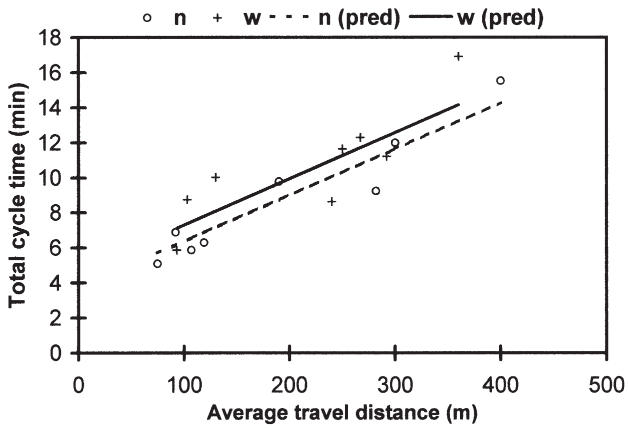

Figure 2—Predicted total cycle time from the cumulative production model and observed sample average total cycle time for wide-tired (+) and narrow-tired (o) skidders versus average travel distance.

distance than those from the continuous timing data, probably because cumulative production data contained the slower production periods that occurred at the beginning and end of the harvests. Total cycle time estimates for the wide-tired skidder were 0.92 minutes/cycle greater than those for the narrow-tired skidder across all travel distances. Significant total cycle time differences did not translate into significant differences in production (figure 3). Variation in load size could have introduced enough variability into production to mask the significant difference observed in total cycle time.

CONCLUSIONS

The relatively dry to moist soil conditions present throughout the study period prevented, for the most part, the demonstration of the advantages of wide-tires evident when used on softer, moister soil. However, wide tires did not produce any significant disadvantage in the travel elements of the cycle. The increased total cycle time for the wide-tired skidder seemed to be the result of accumulated insignificant differences in unloaded and loaded travel time and the significant differences in delay and load building times.

Operators commented that it was more difficult to maneuver around stumps and rocks in the harvest areas due simply to the extra width of the tires. They were operating the wider tires with inflation pressures near 210 kPa due to dealer recommendations to prevent the bead separation from the rim. With the high pressure in the wide tires, jolts from running over obstacles were transferred to the operator rather than at least partially absorbed by the tire. The rougher ride may have been less important in travel element than load building since a relatively obstacle-free path could be chosen for repeated use.

Production bottlenecks might have had considerable effect on both production and total cycle time. Although landing placement emphasized access to all weather roads, the inability of trucks to get to the landing and remove stored wood inhibited production for as many as several days following rains that occurred throughout the study period. Since landing space was constantly limited due to either restricted road access or the supply of trucks, operators had no incentive or ability to increase production. As an example site 5 had similar production across a wide range in travel distance caused by the limited chipper capacity and inadequate supply of chip vans (figure 3).

Differences in site disturbance were also muted somewhat by the firm soil conditions during operation. The near absence of deep rutting indicated the soil's ability to provide both flotation and thrust for both machines. However, the lack of significant difference in areal extent of rutting is a result similar to other studies on much softer and moister terrain. The majority of differences in the areal extent of disturbance and rutting among sites was probably due more to uncontrolled factors such as feller-buncher type and activity, topography, and harvest layout.

The results of this study did not help to significantly clarify issues of comparative productivity of wide-tired and narrow-tired skidders on upland soils similar to these. Unless wet season road and landing access is addressed in harvest planning of upland sites, the operations may be restricted even when the wide-tired machines can provide a production advantage over conventional machines.

Inability to provide access or the lack of production advantages on upland sites would leave the cost of wide tires to be recouped mostly through decreases in site disturbance. The shallow aspen roots which provide the sucker regeneration are severely damaged even by the shallowest ruts (Bates and others 1992, Smidt 1996). Differences in areal extent of disturbance which are controlled mostly by planning on dry and moist upland sites are more critical than disturbance severity.

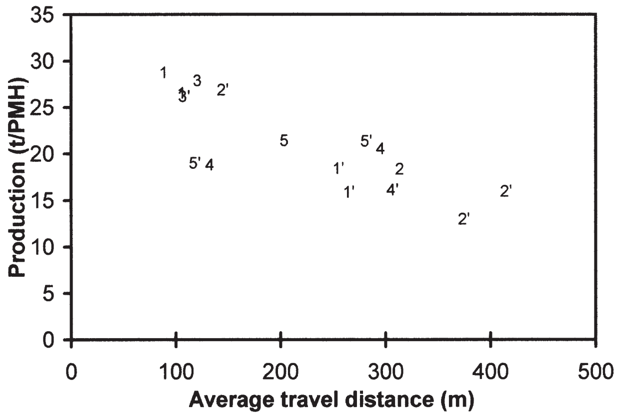

Figure 3—Production estimates versus average travel distance. Observations from narrow-tired harvest areas are indicated by the site numbers (1 - 5) alone. Observations from wide-tired harvest areas are indicated by the site number followed by an apostrophe. 1 and 1' represent narrow-tired and wide-tired skidder observations, respectively, from site 1.

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REFERENCES

Aust, W.M.; Reisinger, T.W.; Stokes, B.J.; Burger, J.A. 1991. Wide-tired skidders operating on wetland sites - preliminary results. In: Forestry and environment: engineering solutions: Proceedings of the American society of agricultural engineers conference; 1991 June 5-6; New Orleans, LA. ASAE Publ. 09-91. American Society of Agricultural Engineers: 41-53.

Bates, P.C.; Blinn, C.R.; Alm, A.A. 1993. Harvesting impacts on quaking aspen regeneration in northern Minnesota. Canadian Journal of Forest Research. 23: 2403-2412.

Groot, A. 1987. Silvicultural consequences of forest harvesting on peat lands: site damage and slash conditions. Inf. Rep. O-X-384. Environment Canada, Canadian Forestry Service, Great Lakes Forest Research Center. 20 p.

Hassan, A.E.; Gupta, R. 1988. Evaluation of wetland harvesting systems. In: Summer meeting; 1988 June 26-29; Rapid City, SD. ASAE Paper 88-5032. American Society of Agricultural Engineering. 22 p.

Heidersdorf, E.; Ryans, M.T.I. 1986. Joint FERIC/MER high-flotation tire trials, Quebec, 1984. Tech. Rep. TR-64. Forest Engineering Research Institute of Canada. 40 p.

Howes, S.W.; Hazard, J.W.; Geist, J.M. 1983. Guidelines for sampling some physical conditions of surface soils. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Range and Watershed Management. 34 p.

Meek, P. 1994. A comparison of two skidders equipped with wide and extra wide tires. Field Note, No. Skidding Forwarding-28. Forest Engineering Research Institute of Canada. 2 p.

Mellgren, P.G.; Heidersdorf, E. 1984. The use of high flotation tires for skidding in wet and/or steep terrain. Tech. Rep. TR-57. Forest Engineering Research Institute of Canada. 47 p.

Neter, J.; Wasserman, W.; Kutner, M.H. 1985. Applied linear statistical models. 2d ed. Homewood, IL: Irwin Inc. 1,127 p.

Novak, W.P. 1988. Downsizing skidders with high-flotation tires. Tech. Note TN-113. Forest Engineering Research Institute of Canada. 5 p.

SAS Institute Inc. 1988. SAS/STAT user's guide. Release 6.03 ed. Cary, NC: SAS Institute Inc. 1,028 p.

Smidt, M.F. 1996. Production, soil disturbance, and regeneration of wide-tired skidder harvests of quaking aspen stands in east-central Minnesota. St. Paul, MN: University of Minnesota. 136 p. Ph.D. dissertation.

IMPACTS OF HARVEST INTENSITY AND SOIL DISTURBANCE ON EARLY TREE GROWTH AND EARTHWORM POPULATIONS IN A MISSOURI OZARK FOREST

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Abstract—The long-term impact of increased removal of forest biomass and nutrients with increased harvest intensity on soil productivity is a general concern. In 1994, a long-term study was initiated in the Missouri Ozarks as part of the National Long-Term Soil Productivity (LTSP) study to study the effects of biomass removal and compaction on soil productivity. The study has three levels each of organic matter removal (boles only, whole tree, and whole tree plus forest floor) and soil compaction (none, moderate, and severe). This report presents 3-year preliminary results from the low and high organic matter removal and soil compaction treatments with and without weed control on survival and growth of planted northern red oak (*Quercus rubra* L.), white oak (*Quercus alba* L.), and shortleaf pine (*Pinus echinata* Mill.) seedlings. Differences in seedling survival were affected by organic matter removal and soil compaction treatments. Trees with weed control were larger in diameter, taller, and had more diameter and height growth than trees without weed control. Organic matter removal and soil compaction treatments significantly affected the height and diameter growth of trees differently. Analysis of spring and fall samplings of earthworm populations showed that soil compaction and time of sampling significantly influenced the number and biomass of earthworms.

INTRODUCTION

Forest harvesting obviously removes nutrients from the ecosystem. Conventional harvesting removes about 5 to 30 percent of the total nutrients in the aboveground stand, but the average for most forests rarely exceeds 10 percent (McColl and Grigal 1979). Intensive harvesting, which removes a greater proportion of the forest biomass than has traditionally been removed in sawtimber harvests, is contemplated as a means to provide more biomass for fiber, fuel, and chemicals. There is worldwide concern, however, that increased removal of biomass and the associated nutrients may cause a decline in forest productivity. Also, the use of heavy equipment may adversely affect physical properties of the soil causing compaction, loss of porosity, and erosion.

These concerns led to the development of a joint National Forest System/Forest Service Research study (Powers and others 1989) on long-term site productivity (LTSP). The national study has two major objectives. The first is to determine how changes in soil porosity and organic matter affect fundamental soil processes controlling forest productivity and sustainability. The second objective is to compare results from similar replicated studies among major forest types and soil groups across the United States and Canada. This report presents some early results in the Missouri LTSP study of effects of organic matter removal, soil compaction and weed control on the survival and growth of planted trees and earthworm numbers and their biomass.

METHODS AND MATERIALS

Study Site

The oak-hickory (*Quercus* L. - *Carya* Nutt.) forest type is the major timber type in the Central Hardwood Region occurring over a variety of soils, relief, and stand conditions. The Missouri LTSP study is located at the Carr Creek State Forest in Shannon County. Shannon County is located in the southeastern Missouri Ozarks. Mean annual precipitation in the area is 112 cm and mean annual temperature is 13.3°C. The study site is located on the upper northeastern-facing side slopes of two parallel ridges. Both convex and concave landforms occur on the sloping (20 to 28 percent slopes) topography. The weathering of the Ordovician and Cambrian dolomite has resulted in a deep mantle of cherty residuum (Gott 1975). Soils derived from this residuum are primarily of the Clarksville series (loamy-skeletal, mixed, mesic, Typic Paleudults). Prior to harvest, the site had a well-stocked, mature, second-growth oak-hickory forest. The site index for 50 year-old black oak (*Quercus velutina* Lam.) ranged from 74 to 80 feet (Hahn 1991).

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, **BO**), (2) All living vegetation removed (whole tree, **WT**), and (3) all living vegetation plus forest floor removed, exposing mineral soil (whole tree + forest floor, **WTFF**). Merchantable boles included trees with diameters at breast height (dbh) of 25 cm or larger. The three levels of compaction included (1) no compaction ($\mathbf{C_0}$),

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(2) moderate compaction (**C_1**), and (3) severe compaction (**C_2**). The targeted bulk density of severe compaction treatment was an increase of 30 percent 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. Each of the nine treatment combinations was replicated three times. Physical and chemical plot variables were also measured in three uncut (unharvested) plots adjacent to treatment plots in the study. For this report, two levels of organic matter removal and two levels of soil compaction, each with and without weed control, were used (Table 1).

Pre-Harvest Measurements, Site Preparation, Treatment Application, and Sampling

After determining that the key soil properties did not vary significantly across the areas, preliminary plot boundaries were established. Contiguous plots, approximately 0.4 hectare (1 acre) in size, were assigned treatments randomly in the summer of 1993. Five-meter-wide buffer strips were included around all plots. A clearcut area with all vegetation removed, whose width approximated or exceeded the height of bordering trees, separated all treatment plots from residual forest. The pre-harvest inventory of the overstory, understory, herbaceous layer and dead and downed woody material was completed in the summer of 1993 by the Missouri Forest Ecosystem Project forester and botany crew of the Missouri Department of Conservation (Ponder and Mikkelson 1995). Other collections before the harvest were biomass samples from overstory canopy trees, understory saplings, ground vegetation, and leaf litter/humus layer. In addition to 2.5 meter deep soil pits being dug and soil profiles described, soil samples were collected for nutrient and bulk density analyses (Ponder and Mikkelson 1995).

Trees were harvested from February thru May 1994. On plots designated as no compaction, all trees with a dbh of 25 centimeters or larger (merchantable trees) on BO, WT, and WTFF plots were directionally felled and removed with a skyline cable logging system. Merchantable trees on plots where the soil was to be compacted, plot borders, and the area within the study boundary were directionally felled and removed with a skidder that traveled only on designated paths within the plots and in plot borders. After the removal of merchantable trees in BO plots, the remaining trees were felled. This standing biomass included trees with dbh less than 25 centimeters (unmerchantable trees), crowns from merchantable trees, standing dead and live snags, and trees in the herbaceous layer with basal diameters greater than 2 centimeters or at least 25 centimeters tall. Trees were cut into lengths that permitted material to be hand carried, lay on or near the ground, and form a layer of uniform height over the plot. Except for crowns, which were retained on the BO plots but not on WT plots, the remaining biomass on WT plots was treated the same as biomass on BO plots. On WTFF harvested plots, all biomass remaining after harvest was removed. This included all understory vegetation plus the forest floor was raked away to the mineral soil. For BO and WT plots requiring compaction, it was necessary to remove and replace these materials (except the leaf/litter layer) after compaction was completed. Skidders and tractors were permitted on compacted plots, but not on plots that were not to be compacted. A 14-ton vibrating sheep-foot roller was used to compact soil. The roller made passes over the severe compaction treatment until there was no change in bulk density after roller passes. For bulk density measurements, soil cores (30 centimeters in length x 9.2 centimeters in diameter) were extracted from each plot using a soil-coring device (Ponder and Alley 1997). The cores were divided into 10-centimeter depth increments, oven dried at 105°C, and weighed. Bulk density for each sample was calculated according to the method of the Soil Survey Staff (1984). Soil bulk density measurements were taken after the roller made one, three, five, and eight passes over plots that received the severe compaction treatment. Changes in bulk density usually ceased after five passes.

Table 1—Treatment combinations for two levels each of organic matter, compaction, and weed control in a Missouri forest

Treatment	Organic matter removal	Soil compaction	Weed[a] control
BOC_0	BO	C_0	With
BOC_0	BO	C_0	Without
BOC_2	BO	C_2	With
BOC_2	BO	C_2	Without
$WTFFC_0$	WTFF	C_0	With
$WTFFC_0$	WTFF	C_0	Without
$WTFFC_2$	WTFF	C_2	With
$WTFFC_2$	WTFF	C_2	Without

[a] Each treatment plot split on weed control treatment.

In late spring of 1994, all plots were planted with 1-0 red oak, white oak, and shortleaf pine in a 3:3:1 ratio, (3 red oak seedling plus 3 white oak seedlings to 1 shortleaf pine seedling), using hoedads. Seedlings were planted in rows at a spacing of 2.5 meters by 2.5 meters. A 3-foot radius area around each seedling was sprayed with a glyphosate and simazine mixture to control weeds. Beginning in the 3rd growing season (1996), half of each plot was kept weed-free to permit planted trees to grow freely. The other half of the plot was allowed to develop naturally. Net primary productivity in these two plant communities will provide direct measures of productivity as influenced by study treatments.

Seedling heights and diameters at 2.5 centimeters above the ground were measured immediately after planting, and annually thereafter. Twelve 900-centimeters^2 subplots of herbaceous or ground flora samples were collected from each plot annually in the fall for dry weight and nutrient element concentration. Samples were clipped, placed in paper bags, air-dried before oven drying, weighed, ground in a Wiley mill and sent to the Ohio Research Analytical Laboratory for elemental analyses.

Earthworm Sampling

Plots were sampled for earthworms in the spring and fall of 1995, the second year after completing the installation of the study. Ten earthworm-sampling units were taken randomly at approximately the same distance apart across each plot: 5 from the upper 1/3 of the plot (top) and 5 from lower 1/3 of the plot (bottom). Each sampling unit measured 30.5 centimeters (length) x 30.5 centimeters (width) x 15 centimeters (depth). Earthworms were also collected from three uncut plots. Earthworms in top and bottom samples were combined for statistical analysis.

Earthworms were hand-sorted, counted, and placed in specimen cups containing about one-third volume of soil and stored in ice chest immediately. Earthworms were later recounted and preserved in formalin before being identified using the internal characteristic identification method of James (1990). Earthworms were dried in an oven at 60°C for 48 hours and dry weights were recorded. The oven-dried earthworms were ashed in a muffle furnace at 500°C for 4 hours and the ash weights recorded. Earthworm biomass was calculated by subtracting the ash weight from the dry weight (Parmelee and others 1990).

Statistical Analyses

The experiment was analyzed as a split-plot design with three replications, with organic matter removal and compaction treatments as the main plots and the weed control treatments as the subplots. Analysis of variance procedures were conducted with the PROC GLM procedures in SAS (SAS Institute, 1987). Survival data were analyzed using SAS (Allison 1995). Prior to analysis, the data were transformed to equalize variance using log10 transformation. Data were analyzed using analysis of variance and comparisons were made using the Least Significance Difference (LSD) test. Unless otherwise noted, all statistical tests were performed at the (α = 0.05 level of significance.

RESULTS AND DISCUSSION

Bulk Density

The severe soil compaction treatment effectively increased the bulk density over the no compaction treatment (Table 2). Bulk density generally increased with depth. The percent change in bulk density between the no compaction treatment (0 passes) and the severe compaction treatment (5 or more passes) was 22, 29, and 26 percent for the 10, 20 and 30 centimeter depth increments, respectively. Although soil compaction levels for the severe soil compaction treatment were not at the targeted level of 30 percent greater than the levels for the no soil compaction treatment, the differences were significant at the 0.05 level.

Table 2—Mean soil bulk density measurements for no compaction and severe compaction treatments. Soil compaction was done with a 14-ton vibrating sheep-foot roller

Depth	Bulk density: No compaction	Bulk density: Severe compaction[a]
Cm	*G/cm^3*	
0 - 10	1.26(±0.2)[b]	1.61(±0.09)
11 - 20	1.33(±0.23)	1.88(±0.08)
21 - 30	1.56(±0.27)	2.12(±0.37)

[a] Five or more passes.
[b] Numbers in parenthesis are standard deviation of mean based on 12 samples.

Survival

After three years, weed control had no significant effect on survival of the planted seedlings (Table 3). However, there were survival differences associated with organic matter removal and soil compaction for each species. Survival for all species was significantly lower in the BO treatments compared to the WTFF treatments. Also, survival was better for all species in the severe soil compaction treatment than in the no compaction treatment. Most of the mortality occurred during the first year (1994) after planting (Table 4). Some of the mortality was undoubtedly caused by the late planting period of May thru June.

Table 3—Survival of planted northern red oak, white oak, and shortleaf pine with two levels each of organic matter removal (removal), compaction, and weed control three years after site preparation and treatment application in the Missouri LTSP study

Treatment	Red oak	White oak	Shortleaf pine
	Percent		
Removal[a]			
BO	76.1a[b]	77.3a	64.0a
WTFF	90.5b	87.4b	68.5b
Compaction			
C_0	79.4a	79.3a	60.4a
C_2	87.0b	85.8b	70.3b
Weed control			
With	80.2a	80.3a	60.1a
Without	78.2a	73.3a	57.6a

[a]BO means boles only and WTFF, whole tree plus forest floor removal.
[b]Values in a column for a tree species and for a parameter are not significantly different (α=0.05) when followed by the same letters.

Table 4—Survival of planted northern red oak, white oak, and shortleaf pine after one two and three years following site preparation and treatment application in a harvested Missouri forest stand

Treatment[a]	With weed control			Without weed control		
	1995	1996	1997	1995[b]	1996	1997
	Percent					
Northern red oak						
BOC_0	77	72	68a[c]	69	66	62a
BOC_2	95	93	84b	89	86	77b
$WTFFC_0$	97	96	91b	97	97	93b
$WTFFC_2$	94	93	90b	94	94	92b
White oak						
BOC_0	78	73	72a	72	71	63a
BOC_2	86	86	82ab	80	80	79a
$WTFFC_0$	93	93	86b	95	95	78a
$WTFFC_2$	96	95	88b	93	93	84b
Shortleaf pine						
BOC_0	47	47	46a	50	47	46a
BOC_2	82	82	82b	90	90	90b
$WTFFC_0$	78	78	75b	63	60	50a
$WTFFC_2$	58	58	58a	71	71	70ab

[a]BO means boles only and WTFF, whole tree plus forest floor removal.
[b]All trees had weed control in 1995.
[c]Values in a column for a tree species are not significantly different (α=0.05) when followed by the same letters.

The lack of survival differences between weed control treatments is not a surprise. All plots received weed control treatments for the first two years after planting. Without weed control during the first several years after planting, seedlings planted in harvested stands using conventional procedures usually have very poor survival. Much better survival has been known to occur when large diameter seedlings are planted beneath shelterwood systems (Johnson and others 1986). It appears that part of the reason for the relative low survival of trees in BOC_0 treatment where only boles were removed and there was no soil compaction treatment may be due to the somewhat better growth of invading herbaceous vegetation. Although the weight of herbaceous vegetation did not differ significantly between treatments, they ranked in the order of BOC_0 > $WTFFC_0$ > BOC_2 > $WTFFC_2$. Also, the uncompacted plots tended to have more surviving natural trees and sprouts. Many existing small trees and shrubs were killed or severely damaged during the soil compacting process. Also, damaged stumps in the severe compaction treatment produced few sprouts. Removing the forest floor also eliminated seeds from the plots and exposed others that might have been eaten by birds and rodents.

Growth

Both diameter and total diameter growth of trees three years after planting were significantly affected by treatments (Table 5). Treatments affected species differently. Organic matter removal affected both the diameter and the total diameter growth of both red and white oak, but not shortleaf pine. Diameter growth was more than two times and nearly two times greater for red oak and white oak, respectively, in the Bo treatment compared to the WTFF treatment. Neither was diameter or total diameter growth of any of the trees tested significantly affected by compaction treatments, but weed control did. Trees with weed control were larger in diameter and had more diameter growth than trees without weed control. There were significant interactions between organic matter removal and weed control, and organic matter removal and compaction for diameter, and between organic matter removal and weed control for total diameter growth for white oak. Both diameter and total diameter growth followed the order of BO with weed control > WTFF with weed control > BO without weed control > WTFF without weed control. The diameters of white oak trees for the significant interaction between organic matter removal and compaction were BOC_2 > BOC_0 > $WTFFC_2$ > $WTFFC_0$.

Except for shortleaf pine, trees with weed control were significantly taller than trees without weed control (Table 6). Total height growth of northern red oak was also significantly better with weed control than without it. Organic matter removal did not affect total height growth of of any of the species tested, but it did significantly affect the height of white oak. White oak in the BO treatment was taller than

Table 5—Diameter and total diameter growth of planted northern red oak, white oak, and shortleaf pine with two levels each of organic matter removal (removal), compaction, and weed control three years after site preparation and treatment application in the Missouri LTSP study

Treatment	Diameter			Total diameter growth		
	Red oak	White oak	Shortleaf pine	Red oak	White oak	Shortleaf pine
	mm					
Removal[a]						
BO	19.8a[b]	11.5a	22.8a	15.2a	7.0a	20.0a
WTFF	11.4b	8.5b	20.5a	6.7b	3.8b	17.7a
Compaction						
C_0	17.8a	10.3a	20.6a	13.2a	5.9a	17.9a
C_2	11.7a	9.7a	20.6a	6.9a	5.0a	17.7a
Weed control						
With	21.2a	11.5a	25.3a	16.6a	7.1a	22.6a
Without	9.5b	8.3b	17.9b	4.8b	3.8b	15.1b

[a]BO means boles only and WTFF, whole tree plus forest floor removal.
[b]Values in a column for a tree species and for a parameter are not significantly different (α=0.05) when followed by the same letters.

Table 6—Height and total height growth of planted northern red oak, white oak, and shortleaf pine with two levels each of organic matter removal (removal), compaction, weed control three years after site preparation and treatment application in the Missouri LTSP study

Treatment	Height			Total height growth		
	Red oak	White oak	Shortleaf pine	Red oak	White oak	Shortleaf pine
	Cm					
Removal[a]						
BO	76.9a[b]	57.6a	105.5a	41.7a	41.0a	90.0a
WTFF	73.8a	51.3b	106.3a	40.9a	35.0a	92.0a
Compaction						
C_0	77.2a	54.5a	98.5a	43.3a	38.2a	83.2a
C_2	72.2b	54.1a	115.7b	37.6a	37.4a	101.3a
Weed control						
With	83.7a	60.5a	106.8a	49.5a	43.8a	92.0a
Without	66.6b	47.8b	105.0a	32.5b	31.5a	89.9a

[a]BO means boles only and WTFF, whole tree plus forest floor removal.
[b]Values in a column for a tree species and for a parameter are not significantly different (α=0.05) when followed by the same letters.

white oak in the WTFF treatment. Soil compaction affected the height of northern red oak and shortleaf pine differently. Northern red oaks in the no soil compaction treatment were taller than red oak in plots that were in the severe soil compaction treatment, but the opposite was true for shortleaf pine. There was a significant interaction between soil compaction and organic matter removal for both height and total height growth for white oak. Treatments for height followed the order BOC_0 > $WTFFC_2$ > BOC_2 > $WTFFC_0$. The order of treatments for total diameter growth followed the order BOC_2 > BOC_0 > $WTFFC_2$ > $WTFFC_0$.

Earthworms

Soil compaction significantly affected the number of earthworms in both spring and fall samples, but it affected their biomass only in the fall samples (Figures 1 and 2). The number of earthworms found in the spring was 50, 42, and 15 per meter^2, respectively, for uncut, no compaction, and severe compaction treatments. The number of

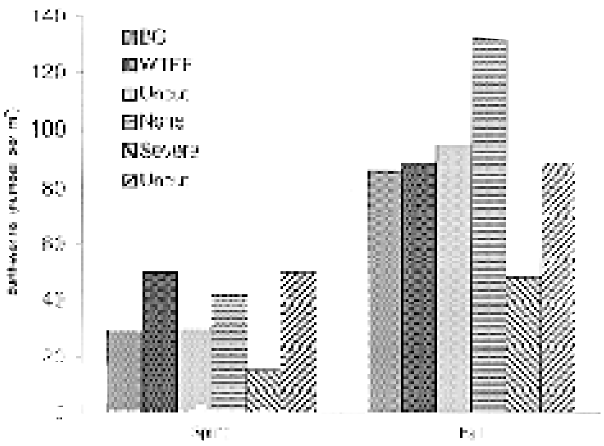

Figure 1—Response of earthworms to organic matter removal (BO, WTFF, and Uncut) and soil compaction (None, Severe, and Uncut) in a Missouri forest.

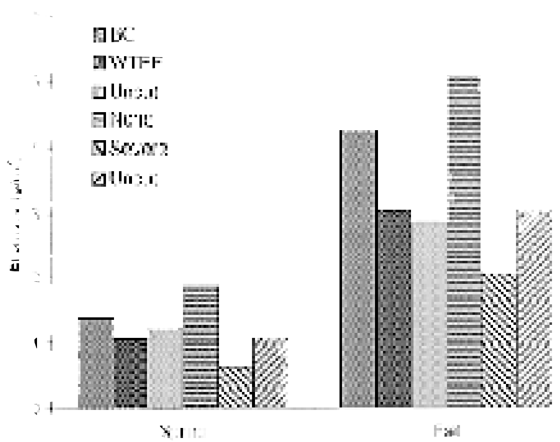

Figure 2—Earthworm biomass affected by organic matter removal (BO, WTFF, and Uncut) and soil compaction (None, Severe, and Uncut) in a Missouri forest.

earthworms was considerably higher in soils sampled in the fall than in soils sampled in the spring. They were 132, 89, and 48 per meter^2, respectively, for no compaction, uncut, and severe compaction treatments. Earthworm biomass for fall samples was 5.1, 3.0 and 2.1 grams/meter^2, respectively, for no compaction, uncut, and severe compaction plots. Neither the number of earthworms nor their biomass was significantly affected by organic matter removal treatments in spring or fall samples.

These observations suggest that soil compaction has, perhaps, been more influential than organic matter removal on tree growth and earthworm activity. Nutrient analyses of the herbaceous vegetation collected in year three of the study showed that except for P, differences between treatments were not significant. The leaf P concentration for herbaceous vegetation from the BOC_0 treatment was significantly higher than the P concentration in herbaceous vegetation from the $WTFFC_0$ and the $WTFFC_2$ treatments. Phosphorus concentration for all four treatments were in the order of BOC_0 > BOC_2 > $WTFFC_2$ > $WTFFC_0$. Herbaceous samples collected in year two showed N to be significantly different between the BOC_0 treatment and other treatments (Ponder 1997). But the difference was not present in year three. Apparently, nutrient differences caused by biomass removal treatments (Ponder and Mikkelson 1995), which may eventually affect soil nutrient supply, are not currently different enough to be detected consistently in herbaceous samples.

The increase in soil bulk density associated with soil compaction can, depending on the soil moisture content affect the soil strength. Soil strength affects the ability of roots, and perhaps earthworms, to penetrate or move through the soil. Soil compaction also reduces soil porosity. Decreased soil porosity can reduce soil water, cause poor aeration, and affect the distribution and growth of earthworms (Edwards and Bohlen 1966). We do not report soil strength or soil porosity measurements for this study. However, researchers investigating other LTSP sites have reported that after five years, soil strength in compacted plots was significantly higher and soil porosity was significantly lower than for soil in no compaction plots, (Powers and Fiddler 1997; Stone and Elioff 1998).

Because of the cherty, well-drained, soils in the study area, we had expected earthworm numbers to be higher in the spring when soil moisture was believed to be greater. The larger number of earthworms in the fall compared to the spring may be due to, in addition to the effects of soil compaction, the results of a number of soil environmental factors acting alone or interacting with each other. These may include soil temperature, soil moisture, organic matter, and perhaps, the species reproduction patterns.

SUMMARY

Early tree survival was significantly higher in $WTFFC_2$ treatments. Both organic matter removal and soil compaction treatments affected diameter, diameter growth, height, and height growth of tree species differently. Tree growth was also significantly better with weed control than without weed control.

These results are preliminary. The earthworm study along with some more recently initiated monitoring of soil and leaf nutrient analyses, soil moisture, and soil temperature is continuing. Also of interest, will be the long-term effects of weed control on tree biomass production, soil moisture, and soil temperature, and interactions of weed control with organic matter removal and compaction to influence vegetation growth (Powers and Fiddler 1997). Analyses of these data should help to better define the below-ground effects of organic matter removal and soil compaction on tree growth and how earthworms respond to these disturbances. Survival and growth differences among treatments should become more pronounced as the stand continues to develop.

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REFERENCES

Allison, P.D. 1995. Survival analysis using the SAS system: a practical guide. Cary, NC: SAS Institute, Inc. 292 p.

Edwards, C.A.; Bohlen, P.J. 1966. Biology and ecology of earthworms. New York: Chapman and Hall. 426 p.

Gott, J. 1975. Soil survey of the Mark Twain National Forest area. Missouri (Parts of Carter, Oregon, Reply, and Shannon Counties). Rolla, MO: U.S. Department of Agriculture, Forest Service. 56 p.

Hahn, J.T. 1991. Timber resource of Missouri. Res. Bull. NC-119. U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 123 p.

James, S.W. 1990. *Oligochaeta: Megascolecidae* and other earthworms from southern and midwestern North America. In: Dindal, ed. Soil Biology Guide. New York: Wiley-Interscience Publication: 379-386.

Johnson, P.S.; Dale, C.D.; Davidson, K.R.; Law, J.R. 1986. Planting northern red oak in the Missouri Ozarks: a prescription. Northern Journal of Applied Forestry. 3: 66-68.

McColl, J.G.; Grigal, D.F. 1979. Nutrient losses in leaching and erosion by intensive forest harvesting. In: Proceedings, impact of intensive harvesting on forest nutrient cycling; 1979 August 13-16; State University of New York, Syracuse, NY: 231-248.

Parmelee, R.W.; Beare, M.H.; Cheng, W. [and others]. 1990. Earthworms and enchytraeids in conventional and no-tillage agroecosystems: a biocide approach to assess their role in organic matter breakdown. Biology and Fertility of Soils. 10: 1-10.

Ponder, F., Jr. 1997. Initiating long-term soil productivity research in Missouri. In: Brookshire, B.L.; Shifley, S.R., eds. Proceedings, Missouri Ozark Forest ecosystem project symposium: an experimental approach to landscape research; 1997 June 3-5; St. Louis, MO. Gen. Tech. Rep. NC-198. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station: 370-378.

Ponder, F., Jr.; Alley, D.E. 1997. Soil sampler for rocky soils. Res. Note NC-371. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 5 p.

Ponder, F., Jr.; Mikkelson, N.K. 1995. Characteristics of a long-term forest soil productivity research site in Missouri. In: Gottschalt, K.W.; Fosbroke, S.L.C., eds. Proceedings, 10th central hardwood forest conference; 1995 March 5-8; Morgantown, WV. Gen. Tech. Rep. NE-197. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 272-281.

Powers, R.F.; Alban, D.H.; Ruark, G.A. [and others]. 1989. Study plan for evaluating timber management impacts on long-term site productivity: a research and National Forest System cooperative study. Jefferson City, MO. On file with: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 33 p.

Powers, R.F.; Fiddler, G.O. 1997. The North American long-term soil productivity study: progress through the first 5 years. In: Proceedings, 18th annual forest vegetation management conference; 1997 January 14-17; Sacramento, CA. Redding, CA: 88-102.

SAS Institute. 1987. SAS/STAT guide for personal computers: version 6 ed. Cary, NC: SAS Institute.

Soil Survey Staff. 1984. Procedures for collection of soil samples and methods for analysis for soil survey. Soil Invest. Rep. 1. Washington, DC: U.S. Printing Office.

Stone, D.M.; Elioff, J.D. 1998. Soil properties and aspen development five years after compaction and forest floor removal. Can. J. Soil Sci. 78: 51-58.

CONTRASTING TIMBER HARVESTING OPERATIONS ILLUSTRATE THE VALUE OF BMPS

James N. Kochenderfer and James W. Hornbeck[1]

Abstract—Our paper compares water yield, water quality, and sedimentation from a harvesting operation conducted in 1957 without BMPs to a harvesting operation in 1986 utilizing BMPs. The comparison illustrates the values of BMPs for protecting soils and streams and provides insight into causes of water quality degradation. Both logging operations were conducted on or near the Fernow Experimental Forest located in north central West Virginia. The 1957 logging operation was conducted on a 74-acre watershed using a crawler tractor equipped with an arch. Roads were unplanned "logger's choice" with no limits on grade or location. West Virginia's BMPs were used on a second 96-acre watershed logged with wheeled skidders in 1986. Results indicated that careless logging greatly accelerated sediment yields but development of erosion pavements and natural vegetation quickly reduced them, even with no post-logging care. Although the percentage of area occupied with dozed roads was actually greater (10.6 versus 3.6 percent) on the area logged in 1986 with wheeled skidders, water quality impacts were much less. Average stream turbidity during logging in the absence of BMPs was 490 ppm versus <10 when using BMPs. Sediment loss during active logging was estimated at 2,880 lb/ac on the carelessly logged watershed compared to 110 lb/ac when BMPs were used.

INTRODUCTION

Bare soil exposed during timber harvesting can be a major source of water degradation, but guidelines developed over the years reduce adverse impacts on soil and water resources (Weitzman 1952; Haussman 1960; Kochenderfer 1970; and Pierce and others 1992). Passage of the Federal Water Pollution Control Act Amendments of 1972 required use of BMPs to control nonpoint-source water pollution from forestry activities. Despite application of past research and BMPs, and the goals of present-day forest managers to sustain and protect site productivity, critics of current timber harvesting operations sometimes equate them with less careful logging operations of the past. Thus it is important to demonstrate the improvements of current logging and BMPs in terms of protecting soil and water resources. This paper does so by comparing magnitude and longevity of impacts from a harvesting operation conducted in 1957 without BMPs to a 1986 harvest utilizing them.

STUDY AREAS

Three gaged watersheds located in the unglaciated Allegheny Plateau region of north-central West Virginia were used in this study. Two of them, watershed 1 (WS1) and watershed 4 (WS4), are located on the Fernow Experimental Forest while the Haddix watershed is 4.0 miles away. WS1 and Haddix were logged while WS4 has been retained as a reference watershed to evaluate treatment effects. Precipitation is distributed evenly across the year on all three watersheds; the annual average is 58 inches on the Fernow and 54 inches on Haddix. Annual runoff from WS4 during the study periods averaged 26 inches, 6 inches during the growing season and 20 inches during the dormant season. Other pertinent characteristics are shown in Table 1.

WS1 supports a 40-year-old stand of mesic hardwoods that originated after cutting in 1957-58, plus some scattered cull trees dating to earlier cutting in 1905-10. Dominant tree species are sugar maple (*Acer saccharum* Marsh.), northern red oak (*Quercus rubra* L.), yellow-poplar (*Liriodendron tulipifera* L.), and basswood (*Tilia americana* L.). Average stand basal area in trees 1-inch and larger averages 128 ft^2/ac. Streambanks are well vegetated but considerable bare soil is still exposed at former skidroad crossing sites along the main stream. The predominant soil series on both WS1 and WS4 is Calvin channery silt loam (loamy-skeletal, mixed, mesic typic Dystrochrepts) with moderate erosion hazard (Losche and Beverage 1967). Soils are underlain with fractured sandstone and shale of the Hampshire formation.

Table 1—Characteristics of study watersheds

	Haddix	WS1	WS4
Area, acres	96	74	96
Aspect	S	E	SE
Average slope, %	40	40	25
Average stream gradient, %	11	16	13
Stream channel area, acres	.47	.26	.30
Sediment source area, acres[a]	10.8	3.0	1.1
Tree basal area, ft^2/ac[b]	109	109	154

S = South, E = East, SE = Southeast
[a] Defined as stream channel area + road area measured upon completion of logging.
[b] Basal area prior to harvest.

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The Haddix watershed supports a 12-year-old, mixed hardwood stand that originated after a 1986 harvest, along with older residuals left from earlier harvests. Approximately one-third of it contains a dense understory of rhododendron (*Rhododendron maximum* L.). After the harvest of 1986, it supported a mixed stand of oak species (*Quercus* spp.), hickory (*Carya* spp.), red maple (*Acer rubra* L.), and yellow-poplar, vegetation reflecting a somewhat xeric site. Residual basal area after the 1986 cut averaged 61 ft^2/ac, about half of the 109 ft^2/ac before harvest. The watershed is underlain by interbedded shale, siltstone, and sandstone of the Chemung geologic formation. The predominant soil series is Berks channery silt loam (loamy-skeletal, mixed, mesic, Typic Dystrochrepts), with moderate erosion hazard (Losche and Beverage 1967). Streambanks are generally well vegetated. The channel is armored with sandstone cobbles and gravel. However, as with many stream channels in the region, there is considerable bare soil along streambanks.

WS4 provided experimental control for water quality and streamflow. It has remained relatively undisturbed since about 1905 when much of the original timber was cut. Dominant tree species are yellow-poplar, sugar maple, and northern red oak, indicating a more mesic site than Haddix. In 1994, average basal area was 154 ft^2/ac for trees 1-inch dbh and larger. The stream channel is well armored with sandstone gravel and cobbles, and streambanks are well vegetated.

TREATMENTS

WS1 was commercially clearcut between May 1957 and September 1958. All merchantable trees >5.0 inches dbh were cut. Cull trees were left standing. An average volume of 8,984 bd ft/ac was harvested. Average basal area in trees >1-inch was reduced by 77 percent to 26 ft^2/ac. Logging was done with a tractor and arch, with no measures taken to protect soil and water resources. Skidroads were constructed on a logger's choice basis with no restrictions on road grade or location. For example, no culverts or bridges were used at stream crossing sites nor were normal post-logging practices such as waterbarring or seeding used. Many of the skidroads were in or immediately adjacent to stream channels (Fig. 1). There were no truck roads or landings located in WS1. Data for road area and location are in Table 2.

The Haddix watershed was cut to a 14-inch stump diameter between May 1986 and February 1987, removing an average volume of 5,344 bd ft/ac. Average basal area was reduced 44 percent in trees 1-inch and larger. A

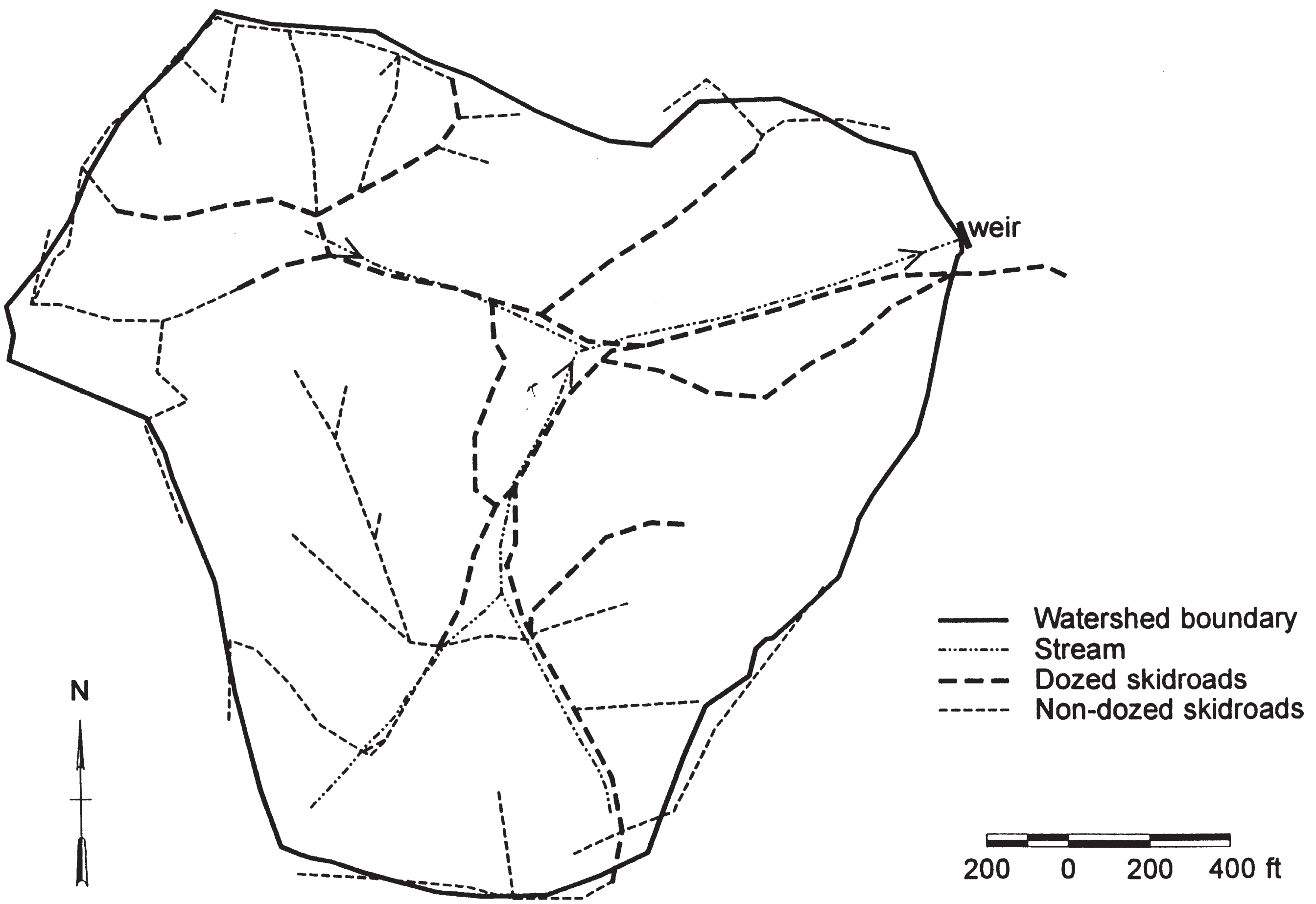

Figure 1—The unplanned road system in WS1 was used to remove 668 Mbf of timber from the 74-acre watershed. No measures were taken to protect soil and water resources.

Table 2—Roads in harvested watersheds

	WS1	Haddix
Dozed skidroads		
Length in miles	1.3	3.0
Area, acres	2.7	6.0
Percent of watershed area	3.6	6.2
By grade class, %		
0-10	22	74
11-20	32	24
21+	46	2
Within distance of stream, %		
<25 ft	40	1
<50 ft	61	3
<100 ft	66	3
Non-dozed skidroads		
Length in miles	2.2	0
By grade class, %		
0-10	31	0
11-20	35	0
21+	35	0
Truck roads		
Length in miles	0	0.9
Area, acres		4.3
Percent of watershed area		4.5
By grade class, %		
0-10	NA	100
11-20	NA	0
21+	NA	0
Within distance to stream, %		
<25 ft	NA	4
<50 ft	NA	5
<100 ft	NA	5

wheeled skidder was used to skid tree-length logs to landings. A D-4 bulldozer was used to construct skidroads as needed during harvesting.

BMPs recommended for use from 1979 to 1989 in West Virginia (West Va. Dep. Natur. Resour. 1979, 1982) were followed for this harvesting operation. The entire road system on Haddix was planned and laid out on the ground before logging to minimize roaded area and environmental impacts (Fig. 2).

A minimum-standard truck road (Kochenderfer and others 1984) was built with a John Deere 850 bulldozer. Broad-based dips (USDA For. Serv. 1940, Hewlett and Douglas 1968) spaced at about 150-foot intervals controlled overland flow. Natural grade breaks reduced the number of constructed dips needed. Except at stream crossings, roads were kept at least 100 ft slope distance from streams (Fig. 2). Metal culverts were installed at four locations where the truck road and skidroads crossed streams. A combination of metal culverts and ditches also was used on the truck road to drain three seeps. Culverts were left in place following the harvesting operation. All four landings were located on dry sites, at least 150 feet slope distance from the nearest stream.

Some rutting was tolerated on the ungraveled truck road, but hauling during extremely wet conditions was prohibited; it would have resulted in deep ruts and damage to the roadbed and water control features. Critical areas such as the two truck-road stream crossings were seeded with grass and slash was scattered on the roadfills immediately following road construction. Skidroads were smoothed and waterbarred at recommended spacing as logging progressed. Skidroads were limed, fertilized, and seeded with Kentucky 31 fescue using a cyclone seeder. In June 1987, the truck road and landings were disced, limed, fertilized, and seeded with a mixture of oats, clover, and Kentucky 31 fescue. Ground lime was applied at the rate of 3.0 ton/ac and 10-10-10 fertilizer at the rate of 500 lb/ac.

DATA COLLECTION AND ANALYSIS

Data collection began on WS1 and WS4 in May 1951 and in May 1982 on Haddix. Precipitation on each watershed was sampled by a network of recording and standard 8-inch gages. Streamflow was measured with 120° V-notch weirs on WS1 and WS4 and with a 3-foot H-type flume on Haddix. Each gaging site was equipped with an FW-1 water-level recorder. Water quality samples were collected by grab sampling above the gaging sites.

Harvesting effects on annual water yield and instantaneous peak flows were determined by using the paired watershed approach described by Hornbeck and others (1997). Since the purpose of the peakflow analysis was to compare the response of the watersheds to similar precipitation inputs, only storms for which the difference in precipitation did not exceed 0.3 inch were used to evaluate changes in peakflow. Linear regression was used to develop calibration relationships between water yield and storm peaks from WS4 and those to be harvested (WS1 and Haddix). After harvest, streamflow values from WS4 were inserted in the calibration equations to estimate what streamflow for the harvested watersheds would have been had they not been harvested. Differences between measured streamflow from the harvested watersheds and estimates of flow had they not been harvested were considered statistically significant and ascribed to forest harvest when the differences exceeded the 95 percent confidence intervals placed about the entire calibration regression.

Estimates of sediment export from WS1 and WS4 during the early 1957-58 logging period were based on turbidity and discharge measurements. Turbidities between 5-25 were determined by reference to standard suspensions in Nessler tubes and are termed Nessler turbidimeter units (NTU). Turbidities above 25 were measured with a Jackson turbidimeter and termed Jackson turbidimeter units (JTU), or filtered to determine suspended solids as parts per million (ppm).

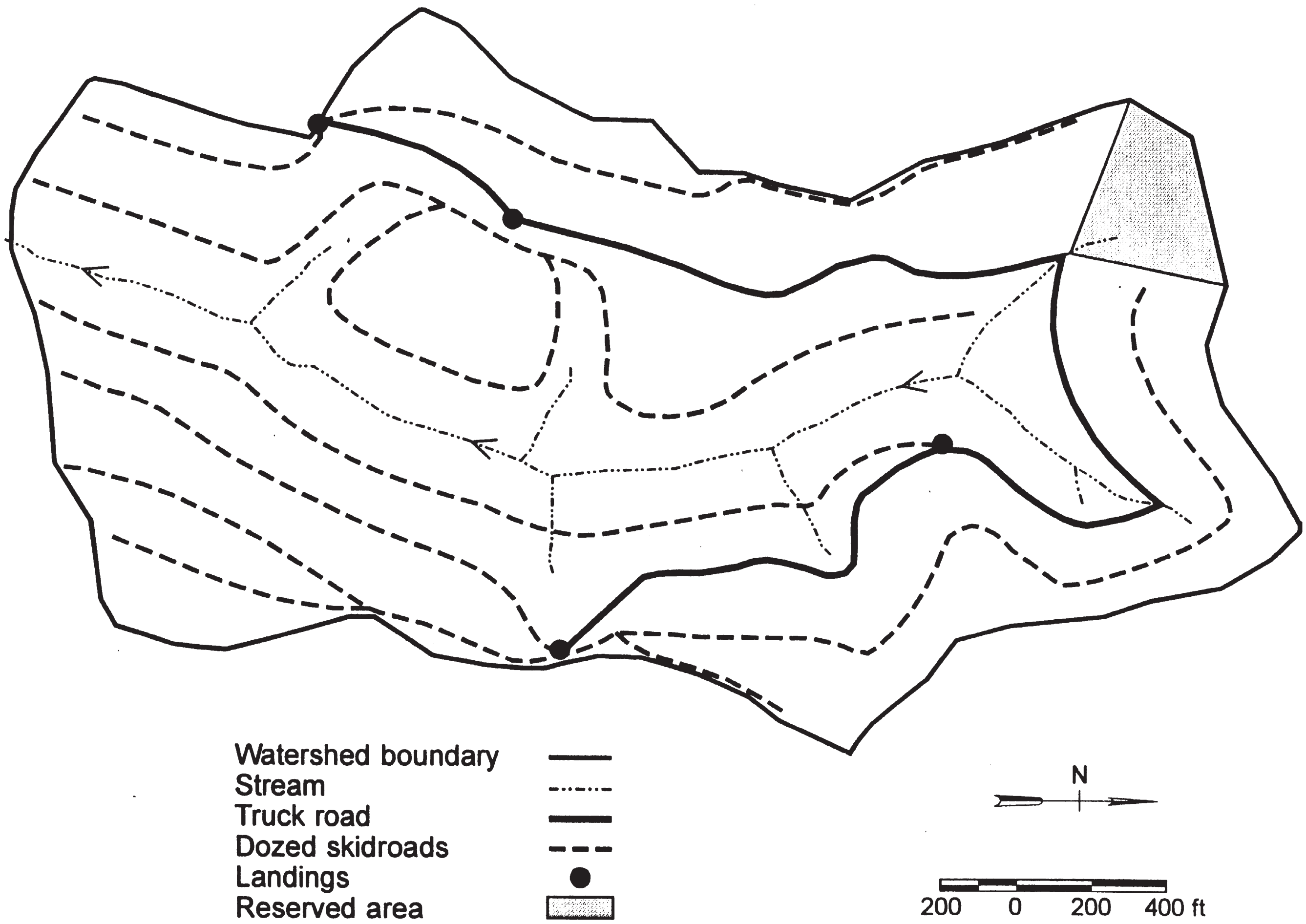

Figure 2—The road system in the Haddix watershed conformed to West Virginia's Best Management Practice Standards. It was used to remove 513 Mbf of timber from this 96-acre watershed.

During the 1982-1993 monitoring period each gaging site was equipped with a Coshocton wheel sediment sampler, which diverts 0.5 percent of the total flow into a storage tank. Two samples of tank contents were taken weekly, during base flow and before the tanks overflowed during storms. Tank contents were agitated vigorously while two 800-ml samples were drawn from spigots at the bottom of the storage tanks. Sediment produced during large storms was quantified by grab sampling and by automatic samplers. For each sample, turbidity was measured in NTU, then filtered to determine suspended sediment concentration in ppm.

Thermometers placed above the gaging stations in each watershed recorded weekly maximum and minimum stream temperatures. During the calibration period on WS1, stream temperatures were measured while collecting water quality grab samples. Beginning in May 1958, maximum-minimum thermometers were placed in the main stream above the gaging station. Water temperature records for 1958 are missing so temperatures observed during the 1959-60 period were compared with mean maximum temperatures observed during the 1961-73 period.

Grab samples for other criteria of water quality were collected weekly above the gaging stations on all three watersheds. In the beginning on WS1 and WS4, samples were analyzed for pH, alkalinity, and electrical conductivity. During the later period when Haddix was being logged, samples were analyzed for major ion concentrations at the Forest Service's laboratory. Analytical procedures and instrumentation information are given in Edwards and Wood (1993).

RESULTS AND DISCUSSION

Roads and Their Impacts

Roads and stream channels are major sources for stream sediment on the study watersheds. Road data for them are shown in Table 2. Road area was determined by length and width measurements from the top of road cuts to the toe of road fills after logging was completed. Haddix contained almost 4 times as much area in bulldozed roads as did WS1. However, when non-bulldozed roads are included, total road length in WS1 (3.5 miles) and Haddix (3.9 miles) are similar (Table 2). The differences in bulldozed road density are attributed to logging methods. The tractor and arch used on WS1 could negotiate fairly steep terrain without roads while the wheeled skidder used on Haddix was largely restricted to roads. Stream channel areas were similar among watersheds. They comprise a much smaller proportion of area than do roads in the logged watershed (Table 1), but are major sources of sediment in streams draining forested watersheds (Lull and Reinhart 1963, Patric 1976).

Area occupied by roads on the watersheds is decreasing as they revert to forest. Kochenderfer and others (1997) found that 6 years after logging, woody vegetation was dominant on half the original truck road area cleared in 1987. They concluded that reforestation should eventually reduce the original road prism area to less than half its original acreage. Several 12- to 16-inch yellow-poplar trees grow in the middle and on fill portions of skidroads in WS1.

Road grades are important from the standpoint of providing efficient access as well as soil and water protection. It is far more difficult to control erosion on steep roads, and they have less residual value for other uses once logging is completed. Roads of gentle grades, that are properly located and maintained, protect soil and water resources while providing effective access for many forest activities. Almost half of the dozed skidroads in WS1 had grades greater than 21 percent as opposed to only 2 percent of the roads on Haddix where BMPs were used (Table 2). Grades on 74 percent of the dozed roads on Haddix were in the 0-10 percent class while only 22 percent of the dozed skidroads on WS1 were in that class. All of the truck roads on Haddix fell into the 0-10 percent class.

The proximity of roads to streams is probably the single most important attribute that determines whether streams will be adversely impacted by timber harvesting operations. The importance of providing minimally disturbed protective zones between disturbed areas and streams has been recognized for a long time (Trimble and Sartz 1957, Lull and Reinhart 1963). Streamside protective zones are normally incorporated into state BMP guidelines. For example, West Virginia BMP guidelines recommend 100 feet protective zones on each side of perennial and intermittent streams and 25 feet for ephemeral streams (West Va. Dept. Natur. Resour. 1997).

There are striking differences between the unplanned road system on WS1 and the planned system on Haddix (Figs. 1 and 2). On WS1 many of the main skidroads were in or very close to stream channels. Sediment generated on roads was often carried by overland flow directly into streams. During storms, streamflow diverted into roads caused further excessive erosion (Lull and Reinhart 1963). On Haddix, the landings and contour road systems were at least 100 feet from streams except at crossing sites (Kochenderfer and others 1997). Machinery was not permitted off the road in streamside areas.

The distribution of bulldozed roads in relation to streams on WS1 and Haddix is quantified in Table 2. Forty percent of the dozed skidroads on WS1 were within 25 feet of streams (Hornbeck and Reinhart 1964) while only 5 percent of roads on Haddix watershed were that close. Sixty-one percent of the roads on WS1 were within 50 feet of streams as opposed to 8 percent on Haddix.

It is important to note that roads often generate more runoff than precipitation alone would indicate. Lull and Reinhart (1963) attributed this excess road runoff in WS1 to intercepted subsurface flow at road cuts. Kochenderfer and Helvey (1987), measuring soil losses from graveled and ungraveled road sections in central West Virginia, found that the percentage of annual precipitation measured as runoff ranged from 41.5 percent to 139 percent. Annual runoff exceeded precipitation on road segments with periodically active seeps.

Water Yield and Peak Flows

In addition to impacts of roads on overland flow, harvesting the forest reduces evapotranspiration and thereby increases water yield. The more intensive harvest on WS1 increased annual water yield 5.2 inches during the second year after harvest (Table 3). Increases disappeared quickly with regrowth and there were no statistically significant changes beyond the 6th year after harvest. Increased water yield was indicated after harvest on Haddix but was not statistically significant (Table 3).

Much of the increase in water yield occurs as augmentation to low flows during the growing season. Such increases result from increased soil water moving laterally through soil and bedrock to streams, thus having little impact on erosion. However, some peak flows also increased (Table 4). Some of the peak flow on WS1 also may have been caused by overland flow from poorly located skidroads (Reinhart and others 1963). Compared with Haddix, the lower residual basal area left after harvesting WS1 resulted in less transpiration and smaller soil water deficits, and thus greater overall increases in storm peaks (Table 4). The increases in peak flows are of interest in that they can cause additional erosion and sedimentation by scouring stream channels. However, the relatively small magnitude of the increases, plus the fairly rapid disappearance of increases with regrowth of the new forest (Table 4), suggests that harvests have minimal impacts on downstream flooding.

Sediment

Annual sediment yields and turbidity (Hornbeck and Reinhart 1964, Kochenderfer and others 1997) for WS1 and Haddix are shown in Table 5. During the logging operation sediment yield was 26 times greater on WS1 than on Haddix. These large differences reflect the characteristics of road systems discussed earlier. Although there was almost four times as much severely disturbed area on Haddix during logging, sediment yields were far less than those on WS1. The first year after logging they decreased to about 4 times more on WS1 than on Haddix and by the second year both returned to preharvest levels. Sediment yields were greatest during logging, when roads were repeatedly disturbed, then decreased rapidly after logging was completed. The rapid decrease in sediment yields on WS1 was attributed to vigorous regrowth and development of an erosion pavement on skidroads (Reinhart and others 1963). These soils contain about 50

Table 3—Impacts of harvesting on annual water yield

Year after harvest	WS1 Actual flow	WS1 Estimated streamflow if unharvested[a]	WS1 Change due to harvest[b]		Haddix Actual flow	Haddix Estimated streamflow if unharvested[a]	Haddix Change due to harvest[b]	
	Inches	*Inches*	*Inches*	*%*	*Inches*	*Inches*	*Inches*	*%*
1	21.3	19.0	2.3[c]	12	28.0	24.2	3.8	16
2	31.6	26.4	5.2[c]	20	20.5	18.1	2.4	13
3	25.1	21.4	3.7[c]	17	28.9	26.7	2.2	8
4	27.5	24.0	3.5[c]	15	37.2	33.3	3.9	12
5	23.4	22.9	0.5[c]	<1	33.9	32.6	1.3	4
6	27.5	25.2	2.3[c]	9	20.2	19.0	1.2	6
7	23.5	23.6	-0.1	<1				
8	21.7	21.1	0.6	3				

[a] Determined from calibration regression.
[b] Determined by subtracting estimated streamflow from actual streamflow.
[c] Change exceeded 95 percent confidence interval about the calibration regression.

Table 4—Impacts of harvesting on instantaneous peak flows >3 $ft^3/sec/mi^2$ (c.s.m.)

Year after harvest	Growing season: No. of peaks (*n*)	Range in peak flows (c.s.m.)[a]	Statistically significant increases (*n*)	Average change	Dormant season: No. of peaks (*n*)	Range in peak flows (c.s.m.)[a]	Statistically significant increases (*n*)	Average change
				%[b]				*%*[b]
WS1								
1	3	20-28	2	69	12	4-77	6	18
2	7	4-14	1	27	18	3-23	8	18
3	0	–	–	–	2	4-21	1	45
4	1	25	1	46	10	4-52	4	26
5	5	4-31	1	9	7	4-70	4	18
6	0	–	–	–	8	5-30	0	–
Haddix								
1	0	–	–	–	7	3-25	1	42
2	4	11-78	2	76	8	10-46	0	–
3	2	14-63	0	–	6	7-48	2	35
4	2	18-27	0	–	9	4-33	0	–
5	1	16	0	–	11	6-31	0	–
6	0	–	–	–	8	6-26	1	36

[a] Estimated peak flows if watersheds not harvested.
[b] Average change for measured peak flows that were statistically significant.

Table 5—Annual suspended sediment yields and mean turbidity from WS1 and Haddix watersheds

	Sediment yields		Turbidity	
	WS1	Haddix	WS1	Haddix
	------ *Lb/ac* ------		*Ppm*	*NTU*
During logging operation	2,880	110	490	8.0
First year after logging	288	69	38	6.0
Second year after logging[a]	7	52	1	5.0

[a] Modern sampling techniques would probably have produced values for WS1 more comparable to Haddix.

percent stone fragments by volume and quickly developed a protective stone cover (Lull and Reinhart 1963).

While sediment yields temporarily doubled on Haddix, they remained within the range of 100 to 200 lb/ac/yr background levels expected from carefully managed forest in the eastern United States (Patric 1976). Sediment exports during a single large flood event on Haddix in 1985 (before harvest) were 2.8 times higher than annual sediment exports during logging. Others have observed that sediment exports are highly variable and related to the occurrence of individual large storms (Edwards and Owens 1991, Martin and Hornbeck 1994).

Careless logging on WS1 resulted in highly turbid water, averaging 490 ppm on WS1 during logging as compared to 8.0 NTU on Haddix (Table 5). The maximum turbidity observed during the logging on WS1 was 56,000 ppm (Reinhart and others 1963) while the maximum turbidity observed on Haddix was less than 100 NTU. Turbidity decreased rapidly on WS1, averaging 38 ppm during the first year after logging.

Water Temperature

Mean maximum growing season temperatures for selected periods are shown for all three watersheds in Figure 3. Highest temperatures usually coincided with low streamflow and high air temperatures during the July-September period. Water temperatures of 75°F are detrimental to brook trout (Embody 1921, Needham 1938). Reinhart and others (1963) concluded that the commercial clearcut on WS1 raised growing season maximum temperatures an average of 8°F during 1958-59, and reduced dormant season minima by 3.5°F. Heavy accumulations of slash covering the main stream channel might have moderated temperatures to some extent because Eschner and Larmoyeux (1963) reported that the

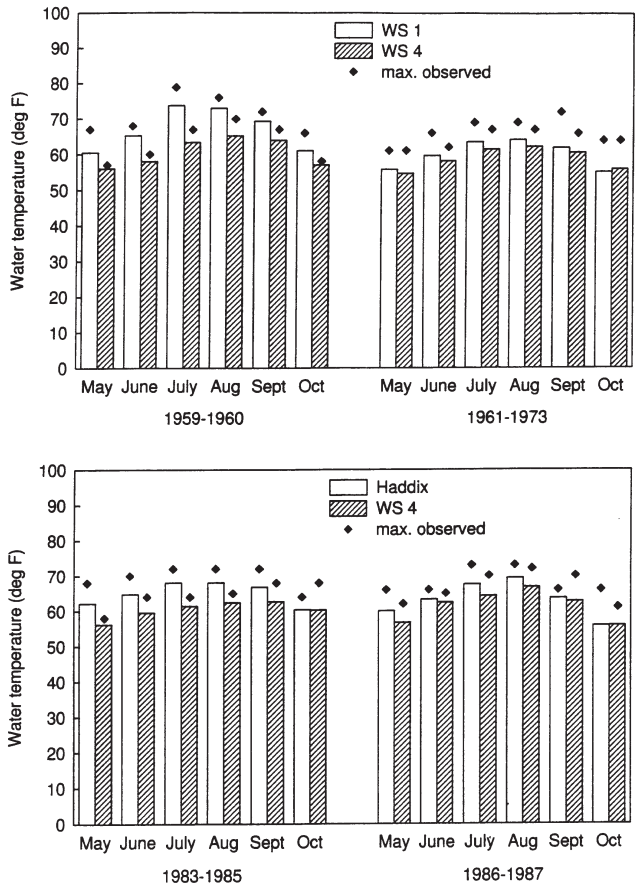

Figure 3—Mean maximum growing-season stream temperatures for WS1 (top) and Haddix (bottom) are compared with the control Fernow watershed (WS4).

highest temperature observed on WS1 (79°F) occurred in July 1959. Temperatures of 75°F and higher were observed several times on WS1 during 1959 but the highest observed in 1960 was 73°F. The highest temperature observed on WS4 during 1959 was 70°F and the maximum in 1960 was 65°F.

Temperatures during pretreatment and logging periods on Haddix are very similar (Fig. 3). Kochenderfer and others (1997) concluded that diameter limit cutting on Haddix did not affect stream temperature because of shading by residual trees and understory vegetation. Although temperatures have consistently remained higher on the more xeric Haddix watershed, they have remained below 75°F.

Electrical Conductivity

Electrical conductivity, an index of total dissolved solids, decreased on WS1 following commercial clearcutting (fig. 4) and has consistently been higher on WS1 than on WS4. These differences probably reflect differences in geology and resultant weathering contributions to streams. There was a small increase in electrical conductivity in the Haddix stream after logging (fig. 4). An earlier study by Aubertin and Patric (1974) showed minimal streamwater ion increase after clearcutting Fernow watershed 3. Thus there

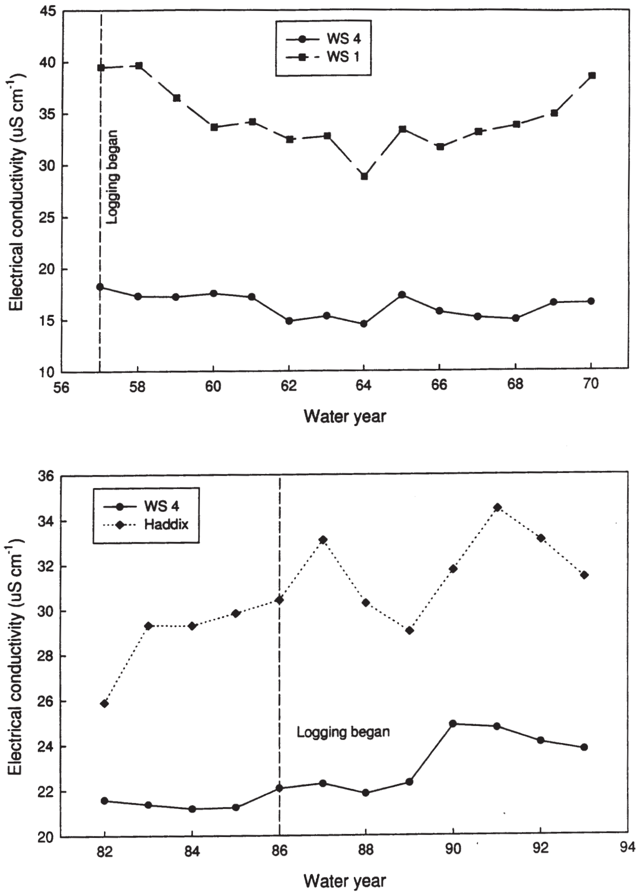

Figure 4—Time trends of electrical conductivity in streamflow from WS1 (top) and Haddix (bottom) compared with that from WS4.

is little reason to expect electrical conductivity in streams draining WS1 and Haddix to show much change after harvest.

SUMMARY

Our comparison of watersheds logged with and without BMPs clearly demonstrates their value. Logging without them on WS1 involved roads with unusually steep grades located close to or in streams, and no attempts were made to control water and revegetate the roads after logging. The result was significant erosion and sedimentation with increased stream temperatures. In contrast, careful adherence to West Virginia's BMPs when logging Haddix, resulted in only minor changes in sediment and water temperature. The changes were within background levels, clearly illustrating that harvesting operations utilizing BMPs will protect water quality.

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REFERENCES

Aubertin, G.M.; Patric, J.H. 1974. Water quality after clearcutting a small watershed in West Virginia. Journal of Environmental Quality. 3: 243-249.

Edwards, P.J.; Wood, F. 1993. Field and laboratory quality assurance/quality control protocols and accomplishments for the Fernow Experimental Forest watershed acidification study. Gen. Tech. Rep. NE-177. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 15 p.

Edwards, W.M.; Owens, L.B. 1991. Large storm effects on total soil erosion. Journal of Soil and Water Conservation. 46: 75-78.

Embody, G.C. 1921. Concerning high water temperatures and trout. Transactions of the American Fisheries Society. 51: 58-64.

Eschner, A.R.; Lamoyeux, J. 1963. Logging and trout: four experimental forest practices and their effect on water quality. Progressive Fish Culturist. 25(2): 59-67.

Haussman, R.F. 1960. Permanent logging roads for better woodlot management. U.S. Department of Agriculture, Forest Service, Eastern Region Division of State and Private Forestry. 38 p.

Hewlett, J.D.; Douglass, J.E. 1968. Blending forest uses. Res. Pap. SE-137. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 15 p.

Hornbeck, J.W.; Martin, C.W.; Eagar, C. 1997. Summary of water yield experiments at Hubbard Brook Experimental Forest, New Hampshire. Canadian Journal of Forest Research. 27: 2043-2052.

Hornbeck, J.W.; Reinhart, K.G. 1964. Water quality and soil erosion as affected by logging in steep terrain. Journal of Soil and Water Conservation. 19(1): 23-27.

Kochenderfer, J.N. 1970. Erosion control on logging roads in the Appalachians. Res. Pap. NE-158. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 28 p.

Kochenderfer, J.N.; Edwards, P.J.; Wood, F. 1997. Hydrologic impacts of logging an Appalachian watershed using West Virginia's best management practices. Northern Journal of Applied Forestry. 14(4): 207-218.

Kochenderfer, J.N.; Helvey, J.D. 1987. Using gravel to reduce soil losses from minimum-standard forest roads. Journal of Soil and Water Conservation. 42(1): 46-50.

Kochenderfer, J.N.; Wendel, G.W.; Smith, H.C. 1984. Cost of and soil loss on "minimum-standard" forest truck roads constructed in the central Appalachians. Res. Pap. NE-544. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 8 p.

Losche, C.K.; Beverage, W.W. 1967. Soil survey of Tucker County and part of northern Randolph County, West Virginia. Morgantown, WV: U.S. Department of Agriculture, Soil Conservation Service; West Virginia Agriculture Experiment Station. 78 p.

Lull, H.W.; Reinhart, K.G. 1963. Logging and erosion on rough terrain in the East. In: Proceedings of the Federal inter-agency sediment conference. Misc. Publ. 970. Washington, DC: U.S. Department of Agriculture, Agricultural Research Service: 43-47.

Martin, C.W.; Hornbeck, J.W. 1994. Logging in New England need not cause sedimentation of streams. Northern Journal of Applied Forestry. 11: 17-23.

Needham, P.R. 1938. Trout streams. New York: Comstock Publishing Co. 233 p.

Patric, J.H. 1976. Soil erosion in the eastern forest. Journal of Forestry. 74: 671-676.

Pierce, R.S.; Hornbeck, J.W.; Martin, C.W. [and others]. 1992. Whole-tree clearcutting in New England: manager's guide to impacts on soils, streams, and regeneration. Gen. Tech. Rep. NE-172. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 23 p.

Reinhart, K.G.; Eschner, A.R.; Trimble, G.R., Jr. 1963. Effect on streamflow of four forest practices in the mountains of West Virginia. Res. Pap. NE-1. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 79 p.

Trimble, G.R., Jr.; Sartz, R.S. 1957. How far from a stream should a logging road be located? Journal of Forestry. 55(5): 339-341.

U.S. Department of Agriculture. 1940. Truck trail handbook. Washington, DC: U.S. Department of Agriculture, Forest Service, Division of Engineering: 502-559.

Weitzman, S. 1952. Mountain logging. Southern Lumberman. 185: 199-202.

West Virginia Department of Natural Resources, Division of Forestry. 1979. Guidelines for controlling soil erosion and water pollution on logging operations in West Virginia. Charleston, WV: West Virginia Department of Natural Resources, Division of Forestry. 26 p.

West Virginia Department of Natural Resources, Division of Forestry. 1982. Clean streams handbook for loggers: mini-workshop manual for keeping mud out of the creeks. Charleston, WV: West Virginia Department of Natural Resources, Division of Forestry. 34 p.

West Virginia Department of Natural Resources, Division of Forestry. 1997. Best management practice guidelines for controlling soil erosion and water siltation from logging operations in West Virginia. Charleston, WV: West Virginia Department of Natural Resources, Division of Forestry. 29 p.

HARVESTING STRATEGIES FOR INCREASING THE AVAILABILITY AND QUALITY OF HARDWOOD FIBER

Chris B. LeDoux[1]

Abstract—Worldwide demand for wood and wood products will continue to increase as global human population increases. These increasing demands for wood will continue to provide economic incentives for non-industrial private forest-land (NIPF) owners to increase the availability and quality of wood fiber harvested from their lands. The challenge is to encourage and facilitate NIPF owners and other commercial forest industries to support these demands without compromising their own short- and long-term economic and other ownership goals.

Over the years silvicultural researchers have developed methods for growing quality trees faster and in more ecologically, environmentally, and socially acceptable ways. Simultaneously, harvesting researchers have developed an enormous database on alternative logging technologies to harvest timber in economically and environmentally acceptable ways. This study demonstrates that integrating what we know about growing trees with what we know about harvesting them can increase the availability of wood fiber and add value to future crops.

Results for the oak/hickory forest type in West Virginia show that up to 1,736.61 ft^3/acre of wood fiber can be harvested 10 years sooner than usual by simply matching size of machine to wood harvested. Specifically, the study focused on the gains that can be made by matching size of machines to size of wood harvested, by utilizing harvesting machines better and more efficiently, and by training machine operators to be more efficient. Gains of up to 40 percent in present net worth can be attained by early thinning of a stand when harvesting machines are matched to wood size harvested. Results of the study benefit loggers, planners, managers, forest industry, NIPF owners, and society in general.

INTRODUCTION

As global human population increases, worldwide demand for wood and wood products will continue to increase. Because the majority of the hardwood forested land in the United States is owned by non-industrial private forest-land (NIPF) owners (Birch 1996), they will be asked to increase the availability and quality of wood fiber harvested from their lands. The challenge for forest industry and NIPF owners is to meet these demands while simultaneously meeting their own short- and long-term economic and ownership goals (Sampson 1996). Another challenge is to communicate the silvicultural and harvesting technology advances to the forest industry and NIPF owners so they can continue to provide wood products to society in a sustainable manner over time (Cantrell 1996).

Researchers have accumulated volumes of knowledge about how to regenerate and grow trees (Smith and others 1988). We know a great deal about how different species of trees respond to alternative silvicultural treatments. Stocking guides have been developed to maximize tree growth for selected species (Lancaster and Leak 1978, Sampson and others 1980). Over the same time, research in logging methods has been accumulated on production, cost, and applicability for a wide range of cable logging (LeDoux 1985), ground-based (Huyler and LeDoux 1989) and cut-to-length/forwarding machines (Huyler and LeDoux 1996). Harvesting studies in clearcuts, thinnings, shelterwoods, and group-selection applications (LeDoux and others 1991, LeDoux and others 1993) evaluate these different processes and silvicultural systems. We know a great deal about how to regenerate, grow, and harvest trees in environmentally acceptable ways. The need is to integrate what we know about silviculture with what we know about logging technology and then to get the information to loggers, land managers, forest industry, and NIPF owners.

METHODS

Description of ECOST Version 3 and MANAGE

ECOST Version 3 and MANAGE were the models used in this study. ECOST Version 3 is a stump-to-mill logging cost-estimating model for Eastern hardwoods. ECOST Version 3 allows for the stump-to-mill cost estimation for cable and ground-based systems. The difference from previous versions is that it includes skidding cost and production functions for four small farm/skidding tractors and for three skidders with small, medium, and large capacity. Specifically, ECOST Version 3 allows the user to estimate the felling, bucking, limbing, yarding/skidding, loading, hauling, and unloading costs for several cable yarders, small tractors, and skidders. The costs can be estimated in components or as stump-to-mill for most conditions loggers will encounter when logging Eastern hardwood stands.

MANAGE (LeDoux 1986), a computer program written in FORTRAN V, integrates harvesting technology, silvicultural treatments, market price, and economic concerns over the life of a stand. The simulation is a combination of discrete

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and stochastic subroutines. Individual subroutines model harvesting activities, silvicultural treatments, growth projections, market prices, and discounted present net worth (PNW) economic analysis. Specifically, the model allows the manager to evaluate how alternative harvesting technology, silvicultural treatments, market price, and economic combinations affect costs and benefits over the life of a stand. The model uses a detailed, individual user-specified tree list and then projects stand growth based on some user-specified silvicultural treatment, harvests the desired volume or stems with the logging system specified, sells the wood, and conducts economic analysis for the respective treatment and entry. MANAGE was run for a stand in the oak-hickory forest type with alternative combinations of logging technology.

Using ECOST Version 3, the user obtains information on skidding costs for alternative logging machines and for any set of silvicultural/stand conditions. Using MANAGE, the user can project the costs and benefits of alternative combinations of silvicultural treatments and logging technology for any stand of interest.

Harvest Treatments

The harvesting treatments evaluated include no thinning and an area-wide low thinning that removed all trees below an average dbh of 12 inches. The objective for each thinning treatment was to leave the larger crop trees in order to grow quality wood products for the final harvest. The wood harvested was sold as pulpwood and sawlogs. The stand was logged with ground-based logging technology. Specifically, the stand was logged with a JD 440C, JD 540B, and a JD 640D skidder. The John Deere machines are all articulated frame, four-wheel drive cable skidders manufactured in the United States. The John Deere 440C is a 70-horsepower skidder, the 540B is 90-horsepower, and the 640D is 120-horsepower. These three machines are representative of the types of cable skidders found on logging jobs in the Eastern United States.

Site and Stand Conditions

In this study, the stand chosen for demonstration is from the oak/hickory forest type in West Virginia and represents 2,971 acres in total land area. The species mix includes northern red oak (*Quercus rubra* L.), American basswood (*Tilia americana* L.), white ash (*Fraxinus americana* L.) , and black cherry (*Prunus serotina* Ehrh.). The average site index of the stand is about 70. The stand is 60 years old and contains 257 trees per acre that are more than 5 inches dbh. The stand has an average tree dbh of 11.13 inches and about 4,412.42 ft^3/acre of merchantable volume. The land is located on gentle to moderate slopes and requires ground-based systems for harvesting. It is assumed that new road construction is not required. The stand is located 25 miles from a pulpmill/sawmill.

RESULTS

Matching Machines to Wood Size

The thinning was simulated at different stand ages using JD 440C, JD 540B, and JD 640D skidders. The resulting delay-free skidding costs were graphed by machine and the average stand diameter at each age (Fig. 1). The cost curve for the JD 440C is truncated at 12.2 inches because turns containing multiple logs of this tree size exceed the capacity of the machine. At a stump-to-road cost of \$0.20/ft^3, the JD 440C would breakeven when operating in stands that average 6.6 inches dbh. The JD 540B would breakeven in stands that averaged about 8.5 inches, and the JD 640D would breakeven at average dbh of about 9.1 inches. By matching the smaller, less expensive skidder with younger stands, the manager/logger can enter younger stands earlier to conduct the thinning. Using a larger machine such as the JD 640D for the thinnings would require that the stand contain bigger trees before reaching breakeven conditions (Fig. 1). Matching skidding machines to tree size could allow managers/loggers to enter younger stands and capture all the benefits of thinning and yet breakeven. Matching the size of machine to the size wood harvested also makes the wood from the thinnings available to fiber markets earlier in the life of a stand and increases the availability of wood fiber to markets.

Impact of Utilization Rate on Entry Timing

Clearly, few logging operations/machines operate in delay-free environments. Delays range from total machine malfunction resulting in a major breakdown/delay to the machine operator taking too many breaks or failing to service the machine. The thinning was simulated at different ages with a JD 540B skidder at utilization levels of 90, 80, and 60 percent (Fig. 2). Utilization rate is measured as the percentage of working time that the machine is actually being used in a productive mode as opposed to being non-productive while in a delay mode. A machine with a high utilization rate will generally produce more wood volume/unit time and cost less/unit produced than the same machine at a lower utilization rate. For this study, at a cost/ft^3 of \$0.20, the JD 540B at 90 percent utilization would breakeven while operating in stands that averaged about 11.5 inches dbh. For the same machine

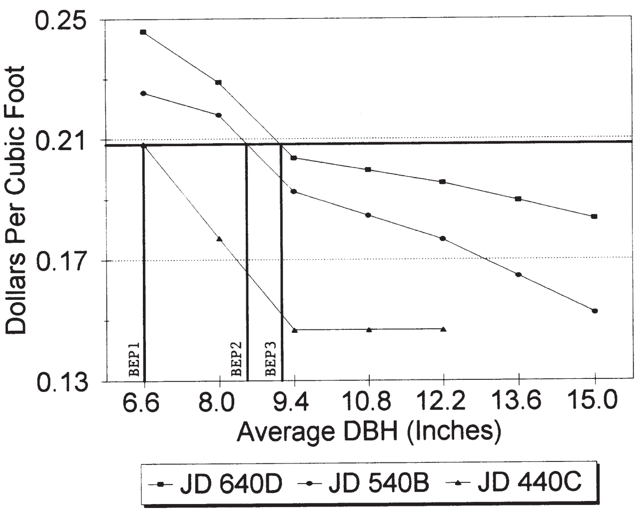

Figure 1—Simulated delay free skidding costs for JD 440C, JD 540B, and JD 640D skidders by average stand dbh (BEP = breakeven point).

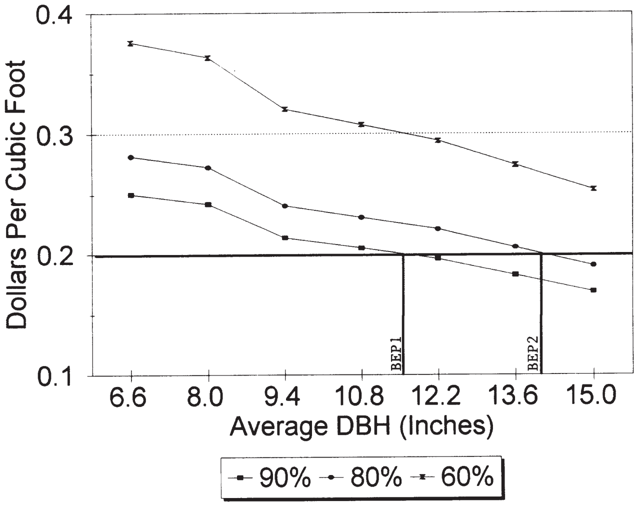

Figure 2—Simulated skidding costs for JD 540B skidder at three utilization levels by average stand dbh (BEP = breakeven point).

and conditions, but at an 80 percent utilization rate, the breakeven point occurs at about 14.0 inches dbh. At a utilization rate of 60 percent, the JD 540B would not breakeven. Operator training strategies to increase the utilization rate of machines can allow managers/loggers to operate in younger/smaller dbh stands, thus making wood fiber available to wood markets earlier in the life of a stand.

Impact on Rotation Age and Wood Quality

The advantages of entering a stand earlier in its life by matching machine size to wood size were further studied by inputting the costs from Figure 2 at the 90 percent utilization level into the MANAGE model. Initially, the stand was not thinned and was projected to its Optimal Economic Rotation Age (ORA). The stand was then thinned using a JD 440C and a JD 540B skidder at the earliest age possible that would result in an economic/breakeven entry to illustrate the impact of matching machines to wood size on rotation age, financial returns, and resulting quality development. The stand was thinned at age 60 with the JD 440C. The stand was thinned at age 70 with the JD 540B because of the higher skidding costs. The thinned stands were then projected to their ORA. The delivered product prices used in this study by species and log quality are shown in Table 1. The results from simulations for the no-thin and thinning treatments with MANAGE are summarized in Table 2. Matching machine size (JD 440C) to wood size would allow the stand to be thinned at age 60 yielding 1,736.61 ft^3/ac of wood fiber. Under the same conditions but thinning the stand at age 70 (with JD 540B), the yield is 1,484.97 ft^3/ac. Using the smaller skidder and thinning all 2,971 acres could yield 5.1 million ft^3 of wood fiber that would be economically available 10 years earlier than if a larger skidder were used in the first thinning. The thinnings do not produce more volume overall, they just make fiber available earlier.

Table 1—Delivered log prices by species and grade, International 1/4-inch (Worthington and others 1996)

Species group	Grade 1	Grade 2	Grade 3	Pulpwood
	- - - - - - - *$/mbf*	*- - - - - - - -*		*$/Cord*
Red oak	561	397	225	40
Basswood	321	239	143	40
Black cherry	571	400	259	40
Ash	420	297	169	40

Table 2—Simulated results by size of skidding machine

Machine	JD 440C	JD 440C	JD 540B
Thinning age (yrs)	No thinning	60	70
Avg. stand d.b.h. (in)	-	8.78	9.13
Trees cut/acre	-	172	134
Vol. removed/ac(ft^3)	-	1736.61	1484.97
Present net worth (PNW-$)[a]	-	38.28	9.11
Cash flow/ac. ($)	-	38.28	12.24
Optimal rotation age (ORA, yrs)	90	100	110
Ave d.b.h. at ORA (in)	14.03	20.32	20.12
Vol/acre at ORA, ft^3	5507.94	4355.94	5047.51
Total vol/acre removed, ft^3	5507.94	6092.55	6532.48
PNW/ac at ORA ($)[b]	1360.67	1872.98	1614.78
Cash flow/ac at ORA ($)	3302.70	6109.72	7079.05

[a] Real discount rate = 3 percent.
[b] Discounted to age 60.

The unthinned stand reached its ORA at 90. The thinned 60-year-old stand would reach its optimal rotation 10 years sooner than if the stand were thinned with the larger skidder at age 70. Both thinned stands would produce wood that would average 20+ inches dbh compared to 14+ inches in the unthinned stand. The larger 20-inch dbh trees would yield higher quality logs than those from the 14-inch stand. Since the thinned 60-year-old stand reaches optimal rotation sooner than the thinned 70-year-old stand, the present net worth (PNW) is $1,872.98 compared to $1,614.78, or an increase of 15.99 percent. This could represent a gain of $853,776.27 in cumulative PNW over the thinned 70-year-old stand if all 2,971 acres were thinned at one time. It is unlikely that all 2,971 acres would be thinned at one time, but for this analysis it demonstrates the potential volume and financial yields possible. The thinned 60-year-old stand produces a cumulative PNW increase of $550.59/acre compared to the nonthinned stand. Although the thinnings do not produce more volume overall, they serve to concentrate the remaining volume on fewer stems but of higher quality.

CONCLUSIONS

Matching machine size to size of wood harvested results in wood fiber available earlier in the life of the stand, shorter ORA for similar size products, and significant gains in PNW—up to 16 percent. Strategies to improve machine utilization also allow managers/loggers to enter stands earlier making wood fiber available earlier and improving the quality/adding value to the future stands. In addition, the combination of carefully matching the size of machines to the size of wood harvested and implementing strategies to reduce skidding delays allows managers/loggers the same benefits. Up to 1,736.61 ft^3/acre of wood fiber can be made available sooner by simply using smaller, less expensive skidders to enter the stands at earlier ages. Thinned stands produce larger dbh/higher quality wood and, thus, larger economic returns compared to unthinned stands. Gains of up to 40.46 percent in discounted cumulative PNW can be realized by early thinning versus no thinning.

In this study, we did not consider the impact of residual stand damage on financial yields over time. We have found that residual stand damage from thinnings can range from none to very high levels. The impact of residual stand damage is best dealt with on a case-by-case basis. Although most NIPF owners own tracts substantially less than 2,971 acres, the results are applicable to small tracts as well. The increased availability of wood fiber along with the value added in quality to the future stand will help meet the world’s demand for fiber and quality hardwoods.

REFERENCES

Birch, Thomas W. 1996. Private forest-land owners of the Northern United States, 1994. Resour. Bull. NE-136. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 293 p.

Cantrell, Rick. 1996. The American Forest & Pulp Association’s sustainable forestry initiative and nonindustrial forest landowners. In: Baughman, M.J., ed. Proceedings, symposium on nonindustrial private forests: learning from the past, prospects for the future; 1996 February 18-20; Washington, DC. St. Paul, MN: University of Minnesota: 397-402.

Huyler, N.K.; LeDoux, C.B. 1989. Small tractors for harvesting fuelwood in low-volume small-diameter hardwood stands. In: Proceedings, 12th annual council on forest engineering meeting; 1989 August 27-30; Coeur ‘D Alene, ID. 6 p.

Huyler, N.K.; LeDoux, C.B. 1996. Cut-to-length harvesting on a small woodlot in New England: a case study. In: Proceedings, planning and implementing forest operations to achieve sustainable forests. Gen. Tech. Rep. NC-186. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Experiment Station. 7 p.

Lancaster, Kenneth F.; Leak, William B. 1978. A silvicultural guide for white pine in the Northeast. Gen. Tech. Rep. NE-41. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station.

LeDoux, Chris B. 1998. ECOST Version 3 - Stump-to-mill production cost equations and computer program. Unpublished report on file at Northeastern Forest Experiment Station, Morgantown, WV.

LeDoux, C.B. 1985. Stump-to-mill timber production cost equations for cable logging eastern hardwoods. Res. Pap. NE-566. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 6 p.

LeDoux, C.B.; Baumgras, J.E.; Sherar, J.; Campbell, T. 1991. Production rates and costs of group-selection harvests with a Christy cable yarder. Forestry and environment ... engineering solutions. ASAE 09-91: 75:84.

LeDoux, Chris B. 1986. MANAGE: a computer program to estimate costs and benefits associated with eastern hardwood management. Gen. Tech. Rep. NE-112. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 7 p.

LeDoux, Chris B.; Erickson, Michael D.; Hassler, Curt C. 1993. Production rates and costs of group-selection harvests with ground-based logging system. In: Proceedings, 9th central hardwood forest conference. Gen. Tech. Rep. NC-161. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 10 p.

Sampson, Neil R. 1996. Keynote address: nonindustrial private forest - learning from the past, prospects for the future. In: Baughman, M.J., ed. Proceedings, symposium on nonindustrial private forests: learning from the past, prospects for the future. 1996 February 18-20; Washington, DC. St. Paul, MN: University of Minnesota: 3-9.

Sampson, T.L.; Barrett, J.P.; Leak, W.B. 1980. A stocking chart for northern red oak in New England. Durham, NH: University of New Hampshire, Institute of Natural and Environmental Resources.

Smith, Clay H.; Perkey, Arlyn W.; Kidd, William E., eds. 1988. Workshop proceeding- guidelines for regenerating Appalachian hardwood stands; 1988 May 24-26; Morgantown, WV. SAF Publ. 88-03. Morgantown, WV: West Virginia University. 293 p.

Worthington, Virginia E.; LeDoux, Chris B.; McWilliams, William H. [and others]. 1996. Methodology for assessing current timber supplies and product demands. Gen. Tech. Rep. NE-226. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 25 p.

Modeling / Inventory

USING DYNAMIC PROGRAMMING TO EXPLORE HARDWOOD SILVICULTURAL REGIMES

Matthew H. Pelkki[1]

Abstract—Forest managers face a difficult problem when required to estimate future physical and economic returns from timber stands when management strategies can vary by number and timing of thinnings, thinning type, thinning intensity and final harvest age. Field-based testing of these variables provides the most reliable information on stand and individual tree response to these practices, but is limited by the time and cost of these studies. Simulation studies based on growth models which derive their tree and stand response data on field observations are one method of exploring the response curve in greater detail and at a much lower cost in time and effort. Dynamic programming is an ideal method for exploring both stand and individual-tree response to new economic conditions and rotation schemes. Dynamic programming can be used to investigate possible responses to different thinning strategies, intensities, and timing of thinnings. It can also provide insights into management responses to changing markets and prices for wood products. However, simulation approaches are hindered by the weakness of long term (40+ years) growth projections and the ability of the computer systems to accurately model real-world constraints. In this paper, a dynamic programming (DP) based computer optimization technique is used and compared with the results of DP-based studies to previously published field studies of yellow-poplar. The similarities found lend credence to the biological reasonableness of the simulations and the ability of the simulation to provide detailed sensitivity analysis demonstrates its strengths in identifying new research questions for field-based studies.

INTRODUCTION — TIMBER MANAGEMENT

Timber managers face a myriad of questions in planning a stand from regeneration to final harvest. Two of the most basic questions center around rotation length and stand density. These two questions are appropriate whether the objective is fiber or financial maximization. To answer these questions, foresters must pull out the “crystal ball” and make predictions, using some form of growth and yield model to estimate future stand characteristics (total volume, diameter distribution, species composition). If financial maximization is desired, then foresters must also attempt to predict future economic conditions such as the cost of capital, inflation, future stumpage prices, and the costs of harvesting. These data are important for making decisions about sustainable forest-wide management decisions as well as good stand-level decisions. For the forest cover types in the central United States, this usually involves planning and prediction of forest and financial conditions on a time frame from 25 to 100 years.

Two approaches have been applied in an effort to provide forest managers with some guidelines for decision making about rotation length and stand density. The first and oldest technique is field testing. Representative sites are selected and management alternatives are actually implemented and the physical results are measured and reported. The second approach has grown in use since the advent of powerful, low-cost computers. Computer simulation of a forest stand is based upon the use of growth and yield models and this is coupled with economics to provide estimations of future outputs.

The following discussion will compare these two approaches and then provide a detailed description and one example of an implementation of dynamic programming which represents one technique for computer simulation of forest stands. One overall assumption will be that the problem under consideration is an even-aged stand, although all techniques are equally applicable to uneven-aged stand structures.

TWO APPROACHES — COMPUTER SIMULATION AND FIELD TESTING

Necessary Resources for Each Method

Actual field testing of silvicultural regimes requires access to representative sites and the ability to actually implement and then observe the results of management actions. Considerable financial, personnel and equipment resources are needed. This method also requires time for the stand to respond to the management actions and for repeated monitoring of stand parameters. It is assumed that environmental conditions (site, climate) will be typical of the region to which the study applies.

Computer simulation, on the other hand, requires good data on initial stand conditions as a starting point for the simulation. These initial conditions may be a selection of actual representative sites or a hypothetical composite site that is considered typical of the region. Growth and yield models that are responsive to the range of actions and stand conditions simulated are necessary. For example, the TWIGS individual-tree growth simulator for the Central United States (Miner and others 1988) was developed from a data base that poorly represented yellow-poplar, Eastern redcedar, and tupelo/gum. Therefore, this growth simulation would not be an optimal choice for computer simulation of these species in the Central United States. Also, whole stand models such as

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yield tables are unable to accurately predict how a stand might grow if the diameter distribution is altered from thinnings from above or below. Predicting future tree quality is also very difficult with whole stand models. Furthermore, computer simulations will need estimates of future financial conditions. Data or additional models that predict stumpage prices, the cost of capital (interest rates), and the cost of harvesting operations must be included.

Strengths and Weaknesses of Each Approach

Table 1 is a list of some major strengths and weaknesses of each approach. Field-based studies have several advantages. They do not rely upon predictions of growth and yield or economic conditions. It is difficult to argue with observed results when properly replicated. However, it should be noted that even observed economic results, when applied in planning future rotations, assume that the future will be the same as the past. The alternatives for management that are chosen can be put into practice with relative ease. Due to the great length of time required and the costs related to acquiring the sites and managing them, field-based studies are expensive. Sensitivity analysis is also difficult and type and number of alternatives is usually kept very low so that adequate replication can be achieved.

Computer simulation, on the other hand, can test a great number of alternatives and combinations. Due to the speed and power of modern personal computers, simulations can test millions of combinations in hours rather than decades and at a far lower cost. However, long-term projections with growth and yield models may be substantially in error (Rauscher and others 1997) and at best might be considered only relatively correct. Another subtle, but important problem with computer simulation is “artifacts.” An artifact represents a difference in a result that is not likely to be observed in the real world. For example, Beck and Della-Bianca developed yield equations for unthinned (1970) and thinned (1975) yellow-poplar stands. The two equations are disjoint at any single given age. When these two models are used together in a computer simulation, when a stand is thinned, it immediately shows a slight increase in volume at the same age! This directs the simulation to thin at an early age to capture this “free” volume which, in reality, does not exist. Another form of an artifact may occur from rounding of values in a simulation. For example, it may be possible to test the sensitivity of thinning in basal area increments of one square foot or less, but would the results be practicable in the real world?

The remainder of this paper presents the basics of one computer simulation technique, dynamic programming (DP). A non-mathematical description of DP will be presented, followed by a mathematical formulation and then a description of a DP-based computer program (NESTER). The results of published research on field-based studies of yellow-poplar will be compared to some NESTER results.

OPTIMIZATION USING DYNAMIC PROGRAMMING

Dynamic programming is a computational method that takes large sequential problems and breaks them down into solvable, related subproblems which can be linked together to achieve an optimal solution to the entire problem. Let’s think about a 20-year-old even-aged hardwood stand. Let us further assume that management operations will occur only once every 10 years, and that the alternatives available are, to do nothing, thin 20 percent or 40 percent of the volume, or clear fell the entire stand (regeneration harvest). Figure 1 shows a diagram with a node or state representing our 20-year-old stand. The state is described by the total cubic foot volume per acre, in this case 700 ft^3/acre. Over the 10 years, this stand will grow 700 ft^3/acre and will have a total of 1400 ft^3/acre. If we do nothing, the stand remains at 1400 ft^3/acre. If we thin, the stand volume will be reduced to either 1120 ft^3/acre (20 percent thinning) or 840 ft^3/acre (40 percent thinning). If we clear fell the stand it has no residual volume. From one state in the initial stage (age 20), we now have four possible states in the first stage. Figure 2 shows a partial diagram of the progression into the second stage. Note that two paths compete for the state described by 1600 ft^3/acre. As each stage continues, we will have more states in the decision space and as that happens, it is more likely that two or more paths will

Table 1—Strengths (+) and weaknesses (–) of field-based studies and computer simulations

Field-based	Computer Simulation
+ Long term growth and yield valid + Economic data is observed, not predicted + Alternatives selected are relatively easy to put into practice – Costly to implement – Difficult to test large numbers of alternatives and combinations/intensities – Long time for study completion	– Long term problems with growth and yield models – Long term economic assumptions can be invalid – Model may reflect artifacts based on structure of model + Inexpensive to test + Excellent sensitivity analysis + Fast results

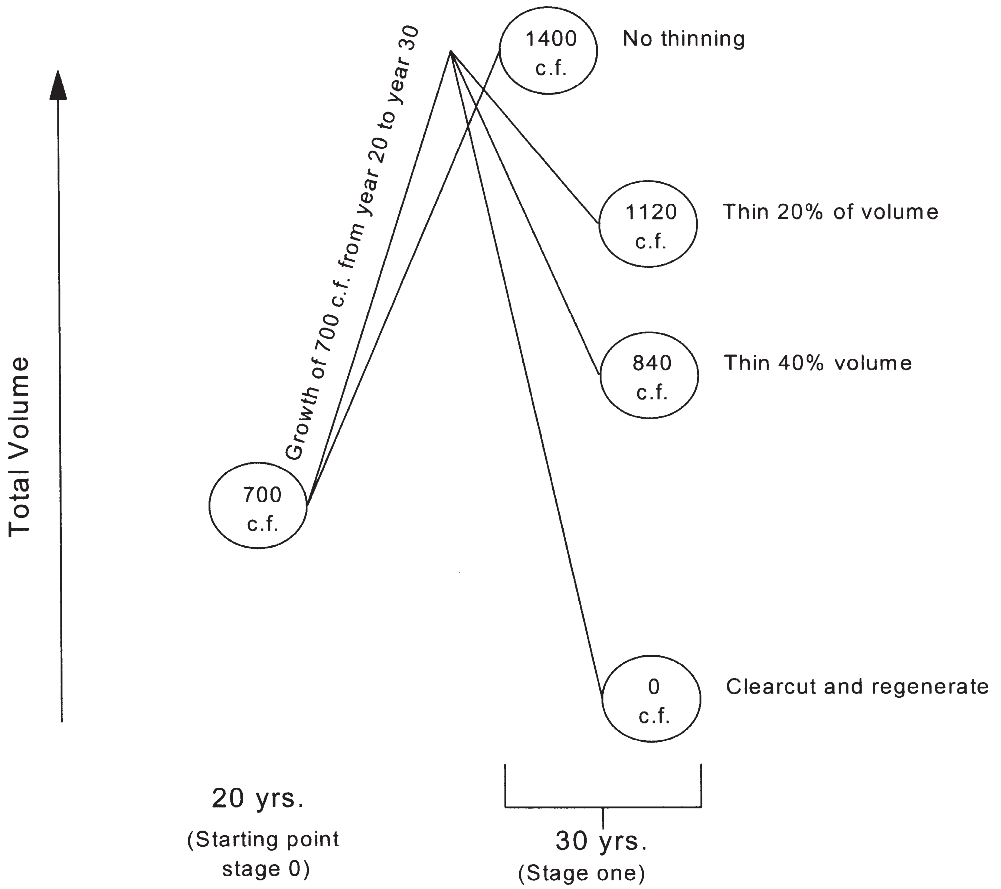

Figure 1—Initial condition and stage one states for forward-recursive dynamic programming network.

compete for a given state. The underlying assumption in dynamic programming is that any two or more paths that compete for the same state represent an identical forest condition with the same future value growth potential. If we assume that it doesn't matter how we achieve 1600 ft^3/acre in a 30-year-old stand, this assumption holds true. This is called the principle of optimality (Dykstra 1984). It is also called the memoryless principle and it means that an optimal path through the network of all possible states can be solved one stage at a time.

More formally, the subproblems are often referred to as stages, and within each stage are multiple states. A recursive relationship links the stages together. This recursive relationship contains a method for choosing optimal states within a stage and for linking optimal stage policies together into an optimal problem solution. To do so, certain information must be stored at each state. This includes 1) a link to a state in an adjacent stage, 2) the objective function value for reaching this particular state, and finally, 3) information necessary to describe the state and make a decision regarding options for moving it to the next adjacent state and determining its value relative to the objective function (Figure 3). The formal mathematics for dynamic programming are provided in the next section.

Dynamic Programming Formulated Mathematically

The objective function shown in equation one relates the value of N management decisions, $f_N^*(Y_N)$, which yield a stand described by Y_N (a regeneration harvest in even-aged stands). A management decision is defined by the variable T_n.

$$f_N(Y_N) = \sum_{n=0}^{N} r_n(T_n) \tag{1}$$

Equation two relates the current state of a stand in stage n to a state in the next stage n+1 ($Y_n \rightarrow Y_{n+1}$) by taking the current stand condition (Yn) and adding the next current growth ($G_{n+1}(Y_n)$) and subtracting out any intermediate harvests (T_{n+1}).

$$Y_n + G_{n+1}(Y_n) - T_{n+1} = Y_{n+1} \qquad (n=0,1,2,\ldots,N-1) \tag{2}$$

Equation three simply relates the volume of a stand after it has grown (X_n) to its final volume in stage n (Y_n) by showing the difference as the amount of volume harvested (T_n).

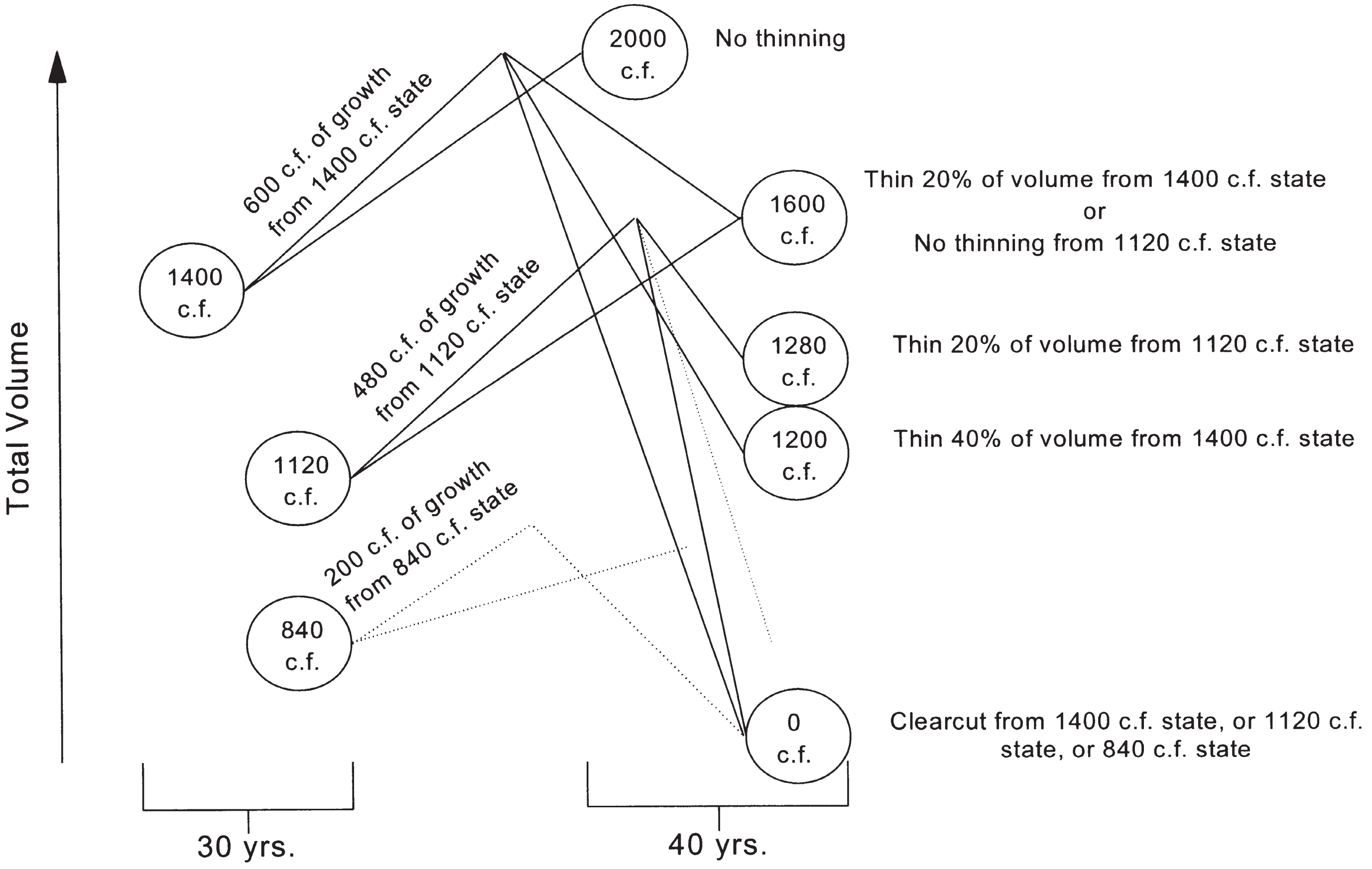

Figure 2—Stages one and two of a dynamic programming network.

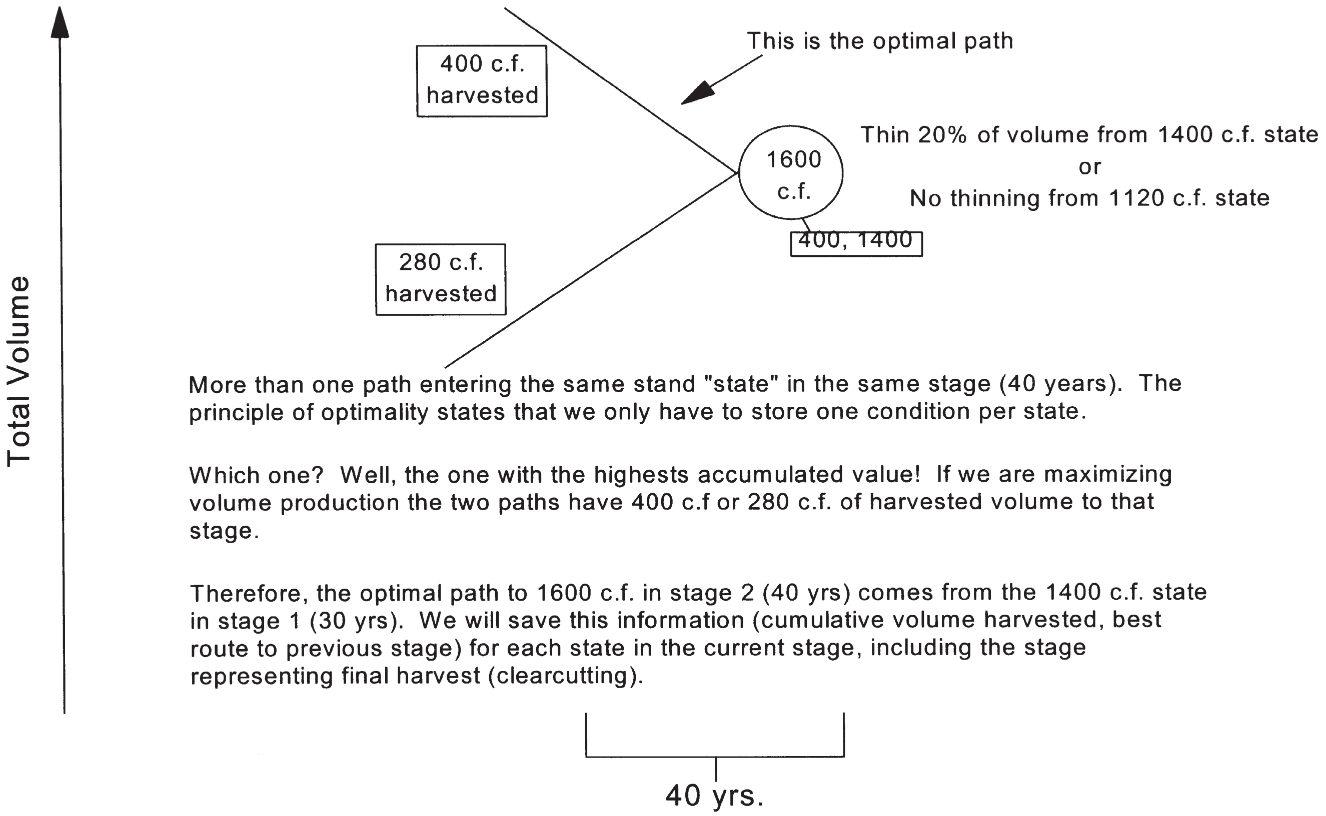

Figure 3—Competition for nodes in the DP network and tracing the optimal path.

$$X_n - T_n = Y_n \qquad (n=0,1,2,...,N) \tag{3}$$

Equation four assures that the final stand condition is a clearcut. If uneven-aged management is applied, then a different ending condition, one with a residual volume, is defined.

$$X_N - T_N = 0 \tag{4}$$

Equation five is the recursive relationship that links each stage together. The function $r_n(X_n,T_n)$ is the return generated at stage n from decision T_n taken on stand described by X_n. The function r can maximize any quantifiable criteria but is usually a maximum volume or value function.

$$f_n^*(Y_n) = \max_{(Y_{n-1},T_n)} [r_n(X_n,T_n) + f_{n-1}^*(Y_{n-1})] \tag{5}$$

The variables are defined again for the reader:

$f_N^*(Y_N)$ = objective function value of N management decisions yielding a stand described by Y_N. In the final stage, N, Y_N is a clear cut stand.

$r_n T_n$ = return generated at stage n by decision T_n.

X_n = state vector describing stand at stage n after it grows from state Y_{n-1}.

Y_n = state vector describing stand at stage n.

T_n = management decision taken at stage n.

$G_{n+1}(Y_n)$ = growth of stand at stage n to stage n+1. Along with T_n, this constitutes the transformation function.

$r_n(X_n,T_n)$ = return generated at stage n from decision T_n taken on stand described by X_n.

Some Comments About Dynamic Programming

A single variable such as total volume is usually insufficient to adequately describe a stand. Two stands, both having a total volume of 1000 ft^3/acre may have very different average diameters and thus different future growth and value potential. Therefore, two or more state variables are used to define a stand condition within a stage. However, this leads to another problem known as the curse of dimensionality (Arthaud 1986). State variables used to describe forest stands are usually continuous. If the variables cubic volume per acre and number of trees per acre are used to define states within a stage and it is assumed that the maximum volume is 5000 ft^3/acre and the maximum number of trees is 500/acre, then the DP program must be able to represent 2.5 million states per stage (assuming that all volumes and trees per acre are rounded to the nearest integer). If yet another state variable is added, the memory requirements are likely to exceed the primary memory capacity of most University mainframe computers. To effectively use continuous variables and to reduce the size of the solution network, the idea of state neighborhoods (Brodie and Kao 1979) was introduced. For example, state neighborhoods of ± 5 trees per acre and ± 5 ft^3/acre would reduce the two state memory requirements from 2.5 million to 100,000 states per stage. Figure 2 demonstrates an example of state neighborhoods. If the neighborhood interval was 100 ft^3/acre, then the paths leading to 1280 ft^3/acre and 1200 ft^3/acre would compete for the same node in the network. The key in using state neighborhoods is making sure that your intervals are not so large as to violate the principle of optimality (Pelkki 1997).

AN EXAMPLE OF DP AND FIELD TESTING - YELLOW POPLAR MANAGEMENT

Description of NESTER

Nester (NEighborhood STate EvaluatoR) is a forward recursive dynamic programming computer program for Windows-based personal computers (Pelkki 1997). It utilizes the GROW subroutine (Brand 1981) from the TWIGS individual-tree growth projection system (Miner and others 1988). NESTER allows the user to choose any combination of six state variables (cubic foot volume per acre, number of trees per acre, average diameter, basal area per acre, number of thinnings, and average tree grade per acre). Stage intervals can be as short as one year or as long as 30 years. As NESTER utilizes an individual-tree growth model, stumpage prices must be provided for each species group included in the stand. NESTER can be used to model mixed species stands, however, at the present time, there are no thinning algorithms that select by species. NESTER does project tree grade using the method presented by Yaussy (1993). Also, because of the individual-tree growth model, NESTER can simulate many types of thinning operations, including mechanical thinning, thinning from above, thinning from below, and thinnings based on tree quality (improvement thinning and high-grading). NESTER can simulate constant, real price increases, or stochastic price changes independently for sawtimber and other roundwood size classes.

NESTER Studies on Yellow Poplar

Two studies have been published to date using NESTER to study yellow-poplar (Pelkki and Arthaud 1998, Pelkki 1999). One study focused upon changing markets and prices and their effects on yellow-poplar management, the other examined changing thinning strategies and the impact on financial returns from yellow- poplar. The results of these studies indicate the strength of dynamic programming for sensitivity analysis and the overall findings of these papers will be compared to published results of field studies reported in Beck and Della-Bianca (1981). The complete results of those studies are not included here. This paper merely highlights some of the new information obtained through computer simulations and identifies some areas where additional field-based work is needed.

Input Data for NESTER

Initial stand conditions were derived from published diameter distributions (McGee and Della-Bianca 1967, Knoebel and others 1986) for 20-year-old, average-stocked, yellow-poplar stands with site indices of 90, 110, and 130 ft at 50 years. Potential grade distributions (Hanks 1976) representing low, average, and high stem quality classes were defined using U.S. Forest Service Forest Inventory and Analysis data for yellow-poplar in the region. Thus, nine different combinations of site index and stem quality distribution served as starting points for the DP simulations.

Stumpage prices were obtained from regional price reports and from Timber Mart South (1994). Capital costs were generally set at 4 percent. Sensitivity to real price increases and higher and lower costs of capital were investigated.

NESTER runs were completed using two states, number of trees per acre and cubic foot volume per acre, with the state neighborhoods of 10 trees per acre and 10 ft^3 per acre, respectively. The stage interval was set at 2 years, but additional runs were investigated using different stage intervals. All thinning options but mechanical thinning were simulated (thinning from above, thinning from below, thinning from above and below, high-grading, and improvement thinning) at intensities ranging from 10 percent to 50 percent of the initial basal area in 5 percent increments. Thus, from each state, 47 options were simulated. The state network represented 46^n+1 states (where n = stage). The initial stage was age 20, and each stage increased by two year increments. Therefore, stage 11 (age = 42) represented $46^{11}+1$ possible ways to manage a stand from age 20 to age 42.

Comparing Results of NESTER-Based Studies to Previous Work

When thinning yellow-poplar stands, Beck and Della Bianca (1981) note that cultural work in sapling and pole-sized stands is very costly and there are few markets for this material. When thinnings are done, Beck and Della-Bianca (1981) recommend thinnings from below to concentrate the value on larger, high-value stems. They also noted that intermediate yellow-poplar trees would respond to thinning from above. In a recently completed DP study, Pelkki (1999) found that improvement thinnings, while earning negative to very low initial returns (-$5.1 to $31.5 per acre) were commonly part of the optimal financial stand regime. These improvement thinnings removed first large trees of low quality, then very small trees, and finally low to higher grade factory sawtimber grade trees until a basal area target was reached. In this manner, the trees with the most valuable future growth potential were retained.

In stands with high-quality stem distributions, the DP-based studies (Pelkki and Arthaud 1998, Pelkki 1999) included thinnings that removed 40-50 percent of the basal area. Furthermore, some of the optimal financial regimes included 4-6 thinnings (not all at the 40-50 percent intensity level). Beck and Della-Bianca (1981) report considerable leeway in manipulating yellow-poplar stocking levels to achieve diameter growth and quality goals without sacrificing volume production. However, the frequency and intensity of the DP-based harvest may initiate advance regeneration which would increase the final harvest and site regeneration costs.

In studies reported in Beck and Della-Bianca (1981), first thinnings are recommended as early as 15-20 years and can be repeated every 5-15 years over the rotation. The DP-based studies (Pelkki and Arthaud 1998, Pelkki 1999) have an initial stand state that is 20 years of age and so cannot simulate thinnings before the age of 22 years (1st stage). However, in the DP-studies, most initial thinnings occurred between the ages of 22 and 28 years. Multiple thinnings in the DP-based studies ranged from 2-12 years apart with most being 4-10 years apart.

In regimes with multiple thinnings, Beck and Della-Bianca (1981) recommend that later thinnings be lighter because basal area growth response in older stands is lower. The DP-studies (Pelkki and Arthaud 1998, Pelkki 1999) had heavier later thinnings, possibly due to the financial, rather than volume objectives of the simulation. Both Beck and Della-Bianca (1981) and the DP-based studies (Pelkki and Arthaud 1998, Pelkki 1999) found that intense thinnings shortened the rotation.

While not explicitly discussed in Beck and Della-Bianca (1981), the yield tables suggest an optimal rotation of 50-70 years for fiber production. The DP-based studies (Pelkki and Arthaud 1998, Pelkki 1999) found maximum financial returns with rotations as short as 32 years or as long as 66 years. Factors contributing to shorter rotations were a good market for pulpwood, OSB, and other fiber-based products, high interest rates, and higher site indices. Factors contributing to longer rotation lengths were high stem quality distributions, lower interest rates, and higher sawtimber prices (or an absence of sub-sawtimber markets). These factors favored longer, sawtimber-focused rotations.

For yellow-poplar management, the DP-based studies uncover three major issues that need additional research in a field-based setting. First and foremost, can improvement thinnings, based on potential tree grading bring the economic returns projected by the computer simulations? Additional studies have shown (Pelkki and Ringe 1998) that the earlier a valid potential tree grade can be applied, the greater the economic returns. Secondly, are harvest costs in poletimber stands prohibitive? The DP-simulations assume a fixed entry cost on all thinnings and a 15 percent thinning penalty on all stumpage prices (to reflect the added cost of thinning over a final harvest). They also charge harvest costs for pre-merchantable stems at the rate of 10 percent of the merchantable price (if the stems were of merchantable size). Studies by Kluender and others (1996) suggest that harvest cost of small diameter stems are greater than proportional to volume in southern pine stands. With the increase in markets for hardwood pulpwood throughout the central Appalachian region, thinning and site preparation cost studies would appear necessary. Finally, the DP-based simulations suggest that multiple thinning regimes with 4-6 entries prior to final harvest may optimize financial returns in some yellow-poplar stands. Some field-based tests of such regimes would answer questions related to problems in stand integrity and advance regeneration treatment costs.

SUMMARY - WHY BOTH APPROACHES ARE NEEDED

Unquestionably, both field-based and simulation-based approaches are necessary for forest researchers in the future. Field-based studies provide on-the-ground confirmation of new practices and working methods for field application. They cannot, however, respond quickly to new markets or changing economic conditions. While computer simulations can perform thorough sensitivity analyses to many variables, they lack the real world operational constraints that are often too complex to model and may even be unanticipated by the researcher. Computer simulations are best left to exploratory research with their results confirmed by field-based studies which can then lead to changes in actual management practices.

REFERENCES

Arthaud, G.J. 1986. Economic comparisons of thinning from above and below in loblolly pine plantations using dynamic programming. Blacksburg, VA: Virginia Polytechnic Institute and State University. 139 p. M.S. thesis.

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

Beck, D.E.; Della-Bianca, L. 1975. Board-foot growth and yield of yellow-poplar after thinning. Res. Pap. SE-123. U.S. Department of Agriculture, Forest Service. 20 p.

Beck, D.E.; Della-Bianca, L. 1981. Yellow-poplar: characteristics and management. Agric. Handb. 583. U.S. Department of Agriculture, Forest Service. 91 p.

Brand, G.J. 1981. “GROW” - A computer subroutine that projects the growth of trees in Lake States’ forests. Res. Pap. NC-207. U.S. Department of Agriculture, Forest Service.

Brodie, J.D.; Kao, C. 1979. Optimizing thinning in Douglas-fir with three descriptor dynamic programming to account for accelerated diameter growth. Forest Science. 25: 665-672.

Dykstra, D.P. 1984. Mathematical programming for natural resource management. McGraw-Hill. 318 p.

Hanks, L.F. 1976. Hardwood tree grades for factory lumber. Res. Pap. NE-333. U.S. Department of Agriculture, Forest Service. 81 p.

Kluender, R.; Lortz, D.; McCoy, W. [and others]. 1996. Harvesting time, costs, and productivity in Arkansas. In: Caulfield, J.; Bullard, S., eds. Proceedings of 1995 southern forestry economics workshop; April 17-19; New Orleans, LA: 260-274.

Knoebel, B.R.; Burkhart, H.E.; Beck, D.E. 1986. A growth and yield model for thinned stands of yellow-poplar. For. Sci. Monograph 27. 62 p.

McGee, C.E.; Della-Bianca, L. 1967. Diameter Distributions in natural yellow-poplar stands. Res. Pap. U.S. Department of Agriculture, Forest Service. 7 p.

Miner, C.L.; Walters, N.R.; Belli, M.L. 1988. A guide to the TWIGS program for the North Central United States. Gen. Tech. Rep. NC-125. U.S. Department of Agriculture, Forest Service.

Pelkki, M.H. [In press]. Impacts of thinning strategies on financial returns from yellow-poplar stands. Southern Journal of Applied Forestry.

Pelkki, M.H. 1997. The effects of neighborhood storage size on dynamic programming solutions. Forest Science. 43(3): 387-395.

Pelkki, M.H.; Arthaud, G.J. 1998. Evaluating changes in yellow-poplar management in response to new low-end markets. Northern Journal of Applied Forestry. 15(1): 37-42.

Rauscher, H.M.; Young, M.J.; Webb, C.D. 1997. Testing the accuracy of growth and yield models for southern hardwood forests. Unpublished manuscript. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station, Bent Creek Research and Demonstration Forest.

Timber Mart South. 1994. First and second quarter reports. Athens, GA: University of Georgia. Vol. 19.

Yaussy, D. 1993. NE-TWIGS update: incorporating tree quality. The Compiler. Vol. 11(1): 8-12.

USE OF GPS AND GIS IN HARDWOOD FOREST INVENTORY

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Abstract—Recent advances in GPS satellite survey, geographic information systems, and a portable electronic distance measurement device are useful new tools which, when combined with classical tree measurement, timber volume calculation, statistical inventory procedures, provide a pathway for generating highly accurate digital timber stand information for effective forest management decision makings. This presentation describes the methods and procedures used in the design of an intensive timber inventory conducted in a mountainous forest watershed.

In this prototype study, GPS, laser ranging, and GIS techniques are used to replace traditional forest inventory tools such as chains, compasses, altimeters, and optical forks. GPS land navigation technique is employed to locate inventory plot centers and a differential GPS point positioning technique is used to permanently determine surveying plot locations. A laser ranging instrument, called Criterion, is used to measurement upper-stem diameters and subsequently to determine merchantable lengths of commercial trees. Georeferenced timber survey data and computer generated environment data such as slope and aspect maps are stored in a geographic information system for subsequent analyses.

The inventory uses a traditional systematic line-plot design to tally timbers in an eastern hardwood forest situated on the western slope of the Appalachian mountain. Using Criterion's laser ranging capability, plot trees are identified by measuring the distance between plot and tree centers. For a plot tree, its dbhob is measured by a caliper and its merchantable length is measured by the criterion. The Criterion's diameter and height measurement functions allow foresters to accurately determine the position of a prescribed upper-stem diameter and the measure the vertical distance between two points on tree stems. The digital display feature provided by this instrument eliminates the need to interpret analogous scales on conventional tree measurement and portable surveying equipment. Tree measurement data are subsequently inputted into a personal computer for the computations of both board-foot and cubic-foot saw-timber volumes. All these measurements exceed functional accuracies specified for intended forest inventory work.

Using statistic estimation methods, plot data are expanded to provide interval estimates of timber stocking on a per acre basis or for the entire forest stand. In addition, all inventory data are entered into a GIS framework which contains environmental data such as aspect and slope maps. The GIS is used to store, display, and query of timber stands information for effective forest management decision makings.

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USE JAVA AND THE INTERNET TO MANAGE DATA AND PREDICT THE FUTURE OF FOREST STANDS

J.J. Colbert and George Racin[1]

Abstract—A Java based software package provides a user with Internet access the ability to store, summarize, and translate field collected data describing forest stands and to run simulations to assess possible future scenarios that include stand management and the effects of gypsy moth defoliation. The model interface is constructed as a Java applet that will run within a Java-compliant Internet world-wide-web browser, allowing a user from any computer that has Internet and web access to use these tools. The model user also has the ability to access data from a large collection of example stands that are available from a server-side database. If the user has a locally available Java Virtual Machine (JVM), then it is possible to use a Java application version of these software programs that permits the local storage, access, and management of input and output data files generated through use of the program. Data from field plots entered through the user interface are summarized on a per acre basis and from these data, the user can directly obtain estimates of stocking, volume, and value, as well as stand structure and related habitat data summaries. Simulations permit the user to look at potential growth scenarios under hypothesized management actions.

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ESTIMATING PREVIOUS DIAMETER FOR INGROWTH TREES ON REMEASURED HORIZONTAL POINT SAMPLES

Susan L. King and Stanford L. Arner[1]

Abstract—The purpose of this study was to develop an improved procedure to estimate a previous diameter for ongrowth and nongrowth trees on plots sampled by variable radius plot sampling. However, the models built can be used whenever a previous diameter is required and the appropriate independent variables are available. Data for this study were from 1,965 remeasured forest inventory plots in the 1989 inventory of West Virginia conducted by the Northeastern Research Station's Forest Inventory and Analysis (NEFIA) unit. The investigation focused on three areas. First, we investigated whether breaking the data into six groups based on the rank average diameter growth was superior to NEFIA's procedure of breaking the data into seventeen species groups. Second, we investigated whether basal-area increment (BAI) was superior to diameter increment (DI) as a dependent variable. Finally, we investigated additional independent variables. Based on the R^2, mean residual, mean absolute error, and root mean square error, the best model had six species groups, BAI as the dependent variable, and a slightly expanded set of independent variables.

INTRODUCTION

Missing observations are common in "real world" data sets. In forestry, there are many instances in which a diameter for a previous period is required but not available. The impetus for this study was a problem encountered by the Forest Service's Forest Inventory and Analysis (FIA) units which are required to produce periodic tables showing net volume change from the previous inventory as well as the components of change: ingrowth (I), survivor growth (S), mortality (M), and cut (C). Equation (1) expresses net change as the sum of these components:

$$V_2 - V_1 = I + S - M - C \qquad (1)$$

where:

V_1 = volume at time period 1;

V_2 = volume at time period 2.

Martin (1982) defined six possible categories of trees encountered on remeasured point samples. The first four, ingrowth, survivor, mortality, and cut trees were measured at the first inventory. The remaining categories, ongrowth and nongrowth trees, were alive and included only in the second inventory. Ongrowth trees were nonmerchantable (below minimum dbh) at the first inventory but of merchantable size at the second inventory. Nongrowth trees were above minimum dbh at the first inventory, but grew sufficiently to be included in the second inventory.

Most FIA units have used multiple subplot horizontal point samples (prism plots) to inventory trees larger than a specified merchantable diameter. With this type of sample, the probability of selecting a sample tree is proportional to the basal area of the individual tree and depends on the basal-area factor of the prism.

Traditional estimation procedures for total net change and its components yielded estimates for which the two sides of equation (1) do not agree because ongrowth and nongrowth trees were excluded as components of growth. Martin (1982) obtained compatible estimators by including in ingrowth both ongrowth and nongrowth trees. Van Deusen and others (1986) improved the compatible estimators by rearranging Martin's equation to include nongrowth with survival growth. They showed that the standard error of this new estimator of survival growth was smaller than Martin's traditional estimator. Also, the estimator for ingrowth used by Van Deusen and others cannot be negative. Roesch and others (1989) showed that the estimation of survival growth could be improved with additional rearrangement of some components. To distinguish between ongrowth and nongrowth trees, these new procedures require the estimation of previous diameter for trees that were not measured at the initial inventory.

NEFIA unit has used several plot designs, including a 10-point variable radius design. To obtain compatible estimates of change with these plots, we need to estimate previous diameter for trees measured only at time period 2.

To estimate previous diameter, NEFIA currently uses trees measured at both inventories to develop regression equations. The model is:

DI = f (DBH2, TRCLS2, CRNCLS2, CRATIO2, CRCC2, DCR2) (2)

where:

DI = annual diameter increment;

DBH2 = tree diameter at time period 2;

TRCLS2 = tree class at time period 2, a measure of tree quality;

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CRNCLS2 = crown class at time 2, a measure of crown position in the canopy;

CRATIO2 = crown ratio at time 2, the proportion of a tree with a live crown;

CRCC2 = CRATIO2/CRNCLS2;

DCR2 = DBH2 · CRATIO2.

The data are divided into seventeen species groups. For each group, a stepwise backward elimination regression procedure finds the functional relationship between the dependent and independent variables. Annual DI is the dependent variable as opposed to total increment between inventories because the trees are remeasured at different intervals. The procedures developed by Roesch and others (1989) and Van Deusen and others (1986) assume that good estimates of previous diameter for ongrowth and nongrowth trees are possible. With the NEFIA procedure, the R^2 for each of the seventeen species groups is low, *e.g.*, the most recent inventory from West Virginia had R^2 values ranging from 0.06 to 0.56.

PROCEDURE

The data for this study are from 1,965 remeasured forest inventory plots in West Virginia (DiGiovanni 1990). Trees were initially measured in 1975 and remeasured in 1988. Only trees larger than 5 inches in diameter at both inventories were used. The data were split into a model data set with 8,723 observations and a validation data set with 7,951 observations.

Using multiple linear regression, we investigated different sets of independent variables, different species-group compositions, and BAI as a dependent variable. Although the explanatory variables have biological meaning with respect to competitive position, size, and tree quality, the model itself is not easily interpreted biologically as is the case with models developed by Teck and Hilt (1991) and Quicke and others (1994). However, our objective was to find a better estimate of annual DI for trees alive at both inventories. It was not necessary to account for ingrowth, cut, or mortality.

Apart from the variables used by NEFIA, we investigated additional variables reported in the literature as important in modeling tree growth. The additional independent variables were restricted to those measured by NEFIA and those that could be calculated:

BA2 = total plot basal area of trees at least 5 inches in diameter at time 2;

TPA2 = total number of trees per acre of trees at least 5 inches in diameter at time 2;

MD2 = plot medial diameter of trees at least 5 inches in diameter at time 2;

QD2 = plot quadratic diameter of trees at least 5 inches in diameter at time 2;

BAL2 = total basal area of trees larger than the sample tree at time 2;

RLD2 = relative diameter; ratio of tree diameter to medial diameter of plot, MD2;

RLQD2 = relative diameter; ratio of tree diameter to quadratic diameter of plot, QD2.

The plot variables, BA2, TPA2, MD2, and QD2, are expressions of site occupancy or total competition and size of the trees on the plot. The tree variables, BAL2, RLD2, and RLQD2, are measures of the competitive position of the sample tree relative to other trees on the plot.

The basal area larger (BAL2) variable is used in the potential growth times modifier type of model found in the NC and NE-TWIGS growth simulators (Hilt and Teck 1989: Miner and others 1988: Quicke and others 1994). Marquis (1991) used relative diameter to model diameter growth.

Variables not considered are tree or stand age and a measure of site productivity. NEFIA does not determine the age of individual trees, and stand age was excluded because many of the plots are classified as uneven-aged. Also, experience has shown a poor relationship between diameter growth and the site-productivity measure determined by NEFIA.

We investigated both DI and BAI as the dependent variable. Because of unequal number of years between plot measurements, annual increment was modeled as:

$$DI = \frac{(DBH2 - DBH1)}{N} \tag{3}$$

or

$$BAI = \frac{(BA2 - BA1)}{N} \tag{4}$$

where:

N = number of years between measurements on the plot;

DBH1 = tree diameter at time period 1;

DBH2 = tree diameter at time period 2;

BA1 = tree basal area at time period 1;

BA2 = tree basal area at time period 2.

The seventeen species groups used by NEFIA are based on form class and are used in our volume equations (Scott 1979). To investigate whether another grouping of species is more appropriate to model diameter growth, we formed six species groups based on rank of the average diameter growth for a species. Mean annual diameter growth ranged from 0.068 for the lowest ranked group to 0.203 for the highest.

RESULTS

We compared the R^2 values, mean residuals, mean absolute residuals, and root mean square errors using the NEFIA independent variables for the total sample of both the model and validation data sets (Table 1). The comparison is for six procedures: 1) the stepwise procedure on each of seventeen species groups using DI as the dependent variable; 2) all variables for each group using DI; 3) all variables ignoring species group using DI;

Table 1—R^2s, mean residuals, mean absolute residuals, and root mean square errors (MSE) for the stepwise procedure using NEFIA variables with three species groupings

Procedure	R^2	Mean residual	Mean abs. residual	Root MSE
		Model data set		
Diameter increment				
Stepwise, 17 groups	0.348	0.0	0.0530	0.0692
Nonstepwise, 17 groups	0.349	0.0	0.0529	0.0691
Nonstepwise, no groups	0.211	0.0	0.0584	0.0762
Nonstepwise, 6 groups	0.351	0.0	0.0530	0.0691
Basal-area increment				
Nonstepwise, 17 groups	0.557	0.0033	0.0440	0.0571
Nonstepwise, 6 groups	0.565	0.0013	0.0436	0.0566
		Validation data set		
Diameter increment				
Stepwise, 17 groups	0.312	0.0017	0.0544	0.0721
Nonstepwise, 17 groups	0.312	0.0017	0.0545	0.0721
Nonstepwise, no groups	0.207	0.0022	0.0585	0.0775
Nonstepwise, 6 groups	0.322	0.0005	0.0545	0.0716
Basal-area increment				
Nonstepwise, 17 groups	0.536	0.0032	0.0451	0.0592
Nonstepwise, 6 groups	0.553	0.0017	0.0448	0.0582

4) all variables using six species groups and DI; 5) all variables using seventeen species groups with BAI as the dependent variable; and 6) all variables using six species groups and BAI. Equation (5) and (6) give the formulas for calculating the R^2 and root mean square errors, respectively, for the combined data.

$$R^2 = 1 - \left(\frac{SSRES}{SSTOT}\right) \tag{5}$$

$$RMSE = \sqrt{\frac{SSRES}{n_i}} \tag{6}$$

where:

$$SSRES = \sum_{i=1}^{g}\sum_{j=1}^{n_i}\left(Y_{ij} - \hat{Y}_{ij}\right)^2;$$

$$SSTOT = \sum_{i=1}^{g}\sum_{j=1}^{n_i}\left(Y_{ij} - \overline{Y}\right)^2;$$

Y_{ij} = observed DI;

$\hat{Y}_{ij}$ = predicted DI;

$\overline{Y}$ = overall mean DI;

n_i = number of trees in group i;

g = number of species in a group, 17, 6, or 1.

The predicted value for the validation data set, $\hat{Y}_{ij}$, was obtained using the coefficients estimated with the model data set.

The stepwise procedure for the seventeen species groups offers little advantage over the nonstepwise procedure for each group. There is little difference in any of the statistics between these two procedures. Each of the NEFIA independent variables was significant in several of the groups, indicating that eliminating one or more of the variables would be inappropriate. Yet there is an advantage in grouping the species. All of the statistics for both the model and validation data sets show improvement when the data are divided into species groups. However, there is little difference between the results using the seventeen groups based on form class and the six groups based on rank of mean DI. As seen in Table 1, there is a marked improvement when BAI replaces DI as the dependent variable. For the BAI model, all results are expressed in reference to DI using the translation:

$$\hat{DI}(BA) = \frac{\sqrt{\frac{(BA1 + N \cdot \hat{BAI})}{K}} - DBH1}{N} \tag{7}$$

where:

$\hat{BAI}$ = predicted BAI;

K = 0.005454154, a conversion factor from diameter in inches to basal area in square feet.

All of the statistics use $\hat{DI}(BA)$ as the predicted DI for the basal-area model.

Table 2 presents a more complete analysis of the differences between results using DI versus BAI for the

Table 2—Comparison statistics for the diameter-increment and basal-area increment models for model and validation data sets

		Diameter-increment model					Basal-area increment model			
Species group	No. of trees	Mean DI	R^2	Mean residual	Mean abs. residual	Root MSE	R^2	Mean residual	Mean abs. residual	Root MSE
Model data set										
1	247	0.0681	0.045	0.0	0.0322	0.0408	0.235	0.0011	0.0288	0.0365
2	947	0.0929	0.085	0.0	0.0415	0.0525	0.315	0.0004	0.0359	0.0454
3	2489	0.1145	0.157	0.0	0.0434	0.0562	0.370	0.0012	0.0374	0.0486
4	2371	0.1434	0.201	0.0	0.0556	0.0714	0.471	0.0010	0.0452	0.0582
5	783	0.1595	0.219	0.0	0.0564	0.0720	0.473	0.0021	0.0464	0.0591
6	1876	0.2026	0.212	0.0	0.0689	0.0880	0.513	0.0022	0.0540	0.0692
All groups	8723	0.1416	0.351	0.0	0.0528	0.0691	0.565	0.0013	0.0435	0.0566
Validation data set										
1	97	0.0700	0.001	0.0018	0.0337	0.0413	0.204	0.0022	0.0300	0.0369
2	616	0.0953	0.079	0.0024	0.0425	0.0546	0.315	0.0029	0.0368	0.0471
3	2417	0.1146	0.183	0.0001	0.0436	0.0557	0.392	0.0017	0.0373	0.0481
4	2448	0.1435	0.161	0.0016	0.0561	0.0732	0.450	0.0018	0.0455	0.0592
5	509	0.1662	0.115	0.0043	0.0598	0.0788	0.401	0.0027	0.0497	0.0649
6	1864	0.2030	0.201	-0.0021	0.0700	0.0898	0.514	0.0014	0.0551	0.0701
All groups	7951	0.1455	0.322	0.0005	0.0545	0.0716	0.552	0.0018	0.0447	0.0582

procedure with six-species groups. Although the bias (as expressed by the mean residual) is slightly greater for the BAI model, the R^2, mean of absolute residuals, and root mean square error are substantially smaller. Mean annual diameter growth (mean DI) for each species group also is included in Table 2.

Table 3 gives the results for different combinations of independent variables for the six species groups. Some of the independent variables discussed previously were added to the NEFIA variables. The models listed are the best 1, 2, and 3 variable combinations. The additional variables express plot occupancy and tree competitive position. In the remaining three models, CRCC2 and CRNCLS2 were deleted. In each of the three models, BAL2, RLD2, or RLQD2 was substituted for CRNCLS2. These variables measure a tree's competitive position relative to the other trees on a plot.

The results for the models with variables substituted for or added to the FIA variables are mixed. Each of the four statistics presented in Table 3 indicate a different "best" model. All differences are small and no model represents a substantial improvement over the others.

Table 3—Comparison statistics for models with variables substituted for or added to NEFIA variables using six-group BAI model and model data sets

Variable	R^2	Mean residual	Mean abs. residual	Root MSE
NEFIA[a]	0.565	0.0013	0.0436	0.0566
DBH2, TRCLS2, CRATIO2, DCR2, BAL2	0.575	0.0014	0.0431	0.0559
DBH2, TRCLS2, CRATIO2, DCR2, RLD2	0.562	0.0010	0.0438	0.0567
DBH2, TRCLS2, CRATIO2, DCR2, RLQD2	0.567	0.0010	0.0435	0.0564
NEFIA + BAL2	0.578	0.0016	0.0429	0.0557
NEFIA + BAL2 + BA2	0.581	0.0015	0.0428	0.0555
NEFIA + BAL2 + BA2+TPA2	0.581	0.0015	0.0428	0.0555

[a] NEFIA= DBH2, TRCLS2, CRNCLS2, CRATIO2, CRCC2, DCR2.

We used the BAI model with six species groups and NEFIA variables to determine how the model performed over the range of tree sizes. The mean residual, mean absolute residual, and root mean square error were determined for 2-inch diameter classes (Table 4). As indicated by the mean residual, for the validation data set there is a small average over prediction for diameters less than 9 inches and greater than or equal to 23 inches. For the diameter classes in between there is a small underprediction.

DISCUSSION AND CONCLUSIONS

With our method there was a marked improvement over current NEFIA procedures in all model valuation statistics except for the mean residual. For the validation data set, the change in the dependent variable from annual DI to annual BAI produced a 72 percent increase in R^2, a 17 percent decrease in mean absolute residual, and an 18 percent decrease in root mean square error.

Grouping by species was better than not grouping them. The grouping based on ranked diameter growth showed only a slight improvement over the seventeen NEFIA form class groups for both the DI and BAI models. Other groupings were not investigated.

There was only a slight improvement when even the best of the other procedures and models was used. As a result, there seems little reason to find a “best” set of independent variables for each group using a stepwise procedure as the outcome using all variables for all groups is essentially the same.

The variables considered as substitutions for or additions to the NEFIA variables produced little improvement. Approximately the same results could be obtained using any of the different combinations of NEFIA and the other plot and tree variables considered here.

Quicke and others (1994) developed a biologically interpretable model using a function with potential growth multiplied by a growth modifier for a single species with plots chosen for specified characteristics. For this study, trees are located on plots chosen at random from a wide range of forest conditions. Disturbance on the plot was not taken into account. There were 78 species and a wide range in size. The results presented here closely approximate those obtained by Quicke and others (1994).

Data used by Teck and Hilt (1991) are from the same type of unstructured design as those presented in this study. They also developed and used a biologically interpretable DI model. With their validation data set, the overall mean prediction error was 0.013 and the root mean square error was 0.085. These statistics are substantially higher than ours.

The coefficients of the models developed here can be used for purposes other than that of NEFIA if the independent variables included in the model are measured. Table 5 contains coefficients for the six-species-group BAI model using NEFIA variables plus BAL2. Table 6 lists the tree species assigned to each of six species groups.

The coefficients developed from this study should be applied to other regions with caution.

Table 4—Comparison statistics by diameter class for six-group basal-area increment model

		Model data set					Validation data set			
Dbh class	No. of trees	Mean DI	Mean Residual	Mean abs. residual	Root MSE	No. of trees	Mean DI	Mean residual	Mean abs. residual	Root MSE
In.										
5 - 6.9	529	0.0573	-0.0043	0.0231	0.0293	490	0.0565	-0.0034	0.0230	0.0289
7 - 8.9	1353	0.0930	-0.0018	0.0313	0.0394	1136	0.0965	-0.0010	0.0323	0.0398
9 - 10.9	1371	0.1246	0.0025	0.0412	0.0518	1202	0.1259	0.0028	0.0401	0.0508
11 - 12.9	1316	0.1401	0.0011	0.0458	0.0577	1179	0.1466	0.0042	0.0454	0.0592
13 - 14.9	1250	0.1587	0.0049	0.0468	0.0614	1087	0.1576	0.0026	0.0487	0.0636
15 - 16.9	946	0.1676	0.0008	0.0483	0.0615	918	0.1741	0.0046	0.0506	0.0651
17 - 18.9	702	0.1787	0.0027	0.0532	0.0680	643	0.1901	0.0089	0.0543	0.0696
19 - 20.9	470	0.1903	0.0064	0.0532	0.0686	480	0.1845	0.0006	0.0564	0.0719
21 - 22.9	296	0.1909	0.0007	0.0554	0.0711	263	0.1882	0.0003	0.0525	0.0665
23 - 24.9	180	0.2018	0.0056	0.0532	0.0698	222	0.1921	-0.0044	0.0580	0.0709
> 25	310	0.1913	-0.0044	0.0499	0.0614	331	0.1851	-0.0103	0.0532	0.0641
All trees	8723	0.1416	0.0013	0.0436	0.0566	7951	0.1455	0.0017	0.0448	0.0582

Table 5—Regression coefficients for six-group basal-area increment model using NEFIA variables and BAL2

Species group	Intercept	DBH2	TRCLS2	CRATIO2	CRNCLS2	DCR2	CRCC2	BAL2
1	0.00107	0.00065	0.00011	0.00028	-0.00043	0.00005	-0.00059	-0.000014
2	0.01448	0.00028	-0.00127	0.00053	-0.00225	0.00020	-0.00367	-0.000033
3	0.02058	0.00064	-0.00309	-0.00115	-0.00272	0.00025	-0.00301	-0.000042
4	0.02239	0.00048	-0.00104	-0.00159	-0.00365	0.00041	-0.00486	-0.000048
5	-0.00150	0.00160	-0.00132	-0.00306	0.00041	0.00021	0.00582	-0.000055
6	0.00132	0.00214	-0.00405	-0.00458	0.00219	0.00016	0.00957	-0.000110

Table 6—Tree species assigned to one of six species groups based on rank of mean diameter increment

Species group	Common name	Scientific name
1	Hawthorn	*Crataegus sp.*
	Shortleaf pine	*Pinus echinata*
	Flowering dogwood	*Cornus flordia*
	European alder	*Alnus glutinosa*
	Table mountain pine	*Pinus pungens*
	Black willow	*Salix nigra*
	Chinkapin oak	*Quercus muehlenbergii*
	Eastern hophornbeam	*Ostrya virginiana*
	Silver maple	*Acer saccharinum*
	Blackgum	*Nyssa sylvatica*
	Willow oak	*Quercus phellos*
	Sourwood	*Oxydendrum arboreum*
2	Maple sp.	*Acer sp.*
	Quaking aspen	*Populus tremuloides*
	Pitch pine	*Pinus rigida*
	Overcup oak	*Quercus lyrata*
	Post oak	*Quercus stellata*
	Red spruce	*Picea rubens*
	Pin cherry	*Prunus pennsylvanica*
	Am. hornbeam, Musclewood	*Carpinus caroliniana*
	Osage-orange	*Maclura pomifera*
	River birch	*Betula nigra*
	Yellow birch	*Betula alleghaniensis*
	Basswood sp.	*Tilia sp.*
	Black walnut	*Juglans nigra*
	Virginia pine	*Pinus virginiana*
	Mockernut hickory	*Carya tomentosa*
	Sweet birch	*Betula lenta*
	Common persimmon	*Diospyros virginiana*
3	Bur oak	*Quercus macrocarpa*
	Ohio buckeye	*Aesculus glabra*
	Pignut hickory	*Carya glabra*
	Hickory sp.	*Carya sp.*
	American beech	*Fagus grandifolia*
	Magnolia sp.	*Magnolia sp.*
	Chokecherry	*Prunus virginiana*
	Buckeye, Horsechestnut	*Aesculus sp.*
	Chestnut oak	*Quercus prinus*
	Sassafras	*Sassafras albidum*
	White basswood	*Tilia heterophylla*
	Shagbark hickory	*Carya ovata*
	Black locust	*Robinia pseudoacacia*
4	Hackberry	*Celtis occidentalis*
	White oak	*Quercus alba*
	Apple sp.	*Malus sp.*
	Eastern hemlock	*Tsuga canadensis*
	Bitternut hickory	*Carya cordiformis*
	Yellow buckeye	*Aesculus octandra*
	Slippery elm	*Ulmus rubra*
	Swamp white oak	*Quercus bicolor*
	Cucumbertree	*Magnolia acuminata*
	American basswood	*Tilia americana*
	Red maple	*Acer rubrum*
	American elm	*Ulmus americana*
	Sugar maple	*Acer saccharum*
	Shellbark hickory	*Carya laciniosa*
	Striped maple	*Acer pennsylvanicum*
	Eastern red-cedar	*Juniperus virginiana*
5	Black oak	*Quercus velutina*
	White ash	*Fraxinus americana*
	Butternut	*Juglans cinerea*
	Elm sp.	*Ulmus sp.*
	Sycamore	*Platanus occidentallis*
	Green ash	*Franxinus pennsylvanica*
	Boxelder	*Acer negundo*
	Black cherry	*Prunus serotina*
	Eastern white pine	*Pinus strobus*
	Scarlet oak	*Quercus coccinea*
6	Northern red oak	*Quercus rubra*
	Bigtooth aspen	*Populus grandidentata*
	Yellow-poplar	*Liriodendron tulipifera*
	Pin oak	*Quercus palustris*
	Prunus sp.	*Prunus sp.*
	Ailanthus	*Ailanthus altissima*
	Black maple	*Acer nigrum*
	Eastern redbud	*Cercis canadensis*
	Southern red oak	*Quercus falcata v. falcata*

REFERENCES

DiGiovanni, D.M. 1990. Forest statistics for West Virginia—1975 and 1989. Resour. Bull. NE-114. U.S. Department of Agriculture, Forest Service. 172 p.

Hilt, D.E.; Teck, R.M. 1989. NE-TWIGS: An individual-tree growth and yield projection system for the Northeastern United States. The Compiler. 7(2): 10-16.

Marquis, D.A. 1991. Independent effects and interactions of stand diameter, tree diameter, crown class, and age on tree growth in mixed-species, even-aged hardwood stands. In: Proceedings, 8th central hardwood forest conference. Gen. Tech. Rep. NE-148. U.S. Department of Agriculture, Forest Service: 442-458

Martin, G.L. 1982. A method for estimating ingrowth on permanent horizontal sample points. Forest Science. 28(1): 110-114.

Miner, C.L.; Walters, N.R.; Belli, M.L. 1988. A guide to the TWIGS program for the North Central United States. Gen. Tech. Rep. NC-125. U.S. Department of Agriculture, Forest Service. 105 p.

Quicke, H.E.; Meldahl, R.S.; Kuch, J.S. 1994. Basal area growth of individual trees: a model derived from a regional longleaf pine growth study. Forest Science. 40(3): 528-542.

Roesch, F.A., Jr.; Green, E.J.; Scott, C.T. 1989. New compatible estimators for survivor growth and ingrowth from remeasured horizontal point samples. Forest Science. 35(2): 281-293.

Scott, C.T. 1979. Northeastern forest survey board-foot volume equations. Res. Note NE-271. U.S. Department of Agriculture, Forest Service. 3 p.

Teck, R.M.; Hilt, D.E. 1991. Individual tree-diameter growth model for the Northeastern United States. Res. Pap. NE 649. U.S. Department of Agriculture, Forest Service. 10 p.

Van Deusen, P.C.; Dell, T.R.; Thomas, C.E. 1986. Volume growth estimation from permanent horizontal points. Forest Science. 32(2): 415-422.

NEURAL NETWORKS VS. MULTIPLE LINEAR REGRESSION FOR ESTIMATING PREVIOUS DIAMETER

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Abstract—A neural network is a nonparametric statistical modeling procedure known for its capacity to process nonlinear relationships. For estimating the previous diameter of a tree, the exact functional relationship between the response variable and the independent variables is unknown. The relationship is most likely nonlinear. Multiple linear regression was used to develop a model for estimating the previous diameter of trees in West Virginia. The data were split into a model data set with 8,723 observations and a validation data set with 7,951 observations. The dependent variable was either basal-area increment or diameter increment. Two different sets of independent variables were evaluated. The data were divided into six species groups based on the rank of the average diameter growth of the species. Basal-area increment was a superior dependent variable for the multiple linear regression model. Basal-area increment is a nonlinear transformation of the diameter increment. It was thought that neural networks with its capacity to capture nonlinear relationships might provide an equivalent or superior solution with the diameter increment response variable as opposed to the basal-area increment response variable. All of the basal-area increment models had a higher R^2 than their diameter increment counterparts. Neither technique was superior in all cases. Other issues such as the stopping criteria, initial weight selection, the optimal number of hidden nodes, and the optimal number of hidden layers in a neural network are also discussed.

INTRODUCTION

The goal of a neural network is to mathematically model the brain and to capture its pattern recognition capabilities. Humans are more efficient at processing pattern information such as speech and visual images than any machine, whereas computers are extremely fast at processing information that can be formulated into a sequence of instructions.

A neural network may provide a superior solution over a traditional statistical approach for certain classes of problems (Burke, 1991). These classes include problems in which the distributions are unknown and possibly nonlinear, where outliers may exist, and where noise is present in the data. These are common conditions in forest inventory data. This paper investigates whether neural networks provide improved estimates over the traditional statistical modeling procedure of multiple linear regression for estimating diameter of a tree at an earlier time period.

In multiple linear regression, the relationship between independent and dependent variables is assumed to be linear and interactions among the independent variables must be specified in advance by the user. In neural networks, there is no assumed relationship between the independent and dependent variables. The relationship between independent and dependent variables and the interactions among the independent variables are learned through an iterative process. Neural networks require no assumptions about the distributions, mean, or correlation of the errors.

PROBLEM

The Northeastern Forest Inventory and Analysis (NEFIA) unit of the USDA Forest Service currently uses trees measured at current and previous inventories to develop multiple linear regression equations to estimate previous diameter for those trees not recorded at the previous inventory. The impetus for this study was to develop an improved procedure to estimate a previous diameter for ongrowth and nongrowth trees on plots sampled by variable radius plot sampling. However, the models built can be used whenever a previous diameter is required and the appropriate dependent variables are available. For West Virginia, King and Arner (1998) developed a new procedure to estimate a previous diameter. They used six groupings based on the rank of the average diameter growth for a species. Also in their study, many different independent variables and combinations of independent variables were evaluated. To investigate whether neural networks can provide a better estimate of previous diameter, the best models from the King and Arner study were selected for comparison.

DATA

The data for this project came from 1,965 remeasured forest inventory plots in West Virginia. The trees were initially measured in 1975 and then remeasured in 1988. Only trees larger than 5 inches in diameter at both time periods were included. The data were randomly split into a model building data set with 8,723 observations and a validation data set with 7,951 observations. Before splitting the data, they were grouped into six species groups. There were two sets of independent variables chosen for comparison. The first set of variables are those currently used by NEFIA to estimate the previous diameter. These variables are:

DBH2 = tree diameter at time period 2;

TRCLS2 = tree class at time period 2, a measure of tree quality;

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CRNCLS2 = crown class at time 2, a measure of crown position in the canopy;

CRATIO2 = crown ratio at time 2, the proportion of a tree with a live crown;

CRCC2 = CRATIO2 / CRNCLS2;

DCR2 = DBH2 • CRATIO.

The analysis by King and Arner showed that the addition of the variable BAL2 improved the results. BAL2 is the sum of the basal areas of the trees on a plot larger than the subject tree. It is a measure of the competition for light. The addition of the variable BAL2 to the first set of independent variables formed the second set of independent variables. Two response variables were investigated: diameter increment (DI) and basal area increment (BAI).

$$DI = \frac{(DBH2 - DBH1)}{N} \qquad (1)$$

or

$$BAI = \frac{K \cdot (DBH2^2 - DBH1^2)}{N} \qquad (2)$$

where:

N = number of years between measurements on the plot;

DBH1 = tree diameter at time period 1;

DBH2 = tree diameter at time period 2;

K = 0.005454154, a conversion factor from diameter in inches to basal area in square feet.

Annual increment was used to account for variation in the measurement period among the plots. Basal area and dbh are related by a transformation. Logic would suggest that BAI would be a better response variable than DI. There is not a one-to-one mapping between DI and BAI. A poletimber and a sawtimber tree may have the same DI, but different diameters at both measurement periods. BAI captures the differences in the size of the trees. Thus, 24 models were compared for both multiple linear regression and neural networks.

NEURAL NETWORKS

The type of a neural network chosen for this study is a feedforward backward propagation network (Figure 1). The network consists of three layers: the input layer, hidden layer, and output layer. The layers consist of processing units called nodes. Arcs connect the layers. Each arc has a weight which represents the strength of the connection. The goal of a neural network is to find the best estimate of the weights.

In the input layer, the number of nodes corresponds to the number of independent variables. In the hidden layer, not only is the number of nodes variable, but also there may be more than one hidden layer. Only one hidden layer is shown in Figure 1. In general, only one hidden layer is required. The third layer is the output layer. The number of nodes in this layer corresponds to the number of dependent variables. A backpropagation network has no cycles. All of the arcs move from left to right.

Each observation in the data set forms an input pattern, p. An observation is called an exemplar in neural networks. A linear combination of the input patterns and the weights is formed at each hidden node. This defines a plane in N - 1 dimensional space, where N is the number of input nodes. The hyperplane passes through the origin unless a bias weight is added to the hidden node. Mathematically, this process is represented as:

$$net_j^p = w_j + \sum_{i=1}^{I} w_{ij} x_i^p \qquad \text{for } j=1,...,J \qquad (3)$$

where:

w_j = the bias term for hidden unit j;

w_{ij} = the weight from input node i to hidden node j;

x_i^p = i^th^ component of the p^th^ exemplar;

I = number of input nodes;

J = number of hidden nodes.

By creating a dummy node with a fixed input value of 1 or -1, the bias can be written as a weight, $w_{I+1,j}$. Thus, equation (3) becomes:

$$net_j^p = \sum_{i=1}^{I+1} w_{ij} x_i^p \qquad \text{for } j=1,...,J. \qquad (4)$$

A squashing or activation function is applied to net_j^p at each hidden node j. This function introduces nonlinearities into the network. Common squashing functions are the logistic and hyperbolic tangent function. Applying the squashing function to net_j^p yields:

$$y_j^p = f(net_j^p) \qquad \text{for } j=1,...,J. \qquad (5)$$

The output from each of the j nodes at the hidden layer becomes the input to the k output nodes.

A linear combination of the output from the hidden nodes and the weights, v_{jk}, is formed. As before, a bias term is added. Mathematically, this may be expressed as:

$$net_k^p = \sum_{j=1}^{J+1} v_{jk} y_j^p \qquad \text{for } k=1,...,K \qquad (6)$$

where:

v_{jk} = the weight from hidden node j to output node k.

A squashing function is applied to net_k^p to obtain the predicted output:

$$o_k^p = f(net_k^p) \qquad \text{for } k=1,...,K. \qquad (7)$$

Backpropagation is a supervised procedure. That is, it requires an observed dependent variable, t_k^p. The estimated value, o_k^p, is compared with t_k^p, to determine if they are close. One measure of closeness is the sum of the

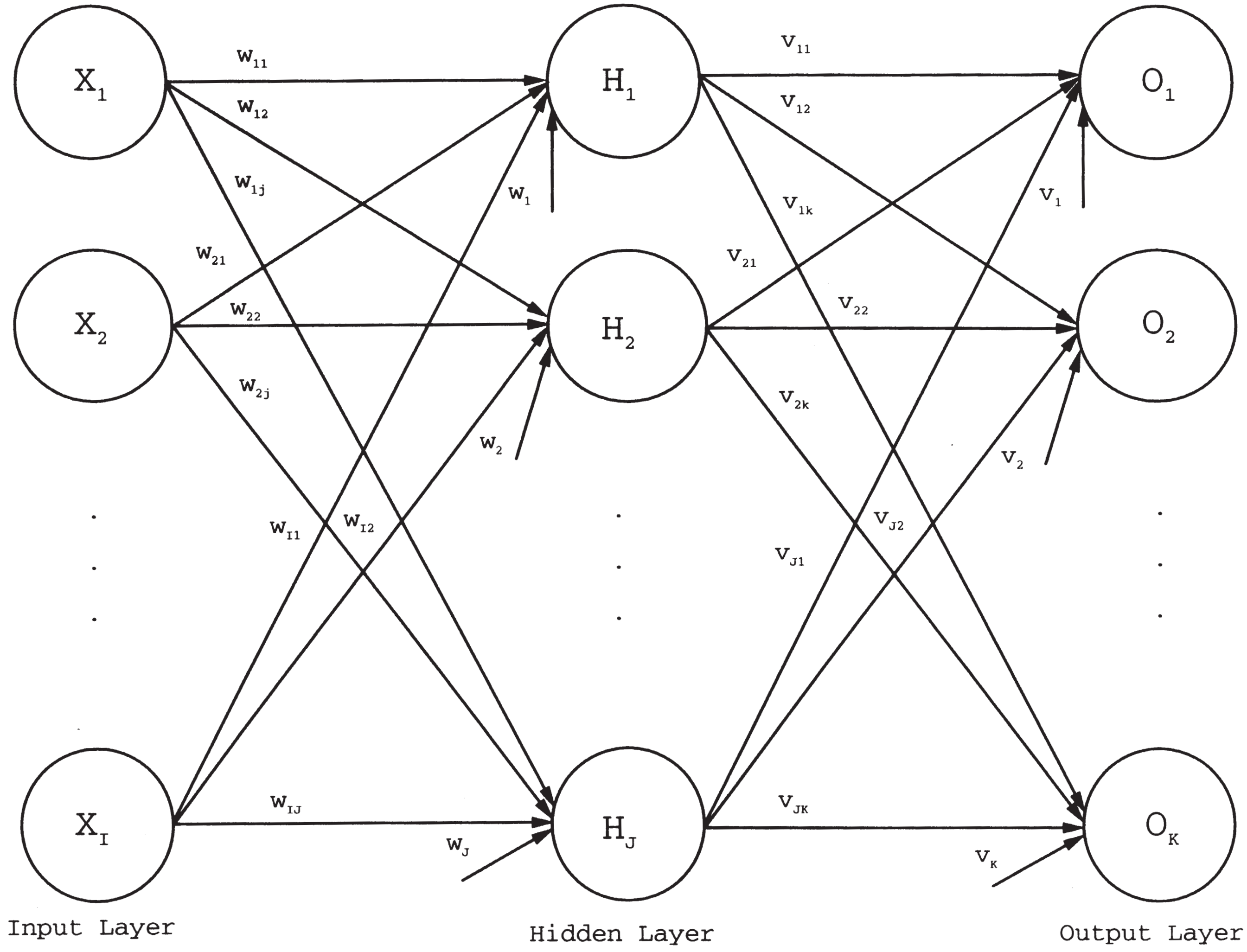

Figure 1—Backpropagation network. The network is fully connected. There is an arc from every node to a node in the next layer.

squared differences between t_k^p and o_k^p. It is used frequently because the derivatives are easy to compute. Thus, the objective function is:

$$\mathrm{Min} \sum_{p} \sum_{k=1}^{K} (o_k^p - t_k^p)^2 . \tag{8}$$

The objective function for neural networks in equation (8) appears to be similar to that in multiple linear regression. Both techniques minimize the sum of the squared differences between the observed and the predicted values. The variables in both procedures are the weights. However, the two procedures are quite different. The predicted values are different functions of the weights. The number of weights in multiple linear regression depends on the number of input variables, whereas the number of weights in neural networks depends on the number of: input variables, hidden nodes, hidden layers, and output nodes. The objective function in both the neural network and multiple linear regression is an unconstrained minimization problem. The special structure in the multiple linear regression problem allows for the optimal set of weights to be found through solving a system of normal equations. Iterative techniques are used to find the optimal set of weights for a neural network. Each iteration is considered a training period. By updating the weights, the neural network is said to be learning.

Many techniques are available for solving unconstrained minimization problems. These techniques include gradient descent, the quasi-Newton techniques of conjugate gradients and Davidon-Fletcher-Powell, the modified Newton technique of Levenberg and Marquardt, stiff differential equations, genetic optimization, and simulated annealing. Historically, gradient descent has been applied by the neural network community to solve equation (8). In fact the name backpropagation refers to the process of applying the chain rule of calculus to compute the error gradient for each weight in the network. The error gradient is used in updating the weights in gradient descent. The error is said to be propagated backwards. Gradient descent may be advantageous if the problem is implemented on a parallel computer. However, most problems are implemented on a serial computer, and gradient descent on these machines has been abandoned by the optimization community in favor of more sophisticated techniques. The difficulties of gradient descent are well documented. Sarle (1994), Masters (1995), Bishop (1995), and Bazarra, Sherali and Shetty (1993) all discuss the limitations of gradient descent and are excellent references on the conjugate gradient method, the Davidon-Fletcher-Powell, and the Levenberg-Marquardt algorithm. Kollias and Anastassiou (1988) discuss applying the Levenberg-Marquardt algorithm to neural networks. Owens and Filken (1989) saw the similarity between a system of stiff differential equations and the gradient descent approach. They claim that stiff differential equations provide a more rapid and accurate convergence than either gradient descent or conjugate gradient methods. Hassoun (1995) provides an introduction to simulated annealing and genetic optimization for neural networks. Masters (1995) is another good introductory reference for simulated annealing in neural networks. Another text by Masters (1993) provides introductory material on both topics.

In selecting the technique to solve equation (8), the number of weights must be taken into account. Sarle (1994) recommends using the Levenberg-Marquardt algorithm for networks with tens of weights, the Davidon-Fletcher-Powell algorithm for networks with hundreds of weights, and the conjugate gradient procedure for large problems with thousands of weights. Most of the 24 subproblems were solved using the Levenberg-Marquardt algorithm. These subproblems had only tens of weights. The other two techniques were tried on a few subproblems, but the value of their objective functions was larger. Gradient descent was also tried on several of the subproblems. It proved to be a superior technique for only the response variable DI in the first subgroup. The gradient descent algorithm was user written in SAS/IML(SAS Institute Inc., 1989). The other three algorithms are part of PROC NLP (SAS Institute Inc., 1997) in SAS/OR. SAS had a Beta release macro that was available in Release 6.10. This macro was modified and used in this study. Updated SAS neural network macros with a GUI interface are now part of the SAS Data Mining Solution.

Several other issues relating to the implementation of the neural network need to be discussed. First, the activation function must be selected. The purpose of an activation function is to induce nonlinearity into the network through a nonlinear transformation. With a linear function, the output is a weighted sum of the inputs. A squashing function is an activation function that maps any real input into a bounded range, usually between 0 and 1 or between -1 and 1. The two most common squashing functions are the logistic function and the hyperbolic tangent function. Other functions may be used so long as they are differentiable. Smooth activation functions decrease the training time. Kalman and Kwasny (1992) argue that the hyperbolic tangent function was the best of the sigmoidal functions. Bishop (1995) states that the hyperbolic tangent function often increases algorithm convergence over a logistic function. The choice of an activation function may be different at the output node. The range of the activation function at the output node should correspond to the range of the dependent variable. A categorical dependent variable would have a different activation function than a continuous output variable. The output data for the diameter increment problem were scaled. The hyperbolic tangent function was used at both the hidden and output layers. The logistic function was tried on some of the subproblems and it did not provide a superior solution.

Second, there are two different philosophies concerning the scaling of the independent and dependent variables. Some believe that it is not necessary to scale the independent or dependent variables. The size of the weights will make any necessary adjustments. In the diameter increment problem, the input and output variables were scaled to values that correspond to the range of the squashing function. Because the hyperbolic tangent function is used as the squashing function, the continuous variables are scaled between -0.9 and 0.9. The endpoints, -1 and 1, in the range are not used because they correspond to the inputs $-\infty$ and $+\infty$, respectively. The class variables are first broken into indicator variables and then scaled like the continuous variables. The lowest value of the indicator variable corresponds to -0.9 and the largest value corresponds to 0.9.

Third, the selection of the initial weights is a major issue. None of these techniques guarantee a global minimum. The choice of initial weights can influence the quality of a local solution. The initial weights in this project were selected by random numbers. There are many heuristic procedures for selecting the initial weights. One procedure by Piovoso and Owens (1991) was tried on several subproblems. In this procedure, the weights between the input layer and the hidden layer are found by principal component analysis. The weights between the hidden layer and the output layer are found by multiple linear regression. In this project, after trying several different seeds, a random number generator always provided a set of weights that found a lower value to the objective function than the Piovoso and Owens procedure.

Fourth, the number of layers must be selected. Hornik, Stinchcombe, and White (1989) showed that a neural network with one hidden layer and an arbitrary squashing function can approximate most functions. In practice, the need for a second hidden layer occurs when a piecewise-continuous function must be approximated. This condition is not present in the diameter or basal-area growth problem, so only one hidden layer was used.

Fifth, there are no equations or formulas for selecting the optimal number of hidden nodes. Each situation is different. Too many hidden units cause an inability to generalize and the data are overfitted. Similarly, too few hidden nodes will cause an underfitting of the data. An underfitted or an overfitted model does not generalize well, that is, predict accurate dependent or output variables from a new set of independent or input variables. One way to control generalization is through the selection of the number of hidden units and their connections. This is model selection. The simplest model selection technique is to determine the optimal number of hidden nodes through experimentation. Starting with one or two nodes, a solution is found. Another node is added and the problem is reoptimized. This process is repeated until the objective function value begins to increase. The number of hidden nodes corresponding to the minimal objective function value is optimal. This procedure was used in the diameter increment model. In most of the 24 problems, two hidden nodes were optimal. An alternative is to start with a large number of hidden nodes and gradually remove complete hidden units or only remove selected connections. This is pruning. Reed (1993) describes several pruning procedures.

Sixth, in any iterative algorithm, a major issue is when to stop. The error in the model data set monatonically decreases as a function of the iteration number. The error in the validation data set decreases and then increases as the neural network starts to overfit. The algorithm is terminated when the validation data set reaches its minimum. This procedure is called stopped training. Regularization procedures such as stopped training improve generalization by controlling the size of the weights. Other regularization procedures such as weight decay, training with noise, and Bayesian estimation are described by Bishop (1995). Stopped training was used in this project.

COMPARISON STATISTICS

The statistics used to compare multiple linear regression with neural networks for both the model and the validation data set are R^2, the mean of the squared errors (MSE), the mean of the absolute errors (MAE), and the mean of the arithmetic errors (ME). The ME indicates bias, whereas the MSE and the MAE both indicate precision as well as bias. The MAE is more robust and less sensitive to outliers than the MSE.

RESULTS AND DISCUSSION

A comparison of the results between neural networks and multiple linear regression for the ranked mean species groups is presented in Tables 1 and 2. The overall results were obtained by combining the response values and the predicted response values for each of the six species groups and forming one group. The appropriate statistics are then calculated. Because the goal of a model is generalization, the results for the validation data set are more important than those for the model data set.

For the BAI models, all of the results are expressed as DI using the translation:

$$\hat{DI}(BA) = \frac{\sqrt{\dfrac{(BA1 + N \cdot B\hat{A}I)}{K}} - DBH1}{N} \tag{9}$$

where:

BA1 = tree basal area at time period 1, K • $DBH1^2$;

$B\hat{A}I$ = predicted basal-area increment;

N = number of years between measurements of the tree;

All of the statistics use $\hat{DI}(BA)$ as the predicted diameter increment for the basal-area models.

From the overall results for the response variable DI, neural networks was superior. It had a slightly higher R^2, a slightly lower MAE, and a slightly lower MSE. For the individual species groups for the response variable DI, neural networks was superior for the NEFIA variables, but the results were mixed for addition of BAL2. Still, neural networks predominates. The ME was larger for the neural network model as expected. An assumption of multiple linear regression is that the expected value of the errors is zero. Neural networks, on the other hand, is a nonparametric procedure and the mean of the arithmetic error will not necessarily be zero. The addition of the variable BAL2 improved the results for both neural networks and multiple linear regression. For the response variable BAI, multiple linear regression more frequently provided a superior solution than neural networks as indicated by the R^2, MAE, and MSE.

The software used in this study was the first beta version of SAS's neural network macros. Later beta versions were significantly enhanced. A subset of the 24 subproblems was selected to access the impact of the new software on the diameter increment prediction problem. The subset had BAI as the response variable and NEFIA + BAL2 as the independent variables. Neural networks had the most difficulty with this subset. The results are in Table 3. With the new macros, neural networks is the winner for the model data set. The R^2 is higher for all species groups, and the MAE and MSE are lower for most of the species groups. However, the new macros did not improve the results for the validation data set. Only the three smallest species groups have a higher R^2 and a lower MAE in the validation data set as compared with the results in Table 2. Neural networks performed slightly worse for the third and sixth species groups. The new macros did not significantly alter the results; and so, it was decided not to pursue modeling the remaining subgroups. Also other independent variables from the King and Arner (1998) study were tried on this subset of the 24 problems. They did not improve the results.

Neural networks does not always outperform multiple linear regression. This conclusion was also reached by Desai and Bharati (1998). They found that for predicting excess returns on large stocks, neural networks outperformed multiple linear regression in periods of high volatility. Otherwise, multiple linear regression was superior. These results parallel a study by Markham and Rakes (1998). Using computer generated data, Markam and Rakes conclude that there is a significant interaction between sample size and variance. Multiple linear regression performs better for low-variance problems and neural networks performs better for high-variance problems. The results are mixed and dependent on sample size for medium-variance problems. Neural networks was superior for large sample sizes, and multiple linear regression was superior for small sample sizes. One explanation of why neural networks did not outperform multiple linear regression in this project is that there is not enough

Table 1—Comparison statistics for neural networks and mutiple linear regression for NEFIA variables

Sub-group	No. of trees	NN	REG	NN	REG	NN	REG	NN	REG
		----------R^2----------		----------ME----------		--------MAE--------		---------MSE---------	
		DI response variable and model data set							
1	257	0.0463	0.0449	-0.0051	0.0000	0.0334	0.0322	0.0017	0.0017
2	947	0.1099	0.0846	0.0004	0.0000	0.0409	0.0415	0.0027	0.0028
3	2489	0.1649	0.1568	0.0016	0.0000	0.0430	0.0434	0.0031	0.0032
4	2371	0.2359	0.2011	0.0019	0.0000	0.0541	0.0556	0.0049	0.0051
5	783	0.2449	0.2193	0.0004	0.0000	0.0557	0.0564	0.0050	0.0052
6	1876	0.2440	0.2116	0.0050	0.0000	0.0664	0.0689	0.0074	0.0077
All	8723	0.3731	0.3512	0.0020	0.0000	0.0517	0.0528	0.0046	0.0048
		DI response variable and validation data set							
1	97	0.0256	0.0011	-0.0020	0.0018	0.0335	0.0336	0.0017	0.0017
2	616	0.1106	0.0787	0.0011	0.0024	0.0419	0.0425	0.0029	0.0030
3	2417	0.1973	0.1832	0.0034	0.0001	0.0436	0.0436	0.0031	0.0031
4	2448	0.2262	0.1606	0.0013	0.0016	0.0544	0.0561	0.0049	0.0054
5	509	0.1184	0.1148	0.0105	0.0043	0.0620	0.0598	0.0062	0.0062
6	1864	0.2334	0.2015	0.0044	-0.0021	0.0686	0.0700	0.0077	0.0081
All	7951	0.3521	0.3216	0.0032	0.0005	0.0537	0.0545	0.0049	0.0051
		BAI response variable and model data set							
1	257	0.2481	0.2349	-0.0011	0.0011	0.0293	0.0288	0.0013	0.0013
2	947	0.2806	0.3155	0.0029	0.0004	0.0366	0.0359	0.0022	0.0021
3	2489	0.4005	0.3704	-0.0003	0.0012	0.0366	0.0374	0.0022	0.0024
4	2371	0.4679	0.4706	0.0007	0.0010	0.0456	0.0452	0.0034	0.0034
5	783	0.4371	0.4734	-0.0044	0.0021	0.0491	0.0464	0.0037	0.0035
6	1876	0.4353	0.5127	-0.0181	0.0022	0.0607	0.0540	0.0055	0.0048
All	8723	0.5418	0.5647	-0.0039	0.0013	0.0451	0.0435	0.0034	0.0032
		BAI response variable and validation data set							
1	97	0.1311	0.2036	0.0097	0.0022	0.0301	0.0300	0.0015	0.0014
2	616	0.3248	0.3153	-0.0043	0.0029	0.0374	0.0368	0.0022	0.0022
3	2417	0.3848	0.3921	0.0036	0.0168	0.0373	0.0373	0.0023	0.0023
4	2448	0.4539	0.4495	0.0018	0.0018	0.0455	0.0455	0.0035	0.0035
5	509	0.3751	0.4008	0.0051	0.0027	0.0508	0.0497	0.0044	0.0042
6	1864	0.4221	0.5143	0.0110	0.0014	0.0594	0.0551	0.0058	0.0049
All	7951	0.5222	0.5518	0.0044	0.0018	0.0458	0.0447	0.0036	0.0034

variability in the data. Another explanation is that the relationship between BAI and the independent variables is linear. In this situation, a neural network can not outperform a linear model.

CONCLUSIONS

Neither neural networks nor the multiple linear regression significantly outperformed the other technique. Neural networks followed the same trend as multiple linear regression. All of the statistics indicate significant improvement by using the response variable BAI instead of DI. The addition of the variable BAL2 also improved slightly both models as indicated by the statistics. King and Arner (1998) found that the addition or substitution of other variables had little impact on the model.

There is contradictory information in books and journals on neural networks. There is no consensus on the selection of the initial weights, model selection, regularization, and scaling of input and output variables. An effort has been made to try new architectures on a subset of the 24 models as they become available in SAS.

Table 2—Comparison statistics for neural networks and multiple linear regression for NEFIA variables + BAL2

Sub-group	No. of trees	NN	REG	NN	REG	NN	REG	NN	REG
		----------R^2----------		----------ME----------		--------MAE--------		---------MSE---------	
		DI response variable and model data set							
1	257	0.0702	0.0671	-0.0003	0.0000	0.0323	0.0324	0.0016	0.0016
2	947	0.1476	0.1200	0.0002	0.0000	0.0397	0.0405	0.0026	0.0026
3	2489	0.1933	0.1925	0.0044	0.0000	0.0419	0.0425	0.0030	0.0030
4	2371	0.2649	0.2311	0.0011	0.0000	0.0529	0.0548	0.0469	0.0049
5	783	0.2696	0.2628	-0.0026	0.0000	0.0553	0.0550	0.0048	0.0049
6	1876	0.3198	0.2882	0.0017	0.0000	0.0628	0.0652	0.0067	0.0070
All	8723	0.4097	0.3907	0.0017	0.0000	0.0501	0.0513	0.0043	0.0045
		DI response variable and validation data set							
1	97	0.0357	0.0515	0.0012	0.0014	0.0334	0.0330	0.0016	0.0016
2	616	0.1355	0.1231	0.0030	0.0010	0.0412	0.0413	0.0028	0.0028
3	2417	0.2076	0.2186	0.0092	-0.0001	0.0436	0.0426	0.0030	0.0030
4	2448	0.2241	0.1964	0.0015	0.0009	0.0547	0.0552	0.0049	0.0051
5	509	0.1578	0.1590	0.0064	0.0023	0.0594	0.0585	0.0059	0.0059
6	1864	0.2895	0.2672	0.0063	-0.0008	0.0664	0.0671	0.0072	0.0074
All	7951	0.3739	0.3612	0.0054	0.0003	0.0531	0.0530	0.0047	0.0048
		BAI response variable and model data set							
1	257	0.2426	0.2549	-0.0037	0.0012	0.0300	0.0287	0.0013	0.0013
2	947	0.3350	0.3391	0.0020	0.0005	0.0351	0.0352	0.0020	0.0020
3	2489	0.4005	0.3876	-0.0003	0.0013	0.0366	0.0369	0.0022	0.0023
4	2371	0.4811	0.4782	0.0006	0.0012	0.0451	0.0452	0.0033	0.0033
5	783	0.4895	0.4914	0.0005	0.0023	0.0457	0.0455	0.0034	0.0034
6	1876	0.5282	0.5355	-0.0120	0.0027	0.0548	0.0528	0.0046	0.0046
All	8723	0.5782	0.5782	-0.0023	0.0016	0.0433	0.0429	0.0031	0.0031
		BAI response variable and validation data set							
1	97	0.1623	0.2342	0.0093	0.0018	0.0296	0.0296	0.0014	0.0013
2	616	0.3599	0.3449	-0.0012	0.0016	0.0357	0.0359	0.0021	0.0021
3	2417	0.3848	0.4048	0.0036	0.0018	0.0373	0.0372	0.0023	0.0023
4	2448	0.4658	0.4619	0.0003	0.0013	0.0454	0.0454	0.0034	0.0034
5	509	0.4098	0.4161	0.0033	0.0019	0.0495	0.0494	0.0041	0.0041
6	1864	0.4655	0.5294	0.0171	0.0026	0.0566	0.0546	0.0054	0.0048
All	7951	0.5416	0.5634	0.0054	0.0019	0.0449	0.0444	0.0035	0.0033

This work represents another step in evaluating the power of neural networks. Finding a ‘good’ set of initial weights can be a time consuming process, and some thought must be given to the frequency of use and the required precision of the final model before abandoning traditional techniques.

Table 3—Comparison statistics for neural networks and multiple linear regression for DEFIA + BAL2 variables using an updated version of SAS macros

Sub-group	No. of trees	NN	REG	NN	REG	NN	REG	NN	REG
		----------R^2----------		----------ME----------		--------MAE--------		---------MSE---------	
BAI response variable and response data set									
1	257	0.4828	0.2549	0.0010	0.0012	0.0229	0.0287	0.0009	0.0013
2	947	0.4104	0.3391	0.0022	0.0005	0.0330	0.0352	0.0018	0.0020
3	2489	0.4022	0.3876	0.0013	0.0013	0.0363	0.0369	0.0022	0.0023
4	2371	0.5030	0.4782	0.0018	0.0012	0.0436	0.0452	0.0032	0.0033
5	783	0.4932	0.4914	-0.0005	0.0023	0.0459	0.0455	0.0034	0.0034
6	1876	0.5634	0.5355	-0.0012	0.0027	0.0513	0.0528	0.0043	0.0046
All	8723	0.5991	0.5782	0.0008	0.0016	0.0416	0.0429	0.0029	0.0031
BAI response variable and validation data set									
1	97	0.1964	0.2342	0.0094	0.0018	0.0278	0.0296	0.0014	0.0013
2	616	0.3665	0.3449	0.0030	0.0016	0.0349	0.0359	0.0021	0.0021
3	2417	0.3813	0.4048	0.0013	0.0018	0.0378	0.0372	0.0024	0.0023
4	2448	0.4659	0.4619	-0.0013	0.0013	0.0451	0.0454	0.0034	0.0034
5	509	0.4121	0.4161	0.0007	0.0019	0.0495	0.0494	0.0041	0.0041
6	1864	0.4619	0.5294	0.0247	0.0026	0.0555	0.0546	0.0054	0.0048
All	7951	0.5403	0.5634	0.0062	0.0019	0.0446	0.0444	0.0035	0.0033

REFERENCES

Bazaraa, Mokhtar S.; Sherali, Hanif D.; Shetty, C.M. 1993. Nonlinear programming. New York: John Wiley. 638 p.

Bishop, Christopher M. 1995. Neural networks for pattern recognition. New York: Oxford University Press. 482 p.

Burke, Laura Ignizio. 1991. Introduction to artificial neural systems for pattern recognition. Computers and Operations Research. 18(2): 211-220.

Desai, Vijay S.; Bharati, Rakesh. 1998. Annals of Operations Research. 78(1998): 127-163.

Hassoun, Mohamad H. 1995. Fundamentals of artificial neural networks. Camdridge, MA: The MIT Press. 511 p.

Hornik, Kurt; Stinchcombe, Maxwell; White, Halbert. 1989. Multilayer feedforward networks are universal approximators. Neural Networks. 2: 359-366.

Kalman, Barry L.; Kwasny, Stan C. 1992. Why tanh: choosing a sigmoidal function. Baltimore, MD: International Joint Conference on Neural Networks: 578-581.

King, Susan L.; Arner, Stanford L. [In press]. Estimating previous diameter for ingrowth trees on remeasured horizontal point samples. Proceedings 12th central hardwood forest conference; 1999 February 28 - March 3; Lexington, KY.

Kollias, Stefanos; Anastassiou, Dimitris. 1988. Adaptive training of multilayer neural networks using a least square estimation technique. In: Proceedings of IEEE international conference on neural networks I: 383-390.

Markham, Ina S.; Rakes, Terry R. 1998. The effect of sample size and variability of data on the comparative performance of artificial neural networks and regression. Computers and Operations Research. 25(4): 251-263.

Masters, Timothy. 1993. Pratical neural networks recipes in C++. San Diego: Academic Press. 493 p.

Masters, Timothy. 1995. Advanced algorithms for neural networks: a C++ sourcebook. New York: John Wiley. 431 p.

Owens, A.J.; Filkin, D.L. 1989. Efficient training of the back propagation network by solving a system of stiff ordinary differential equations. Washington, DC: International Joint Conference on Neural Networks II: 381-386.

Piovoso, Michael J.; Owens, Aaron J. 1991. Sensor data analysis using neural networks. San Padre Island, TX. Proceedings of the 4th international conference on chemical process control: 101-118.

Reed, Russell; Pruning Algorithms-A Survey. 1993. IEEE Transactions on Neural Networks. 4(5): 740-747.

Sarle, W.S. 1994. Neural network implementation in SAS software. Proceedings of the 19th annual SAS users group international conference. Cary, NC: SAS Institute Inc.: 1551-1573.

SAS Institute Inc. 1989. SAS/IML® Software: usage and reference. Version 6, 1st ed. Cary, NC: SAS Institute Inc. 501 p.

SAS Institute Inc. 1997. SAS/OR technical report: the NLP procedure. Cary NC: SAS Institute Inc. 146 p.

Wildlife

AUTUMN ROOSTING HABITAT OF MALE INDIANA BATS (*MYOTIS SODALIS*) IN A MANAGED FOREST SETTING IN KENTUCKY

John R. MacGregor[1], **James D. Kiser, Mark W. Gumbert, and Timothy O. Reed**

Abstract—In early October 1996 and in late August and late September 1997, during the autumn pre-hibernation swarming period, a total of 22 male Indiana bats (*Myotis sodalis*) were captured by the use of a portable harp trap at the entrance to a cave hibernaculum located on the Daniel Boone National Forest in Pulaski County, Kentucky. Each bat was fitted in the field with a lightweight (0.52 g) transmitter (Holohil Systems Ltd, model LB-2) attached between the scapulae with surgical adhesive (Skin-bond), released at the point of capture within 4 hours after being caught, and subsequently tracked daily to roost trees by the use of a three-element Yagi antenna and receiver (Wildlife Materials, Model TRX-1000), during daylight hours, until its transmitter failed or it entered the hibernaculum for the winter. Each roost tree that was used by a transmittered Indiana bat was marked in the field with paint and/or plastic flagging and its location was plotted on a topographic map. Information that was collected for each tree included tree species, condition (live/dead), diameter at breast height (dbh), and (if available) the past management of the stand in which the tree was located. During 1997, percent canopy closure was measured at each roost tree by means of a concave spherical densiometer (Robert E Lemmon Forest Densiometers, Model-C). Habitat use in response to past management was evaluated by: (a) defining the analysis area to include all land (5740 ha) located within the smallest circle that could be drawn, with the bat hibernaculum as the center, that would include all known roost trees (N = 102); (b) determining the relative proportions of each type of managed habitat available to the bats within the analysis area; and (c) comparing habitat use versus availability with respect to past management practices that had taken place within each stand where known roost trees were located. The analysis area as defined above was virtually 100 percent forested, with various oak-pine, oak, yellow pine, and pine-oak forest types on the ridgetops and upper slopes and cove hardwood forest types on the lower slopes and in the stream valleys. Nearly 74 percent (4001 ha) of the analysis area was in public ownership (U. S. Forest Service); the remaining 26 percent (1445 ha) was privately owned. An impounded section of the Cumberland River, forming a 294 ha strip which extended through the northern portion of the circle, was excluded from the analysis.

During both years combined, roosting Indiana bats were located a total of 212 times in 102 different roost trees. Dead trees (snags) accounted for 86 percent (N = 88) of all roost trees found and 92 percent (N = 194) of all bat days (each day that a bat was tracked to a roost tree was defined as 1 bat day regardless of whether or not a bat had previously been found roosting in that particular tree); live trees accounted for 14 percent of all roost trees (N = 14) and 8 percent of all bat days (N = 18). Although Indiana bats were found roosting in 13 species of trees, the majority of these (79 percent) were pines (*Pinus* sp.) and oaks (*Quercus* spp.). Pine snags made up 43 percent of all roost trees (N = 44) and nearly 50 percent of all bat days (N = 105); no bats were found roosting in live pines. Oak snags (N = 25) and living oaks (N = 12) comprised 36 percent of the roost tree sample and 27 percent (N = 57) of all bat days. The most frequently used roost tree species were shortleaf pine, *Pinus echinata* (33 roost trees, 83 bat days), Virginia pine (*P. virginiana*) (11 roost trees, 21 bat days), scarlet oak, *Quercus coccinea* (15 roost trees, 19 bat days), and white oak, *Q. alba* (12 roost trees, 16 bat days).

Roost trees used by male Indiana bats during this study ranged in size from 8.4 cm to 86.6 cm (mean = 30.8 cm) dbh. Although no evening emergence counts were conducted at the individual trees, 3 different snags (2 shortleaf pines, 1 hardwood) were used simultaneously by 2 transmittered bats on a total of 8 different days, and 1 shortleaf pine snag was used simultaneously by 3 transmittered bats on 2 different days a week apart. Another shortleaf pine snag was used by different transmittered bats during successive years. These observations indicate that certain trees may be locally important as autumn roosting sites to Indiana bats that hibernate in nearby caves.

Roost tree switching was frequent for most individual Indiana bats that were monitored. In 1996, 10 bats used 1-8 different roost trees each during the 1-18 days that they were found, switching roosts a total of 46 times (including some returns to previously used trees) at an average of once every 2.0 days. In 1997, 12 bats used 2-11 roost trees each during 4-15 days of tracking, changing trees a total of 75 times for an average of once every 1.6 days. Several switched roost trees virtually every day, while others returned repeatedly to 2 or 3 particular trees. Although frequent roost tree switching was normal for most bats, in many cases all of the trees used by any individual Indiana bat during its tracking period were relatively close to one another. For the 20 bats that were found 4 times or more during both years combined, distances between roost

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trees ranged from 48 m to 2688 m, and the area of the smallest circle that could be drawn to include all roost trees used by an individual bat ranged from 0.4 ha to 568 ha. The most sedentary Indiana bats (N = 6) used 2-7 different roost trees each within a total area of 4.5 ha or less; an additional set of bats (N = 11) used 3-11 different roost trees each within a total area of 10 ha to 105 ha; and the most nomadic bats (N = 3) used 3-5 roost trees scattered over areas ranging from 518 ha to 568 ha. There did not appear to be any direct relationship between the number of times that a bat was found during a tracking period and the total area within which all of its documented roost trees were located.

As measured from the ground, canopy closure (cc) at all roost trees (N = 70) used by transmittered Indiana bats during the 1997 fall season ranged from 20 percent to 93 percent (mean = 80 percent), with 19 trees (34 bat days) located in fairly open canopy forest (<60 percent cc), 17 trees (27 bat days) in an intermediate canopy range (60-80 percent cc), and 34 trees (61 bat days) in closed canopy situations (>80 percent cc). Since most of the bats were roosting beneath loose bark in the upper portions of snags, however, there was no reasonable method available which would allow canopy closure to be measured at the actual roosts. An attempt was made to remedy this situation by using only the most open measurement that was made at each roost tree. This resulted in a canopy closure range from 0 percent (for roost trees located at the edges of large openings) to 92 percent, with 44 bat days spent in open canopy roosts (<60 percent cc), 35 bat days in intermediate canopy roosts (60-80 percent cc), and 43 bat days in closed canopy roosts (>80 percent cc).

Two separate tracts within the study area had been managed under a prescribed burning regime designed to control hardwood regeneration and maintain an open forest dominated by shortleaf pine - conditions geared toward the restoration of habitat for the federal endangered red-cockaded woodpecker (*Picoides borealis*). These tracts totaled 8 percent (436 ha) of the total amount of habitat within the analysis area, and harbored 12 of the 102 Indiana bat roost trees that were documented during both years combined. Roost tree use in prescribed burns (6 of 33 roost trees) was double the expected level (based upon the total amount of that habitat type available) during the 1996 tracking period, and equal to the expected level (6 of 70 roost trees) during the 1997 tracking period.

Although 26 percent (1445 ha) of the potential Indiana bat habitat within the study area was under private ownership, nearly all (100 of 102) of the roost trees documented were on National Forest System lands, some of which included stands where some form of timber management had taken place in recent years. During the 1996 and 1997 fall telemetry periods, approximately 17 percent of the study area (897 ha) that had been clearcut during the past 35 years yielded 0 roost trees and 0 bat days during both years combined; this was much lower than the expected level of use (16 roost trees, 35 bat days) based upon the total amount of that habitat available. Forested habitat which had not been actively managed during the past 50 years made up 44 percent of the study area (2367 ha) in 1996 and harbored 28 roost trees with 82 bat days of use as compared to an expected 15 roost trees and 39 bat days; this habitat was thus used at about twice the expected level based on its availability. This habitat type covered about 42 percent (2299 ha) of the study area in 1997, yielding 47 roost trees and 76 bat days, about 1.5 times the expected 30 roost trees and 51 bat days. Two-age shelterwood cuts (harvested during the past 5 years) and high-graded stands (up to 10 years old) comprised about 2.6 percent of the study area (143 ha) in 1996 and held 5 Indiana bat roost trees with 8 bat days of use, 4-5 times the expected level based on availability. In 1997, additional 2-age shelterwood cutting had increased the proportion of this habitat type to nearly 4 percent of the study area (211 ha), with 18 roost trees and 36 bat days documented here (6-7 times the expected levels). During both years combined, uninventoried habitats (including privately-held tracts and portions of a designated Forest Service wilderness) made up about 37 percent of the study area (2039 ha) and harbored 5 documented roost trees (of 38 expected) and 10 bat days (of 80 expected).

Although the total proportion of the study area that had been recently managed by the use of the 2-age shelterwood harvest method was too small to allow for a statistical verification of these preliminary results (i.e. that Indiana bats may actually be selecting this habitat type for roosting), some additional observations made during this telemetry study appear appropriate for this presentation. Stands that were harvested by the 2-age shelterwood method from 1993-1995, under Daniel Boone National Forest guidelines that called for the retention of 40 live trees and 5 snags/ha (16 live trees and 2 snags/acre), yielded 1 documented Indiana bat roost tree and 1 bat day during both years combined - slightly below expected levels of use. Stands that were harvested by the 2-age shelterwood method from 1996-1997, however, under different guidelines that called for the retention of 40 live trees/ha and additionally all snags, shagbark hickories, hollow trees, and trees with large dead limbs, harbored 15 Indiana bat roost trees and 27 bat days - well above expected levels of use. These observations and results suggest that timber harvesting by the 2-age shelterwood method, in concert with the retention of good numbers of snags and other suitable types of roost trees, can provide favorable roosting conditions for male Indiana bats during the autumn pre-hibernation period, at least over the short term.

FORAGING BEHAVIOR AND HABITAT USE OF RED BATS IN MIXED MESOPHYTIC FORESTS OF THE CUMBERLAND PLATEAU, KENTUCKY

Jeffrey T. Hutchinson and Michael J. Lacki[1]

Abstract—Although the red bat (*Lasiurus borealis*) is a common forest-dwelling bat, limited information is available on the foraging requirements and habitat use of this species. We radiotracked red bats on the Cumberland Plateau in eastern Kentucky during July and August 1996, and May, July and August 1997. We estimated size of foraging areas and evaluated use of habitats available within foraging areas. We placed transmitters on 10 females and nine males. Size of foraging areas ranged from 113 to 850 hectares for females, and 134 to 925 hectares for males; however, most radiotagged males moved considerable distances among nights and changed foraging areas frequently. Size of foraging areas and commuting distance varied temporally, with bats exhibiting slightly larger foraging areas and longer commuting distances in late summer compared to early summer. The maximum distance bats were recorded foraging away from the day roost ranged from 1.2 to 5.5 kilometers for females, and 1.4 to 7.4 kilometers for males. Red bats were located in non-forested habitats and aquatic habitats at proportions slightly higher than the availability of these habitats. The red bat appears to be a generalist species that can tolerate a range of habitat conditions, foraging temporally and spatially over many habitats.

INTRODUCTION

Although bats represent a large percentage of the mammals in deciduous forests (Barbour and Davis 1969) and account for 25 percent of all mammal species (Altringham 1996), the importance of bats as part of forest ecosystems in North America only recently has received much attention (Barclay and Brigham 1996). Studies on the ecology of bats in forests in western North America have been completed (Brigham and others 1997, Grindal and Brigham 1998, Kalcounis and Brigham 1998, Ormsbee and McComb 1998, Rabe and others 1998), but data for only a few species of bats in eastern North America exist (Krusic and others 1996). Most studies emphasize threatened and endangered species, particularly bats that roost in caves during all or part of the year (e.g., Adam and others 1994, Humphrey and others 1977, Tuttle 1979).

Despite the general longevity (Paradiso and Greenhall 1967) and early sexual maturity (Tuttle and Stevenson 1982) of insectivorous bats in North America, most species are monestrous and raise only one young per year (Hill and Smith 1984). An exception to this general rule are the tree-dwelling bats of the genus *Lasiurus* that have litters of usually 3-4 young (Barbour and Davis 1969, Constantine 1966, Mumford 1973). The alteration of significant amounts of forest habitat could impact populations of tree-dwelling bats to where bats presumed to be common, such as the red bat (*Lasiurus borealis*), become imperiled. Red bats are known to roost and forage in habitats such as open fields or urban areas (Constantine 1966, Mumford 1973); however, research has not addressed foraging requirements of red bats in areas where large, contiguous tracts of forest exist. The objectives of this study were to determine the location, size, and habitat of foraging areas used by red bats.

METHODS

Description of Study Areas

Research took place in the Cumberland Plateau physiographic province in eastern Kentucky. The region covers ca. 28,500 square kilometers consisting of rugged, forested terrain, locally interspersed with sandstone cliffs (McGrain 1983). Research was conducted in the northeastern portion of the Cumberland province in Carter and Elliott counties (NCP), and in the southeastern corner of the province in Breathitt, Knott, and Perry counties (SCP). Forests in both areas are classified as mixed mesophytic forest (Braun 1950). Research sites represent mature, second-growth forest that largely was undisturbed by silvicultural activities and contained mature stands of timber (i.e., an average d.b.h. ≥ 25.4 centimeters; Personal communication. 1996. Paul Kalisz, Professor of Silviculture, University of Kentucky, Lexington, KY 40546-0073).

In NCP large tracts of unbroken mature, second-growth forest, 400 to 3,200 meters wide and 8 to 20 kilometers long, were present along tributaries that run into the Little Sandy River or Grayson Lake. Grayson Wildlife Management Area, adjacent to Grayson Lake, consisted of 4,190 hectares and supported a large contiguous tract of mature, second-growth forest. Upland areas of fragmented forest, agricultural lands, pastures, and residential areas surrounded the forested-tributaries and Grayson Wildlife Management Area. Dominant forest vegetation in the area consisted of yellow-poplar (*Liriodendron tulipifera*), black walnut (*Juglans nigra*), and white oak (*Quercus alba*) on the north and east slopes (Weisenberger and others 1965). Black oak (*Q. velutina*) scarlet oak (*Q. coccinea*), and hickories (*Carya* spp.) dominated the south and west slopes, and chestnut oak (*Q. prinus*), scarlet oak, and patches of shortleaf (*Pinus echinata*) and pitch (*P. rigida*) pine were common on the upper slopes and ridges

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(Weisenberger and others 1965). Mesic and riparian vegetation of the lower slopes and creek bottoms was dominated by American beech (*Fagus grandifolia*), eastern hemlock (*Tsuga canadensis*), American sycamore (*Platanus occidentalis*), sweet birch (*Betula lenta*), and thickets of rhododendron (*Rhododendron maximum*).

Robinson Forest, located in the southeastern portion of the Cumberland Plateau physiographic province (SCP), is a 5,984-hectare tract of mature, second-growth forest 70 to 80 years old, with occasional wildlife clearings 1 to 2 hectares in size (Overstreet 1984). The area surrounding Robinson Forest was almost entirely surface-mined land; thus, Robinson Forest, served as an island of forest enclosed within a disturbed landscape. Dominant tree species included yellow-poplar, scarlet oak, white oak, black oak, American beech, chestnut oak, northern red oak (*Quercus rubra*), mockernut hickory (*Carya tomentosa*), eastern hemlock, American sycamore, and pitch pine (Overstreet 1984). Upland and ridge tops were dominated by oaks, hickories, and pines. Mesic and riparian vegetation consisted primarily of American beech, American sycamore, eastern hemlock, and white oak.

Capture and Handling Procedures

Mist nets, 5.5 or 9.1 meters wide, were placed across streams, trails, road-rut ponds, and small upland ponds to capture red bats. Bats were removed from the net, their age, sex and reproductive condition was determined, and they were weighed to the nearest 0.5 gram. Age was determined based on the time of year and the closure of the cartilaginous, epiphyseal growth plates in the finger bones (Anthony 1988).

Radiotransmitters (Type LB-2, 172-173 megahertz, Holohil Syst., Ontario, Canada) weighing 0.51 gram, with whip antennae 17.5 centimeters in length, were placed on 19 red bats (10 females and nine males). Radiotransmitters were attached using surgical cement between the scapulae on the upper back of the bat. A small section of hair was clipped between the scapulae to ensure proper bonding between the transmitter and the skin, and to lengthen the period that the transmiter remained on the bat. Each bat was held with the transmitter in place for ca. 5 minutes until the cement hardened before being released. Radiotagged bats were monitored continuously for 30 minutes following release to insure that each bat was able to fly properly and to identify any complications.

Radiotelemetry Procedures

Red bats were tracked in the NCP in July and August in 1996 and 1997, and in the SCP in May and August 1997. Telemetry stations were established at various high points in close proximity, i.e., 400 to 1,000 meters, to diurnal roost sites. Field personnel, equipped with a TRX 1000s receiver (Wildlife Materials, Inc., Carbondale, IL), a 3-element yagi antenna, and a compass, maintained contact using two-way radios. Once field personnel located a signal, azimuths were obtained simultaneously from stations at 2- to 3-minute intervals until either the bat perched in a night roost or the signal was lost. Azimuths were taken simultaneously from two or three stations, depending on the location of the bat and the movement of personnel to alternate stations to improve signal reception. Azimuths were plotted immediately at a base station on laminated 7.5-minute topographic maps to determine the location of a bat.

Data Analyses

A minimum of 40 locational crosses was used to calculate a foraging area estimate of a bat, as size of foraging areas of red bats was not found to increase beyond this sample size (Hutchinson 1998). The Calhome Software package (Kie and others 1996) was used to analyze spatial resolution on bat foraging locations, with estimates of foraging area size calculated using the adaptive kernel method. We used the adaptive kernel method because it is a nonparametric technique suitable for smaller sample sizes, does not require the assumption of normality, and is robust to changes in spatial resolution (Hansteen and others 1997). The adaptive kernel method calculates the size of a foraging area based on the intensities of locational crosses within areas, and is less biased by outliers than other existing methods (Worton 1989).

Habitat use and availability were determined by placing foraging area polygons and locational crosses over 7.5-minute topographic maps. Habitats were grouped into the following categories: forested habitat, non-forested habitat, such as clearings, agricultural land and human development, and aquatic habitat, including streams, ponds and lakes. The percentage of each habitat available and the number of radiolocations in each habitat was determined within the foraging area of each red bat for which ≥ 40 locational crosses were obtained.

Statistical summaries are based on the mean and standard error of the mean. All statistical tests were made using the Mann-Whitney U test (Hollander and Wolfe 1973).

RESULTS

The mean body mass of radiotagged adults was 12.9 ± 0.59 (SE) grams, while juveniles averaged 9.83 ± 0.60 (SE) grams (table 1). Body mass of adult females (14.1 ± 0.68 (SE) grams) was significantly greater ($U = 2.0$; $P = 0.006$) than for adult males (10.9 ± 0.40 (SE) grams). Three radiotagged females were pregnant, and two other adult females were in various stages of lactation. Female #606 roosted with two non-volant young, and female #745 roosted with two volant young. The post-lactating female (#745) was observed on several occasions leaving the roost shortly after her young emerged. Four males had descended testes, and the remaining bats showed no sign of reproductive activity.

No radiotagged adult red bats appeared to exhibit difficulty with flight after release. Commonly, radiotagged bats flew a short distance (ca. 50 to 100 meters) from the processing station to roost, and remained for 5 to 20 minutes. The bats probably were adjusting to the handling process and the added burden of the transmitter. After the initial adjustment period, the bats exhibited normal foraging habits and often foraged until we left the study area for the night, usually around 0100 hours. Twelve bats were radiotracked for a

Table 1—Sex, age, body mass, reproductive status, and tracking success of radiotagged red bats

Bat no.	Sex	Age	Mass	Status	Date of attachment	No. locations
			g			
389	F	A	14.5	Not active	7/09/97	120
430	F	A	13.5	Not active	7/21/97	46
432	M	A	11.0	Testes	7/25/97	59
451	M	J	11.0	Not active	8/05/97	0
504	F	A	15.0	Not active	8/11/97	24
521	M	A	11.0	Not active	8/11/97	0
546	F	A	11.0	Not active	8/12/97	74
575	F	A	11.5	Not active	8/12/97	54
606	F	A	14.5	Lactating	7/11/96	123
622[a]	M	J	9.5	Not active	7/24/96	0
645[a]	M	J	9.0	Not active	7/24/96	0
622[a]	M	A	12.5	Testes	7/30/96	0
645[a]	M	A	10.5	Testes	8/06/96	49
745	F	A	12.5	Post-lactating	8/09/96	40
845	M	A	11.0	Testes	8/09/96	0
883	M	A	9.5	Not active	5/20/97	41
944	F	A	14.0	Pregnant	5/20/97	66
963	F	A	16.5	Pregnant	5/20/97	53
984	F	A	18.0	Pregnant	5/20/97	46

[a] Indicates transmitter used on > 1 red bat.

sufficient period (Mean = 7.75 nights) to obtain an estimate of foraging area size.

Eighty-two percent of the azimuths resulted in locational crosses. Adult females, particularly those that were pregnant, lactating or post-lactating, were easier to monitor than adult males and often used the same foraging area for five to 12 nights (table 2). If a bat remained in the area, we usually observed little variation in the location of crosses obtained after the first three to four nights. Female bats that were not reproductively active exhibited foraging patterns more typical of adult males. All red bats were difficult to monitor in August.

The shape of foraging areas typically was bivariate and there was substantial overlap in the foraging areas (fig. 1). The foraging areas of bats contained their diurnal roost sites in all but one instance (#645) and most foraging areas included a permanent water source. Two bats (#606, #645) had foraging areas that were transected by secondary or major roadways and the foraging areas of all bats contained either forested trails or seldom used gravel or paved roads. Bats #645 and #430 each had two distinct foraging areas, separated by distances of 4 kilometers and ca. 75 meters, respectively.

The overall size of foraging areas of red bats, pooling females and males, was 334 ± 82.1 (SE) hectares (table 2). No difference (U = 11.0; P = 0.29) was observed in the mean size of foraging areas for bats in the NCP and the SCP. Males (Mean = 450 ± 242 (SE) hectares) used a foraging area almost 1.5 x larger than females (Mean = 295 ± 82.1 (SE) hectares), but this difference was not significant (U = 9.0; P = 0.46).

Reproductively active females had a mean foraging area size of 176 ± 28.0 (SE) hectares compared with a mean foraging area size for non-reproductive females of 444 ± 161 (SE) hectares, but the difference was not significant (U = 5.0; P = 0.20). During late summer, foraging areas of females appeared to increase (table 2). Although females remained in the proximity of their diurnal roost sites in late summer, they occasionally made forays out of the range of receivers. These females, all non-reproductive, usually returned within 45 minutes.

The maximum distance that bats were recorded foraging away from the day roost ranged from 1.2 to 5.5 kilometers for females, and 1.4 to 7.4 kilometers for males (table 2). Bats traveled somewhat further to foraging sites in the NCP than the SCP. When the distance traveled by red bats was examined temporally, bats monitored in late summer (August) were detected an average of 5.02 kilometers from the roost, whereas remaining bats traveled an average of 1.55 kilometers from the roost.

The foraging area polygons of red bats consisted of 87.2 percent forested habitat, 11.8 percent non-forested areas,

Table 2—Estimates of foraging area size and maximum distances traveled from day roosts of radiotagged red bats[a]

Bat no.	Study area	Days monitored	Foraging area	Maximum distance
			Ha	*Km*
389	NCP	12	193	1.4
430	NCP	2	179	1.3
432	NCP	4	291	2.7
606	NCP	12	262	1.7
645	NCP	8	925	7.4
745	NCP	7	120	5.5
546	SCP	6	554	3.3
575	SCP	9	850	3.9
883	SCP	9	134	1.4
944	SCP	10	116	1.2
963	SCP	6	113	1.2
984	SCP	8	192	1.5

[a] Includes only bats for which ≥ 40 locational crosses were obtained.

and 1.0 percent aquatic habitat (table 3). The bats were recorded foraging over forested habitat 78.3 percent of the time, non-forested areas 16.0 percent of the time, and aquatic habitat 5.7 percent of the time. In the NCP where the landscape was more fragmented, 76.8 percent of the foraging area of red bats was forested and bats spent 68.6 percent of their foraging time over those habitats. The total foraging area of red bats in the SCP was 97.5 percent forested habitats and the bats used these areas 88.0 percent of the time. Aquatic habitats accounted for ≤ 1 percent of the total habitat within all foraging areas of red bats, yet accounted for 5.0 percent and 6.5 percent of the foraging locations of bats in the NCP and SCP, respectively.

DISCUSSION

The characteristics of the landscape appeared to dictate the habitats that red bats used as foraging habitat. Although forested habitat predominated in both study areas, red bats also used other habitats such as grazing lands, agricultural fields, cemeteries, street lights and other human residential areas, especially in the more fragmented NCP. These data are consistent with those obtained for red bats in other locations where this species was frequently detected foraging in non-forested habitat (Hickey and Fenton 1990; Hickey and others 1996; McCracken and others 1997).

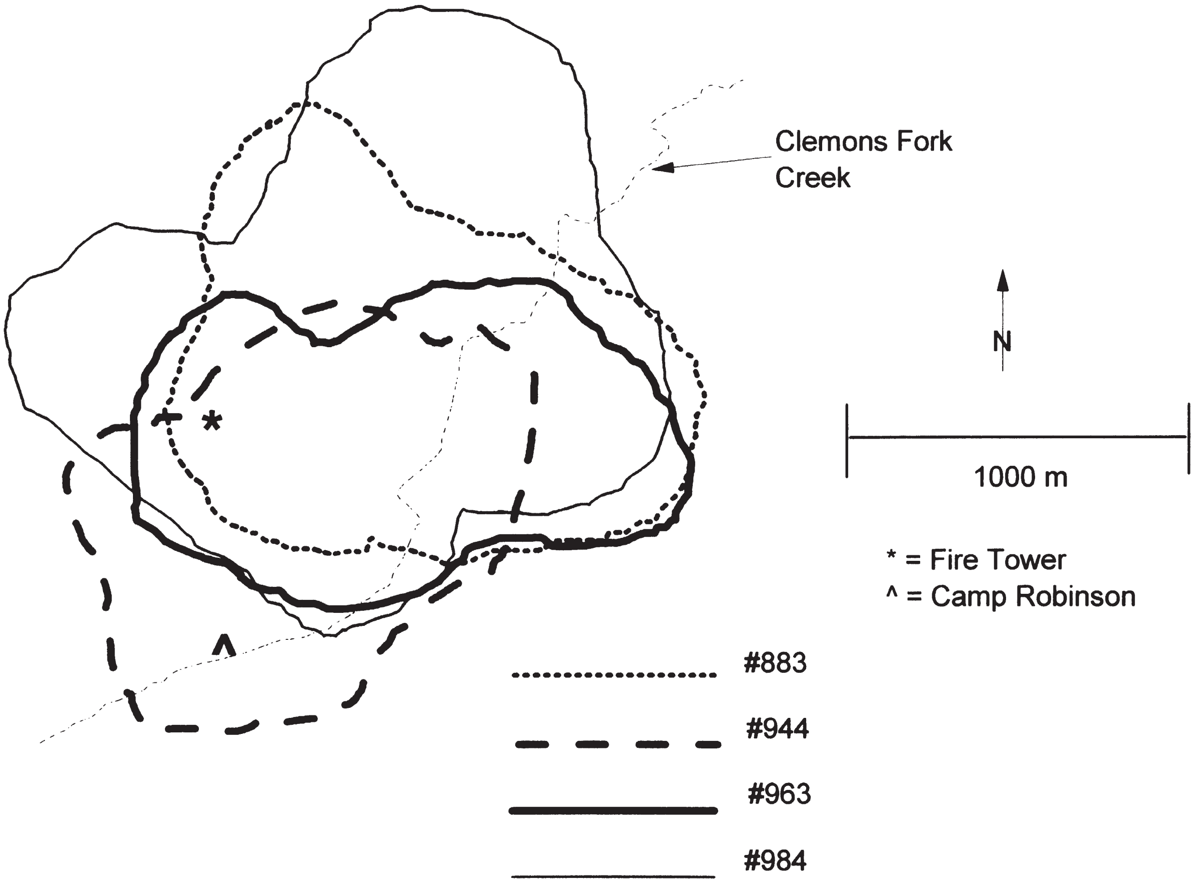

Figure 1—Foraging area polygons for four radiotagged red bats showing overlap in use of available habitat.

Table 3—Comparison of habitat available within foraging area polygons to the percentage of locational crosses by habitat (in parentheses) of radiotagged red bats in eastern Kentucky[a]

Bat no.	Study area	Forested habitat	Non-forested habitat	Aquatic[b] habitat
		Percent		
389	NCP	99.0 (95.8)	0.0 (1.7)	1.0 (2.5)
430	NCP	79.0 (56.5)	20.0 (36.9)	1.0 (6.6)
432	NCP	90.0 (67.8)	9.0 (30.5)	1.0 (1.7)
606	NCP	64.0 (57.7)	35.0 (36.6)	1.0 (5.7)
645	NCP	76.0 (71.4)	23.0 (20.4)	1.0 (8.2)
745	NCP	53.0 (62.5)	46.0 (32.5)	1.0 (5.0)
546	SCP	98.0 (92.0)	1.0 (4.0)	1.0 (4.0)
575	SCP	98.0 (87.0)	1.0 (3.7)	1.0 (9.3)
883	SCP	98.0 (92.7)	1.0 (4.9)	1.0 (2.4)
944	SCP	95.0 (75.8)	4.0 (16.7)	1.0 (7.5)
963	SCP	98.0 (86.8)	1.0 (1.9)	1.0 (11.3)
984	SCP	98.0 (93.5)	1.0 (2.2)	1.0 (4.3)

[a] Includes only bats for which ≥ 40 locational crosses were obtained.
[b] Availability of water in foraging area estimated to be ≤ 1.0 percent for all red bats monitored.

Because red bats selectively foraged over water at proportions greater than the availability of this habitat in the landscape, a source of permanent water in the vicinity of roosting sites of red bats apparently is important. However, in the NCP where large bodies of water were present, i.e., Grayson Lake and the Little Sandy River, red bats were seldom detected foraging over these sites. The majority of locational crosses of red bats over water occurred while the bats were foraging over small streams during the first 30 to 45 minutes after emergence. Mumford and Whitaker (1982) reported that red bats needed bodies of water for drinking and observed this species foraging over small pools of water ca. 1.2 x 1.8 meters in size and as shallow as 2.5 centimeters in depth. A small permanent source of water transected the foraging areas of all but two of the red bats monitored.

Despite their use of aquatic and non-forested habitats, red bats foraged in forested habitat more often than the other habitats available, but used forests at proportions less than their availability in the landscape. This was especially true for bats in the SCP where large disturbed habitats in the form of reclaimed strip mines and numerous upland ponds were nearby, but were never used as foraging areas. Instead, red bats remained in contiguous forest almost 90 percent of the time. Barclay (1984) observed similar patterns for red bats in Manitoba. He found that red bats used forested ridges 75 percent of the time they were foraging and spent much less time foraging in the other habitats.

We observed substantially larger foraging areas of red bats than previously reported (Hickey and Fenton 1990; McCracken and others 1997). In these studies, red bats centered their foraging activity around street lights and, in one study, averaged 127 minutes/night foraging around lights (Hickey and Fenton 1990). The echolocation calls of red bats foraging around street lights may be homed in on by other red bats in search of prey (Hickey and Fenton 1990). Although more than one red bat was seen occasionally foraging around street lights, we never observed two radiotagged bats using street lights at the same time despite substantial overlap in the foraging areas of red bats monitored simultaneously (fig. 1). The red bats monitored in the Cumberland Plateau physiographic region of eastern Kentucky did not exploit the prey densities around street lights in rural areas to the extent hypothesized by Furlonger and others (1987).

By recording azimuths every 2 to 3 minutes, the possibility exists that our data are affected by autocorrelation (White and Garrott 1990). Autocorrelation leads to a biased underestimate of foraging area size (Hansteen and others 1997, White and Garrott 1990). Extending the length of time between azimuths will correct for autocorrelation, but will also lead to a reduction in sample size (Hansteen and others 1997). Given the length of time that the transmitters were likely to remain active (ca. 1 to 2 weeks), we felt that sample size was a more important consideration in our study. Further, our estimates of foraging area size of red bats were larger than any reported in the literature (Hickey and Fenton 1990, McCracken and others 1997); therefore, it is unlikely that autocorrelation, if present, altered the conclusions drawn from this study.

The maximum distance red bats foraged away from their roosting sites was slightly greater for bats in the fragmented landscape of the NCP than for bats inhabiting the contiguous forests of the SCP. These distances increased in late summer, likely coinciding with the breakup of family units as suggested by Constantine (1966). Kunz (1982) proposed that roost lability, especially among tree-roosting species of bats, results in decreased commuting costs to foraging areas; however, the distances that red bats were recorded traveling from roosting sites to feed were comparable in length to those recorded for cliff-roosting bats, i.e., *Corynorhinus* species, on the Cumberland Plateau of eastern Kentucky (Adam and others 1994, Hurst 1997). Based on Kunz (1982), we anticipated shorter commuting distances for red bats than were observed for cliff-roosting species, because of a higher availability of potential roosting sites for red bats. Because red bats traveled long distances to feed and often used large foraging areas, roost lability probably was not a factor influencing commuting cost or foraging area size in the red bats that we radiotracked. Instead, we propose that commuting cost and foraging area size in red bats were a function of the seasonal cycle and the reproductive status of individual bats. The location of foraging areas ultimately may be determined by the presence of preferred diurnal roosting sites.

Because all bats in which ≥ 40 locational crosses were obtained met the 5 percent load-carrying rule for transmitter mass (Aldridge and Brigham 1988), we believe that these data accurately depict the foraging behavior of red bats. All radiotagged red bats exhibited normal foraging patterns within 20 minutes after release, with the exception of two juvenile males in July 1996. We found adult males to be much more difficult to track than adult females, with increased movements for bats radiotagged after 24 July in both years. Saugey and others [In Press] reported similar problems in radiotracking male red bats back to roosting sites in Arkansas, referring to the roosting pattern of males as “vagabond.”

CONCLUSIONS

Although there is considerable plasticity in the selection and use of foraging habitat by red bats, the extensive use of forests by this species suggests that large blocks of contiguous forest provide suitable habitat. Shorter commuting distances for red bats in the less fragmented SCP and the complete absence of radiotagged bats in the heavily disturbed habitat (i.e., surface-mined lands) nearby, indicate that some deforested areas and habitat fragments may provide less than optimal habitat conditions for this species. Studies comparing the survivorship and fecundity of red bats inhabiting forested and fragmented landscapes, including agricultural and surface-mined lands, are warranted.

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REFERENCES

Adam, M.D.; Lacki, M.J.; Barnes, T.G. 1994. Foraging areas and habitat use of the Virginia big-eared bat in Kentucky. Journal of Wildlife Management. 58: 462-469.

Aldridge, H.D.J.N.; Brigham, R.M. 1988. Load carrying and maneuverability in an insectivorous bat: a test of the 5 percent “rule” of radio-telemetry. Journal of Mammalogy. 69: 379-382.

Altringham, J.P. 1996. Bats: biology and behavior. New York: Oxford University Press. 262 p.

Anthony, E.L.P. 1988. Age determination in bats. In: Kunz, Thomas H., ed. Ecological and behavioral methods for the study of bats. Washington, DC: Smithsonian Institution Press: 47-58.

Barbour, R.W.; Davis, W.H. 1969. Bats of America. Lexington, KY: University Press Kentucky. 286 p.

Barclay, R.M.R. 1984. Observations on the migration, ecology and behavior of bats at Delta Marsh, Manitoba. Canadian Field-Naturalist. 98: 331-336.

Barclay, R.M.R.; Brigham, R.M., eds. 1996. Bats and forests symposium; 1995 October 19-21; Victoria, British Columbia, Canada. Work Pap. 23/1996. Res. Br., B.C. Min. For., Victoria, B.C.

Braun, E.L. 1950. Deciduous forest of Eastern North America. New York: Hafner. 569 p.

Brigham, R.M.; Vonhof, M.J.; Barclay, R.M.R.; Gwilliam, J.C. 1997. Roosting behavior and roost-site preferences of forest-dwelling California bats (*Myotis californicus*). Journal of Mammalogy. 78: 1231-1239.

Constantine, D.G. 1966. Ecological observations on Lasiurine bats in Iowa. Journal of Mammalogy. 47: 34-41.

Furlonger, C.L.; Dewar, H.J.; Fenton, M.B. 1987. Habitat use by foraging insectivorous bats. Canadian Journal of Zoology. 65: 284-288.

Grindal, S.D.; Brigham, R.M. 1998. Short-term effects of small-scale habitat disturbance on activity by insectivorous bats. Journal of Wildlife Management. 62: 996-1003.

Hansteen, T.L.; Andreassen, H.P.; Rolf, I.A. 1997. Effects of spatiotemporal scale on autocorrelation and home range estimators. Journal of Wildlife Management. 61: 280-290.

Hickey, M.B.C.; Acharya, L.; Pennington, S. 1996. Resource partitioning by two species of vespertilionid bats (*Lasiurus cinereus* and *Lasiurus borealis*) feeding around street lights. Journal of Mammalogy. 77: 325-334.

Hickey, M.B.C.; Fenton, M.B. 1990. Foraging by red bats (*Lasiurus borealis*): do intraspecific chases mean territoriality? Canadian Journal of Zoology. 68: 2477-2482.

Hill, J.E.; Smith, J.D. 1984. Bats: a natural history. Austin, TX: University Texas Press. 243 p.

Hollander, M.; Wolfe, D.A. 1973. Nonparametric statistical methods. New York: John Wiley. 503 p.

Humphrey, S.R.; Richter, R.A.; Cope, J.B. 1977. Summer habitat and ecology of the endangered Indiana bat, *Myotis sodalis*. Journal of Mammalogy. 58: 334-346.

Hurst, T.E. 1997. Foraging area, habitat use, population estimates and food habits of Rafinesque's big-eared bat in southeastern Kentucky. Lexington, KY: University of Kentucky. 112 p. M.S. thesis.

Hutchinson, J.T. 1998. Summer roost site selection of red bats in mixed mesophytic forests. Lexington, KY: University of Kentucky. 207 p. M.S. thesis.

Kalcounis, M.C.; Brigham, R.M. 1998. Secondary use of aspen cavities by tree-roosting big brown bats. Journal of Wildlife Management. 62: 603-611.

Kie, J.G.; Baldwin, J.A.; Evans, C.J. 1996. CALHOME: a program for estimating animal home ranges. Wildlife Society Bulletin. 24: 342-344.

Krusic, R.A.; Yamasaki, M.; Neefus, C.D.; Pekins, P.J. 1996. Bat habitat use in White Mountain National Forest. Journal of Wildlife Management. 60: 625-631.

Kunz, T.H. 1982. Roosting ecology. In: Kunz, Thomas H., ed. Ecology of bats. New York: Plenum Publishing Corp.: 1-54.

McCracken, G.F.; Hayes, J.P.; Cevallos, J. [and others]. 1997. Observations on the distribution, ecology, and behavior of bats on the Galapagos Islands. Journal of Zoology (London). 243: 757-770.

McGrain, P. 1983. The geologic story of Kentucky. Spec. Publ. 8, Series XI. Lexington, KY: Kentucky Geologic Survey. 74 p.

Mumford, R.E. 1973. Natural history of the red bat (*Lasiurus borealis*) in Indiana. Period. Biology. 75: 155-158.

Mumford, R.E.; Whitaker, J.O., Jr. 1982. Mammals of Indiana. Bloomington, IN: Indiana University Press. 537 p.

Ormsbee, P.C.; McComb, W.C. 1998. Selection of day roosts by female long-legged myotis in the central Oregon Cascade range. Journal of Wildlife Management. 62: 596-603.

Overstreet, J.C. 1984. Robinson Forest inventory, 1980-82. Lexington, KY: University of Kentucky. 52 p.

Paradiso, J.L.; Greenhall, A.M. 1967. Longevity records for American bats. American Midland Naturalist. 78: 251-252.

Rabe, M.J.; Morrell, T.E.; Green, H. [and others]. 1998. Characteristics of Ponderosa pine snag roosts used by reproductive bats in northern Arizona. Journal of Wildlife Management. 62: 612-621.

Saugey, D.A.; Vaughn, R.L.; Crump, B.G.; Heidt, G.A. [In press]. Notes on the natural history of the red bat, *Lasiurus borealis*, in Arkansas. Journal of the Arkansas Academy of Science.

Tuttle, M.D. 1979. Status, causes of decline, and management of endangered gray bats. Journal of Wildlife Management. 43: 1-17.

Tuttle, M.D.; Stevenson, D. 1982. Growth and survival of bats. In: Kunz, Thomas H., ed. Ecology of bats. New York: Plenum Publishing Corp.: 105-150.

Weisenberger, B.C.; McDonald, H.P.; Wallace, H.A. 1965. Soil survey of Elliott County, KY. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 55 p.

White, G.C.; Garrett, R.A. 1990. Analysis of wildlife radiotracking data. San Diego, CA: Academic Press, Inc. 383 p.

Worton, B.J. 1989. Kernel methods for estimating the utilization distribution in home range studies. Ecology. 70: 164-168.

WHITE-TAILED DEER IMPACT ON FOREST REGENERATION: MODELING LANDSCAPE-LEVEL DEER ACTIVITY PATTERNS

Linda S. Gribko, Michael E. Hohn, and William M. Ford[1]

Abstract—White-tailed deer (*Odocoileus virginianus*) herbivory has been identified as a major impediment to the survival and growth of forest regeneration in the northeastern United States. As a supplement to direct control of deer densities through hunting, it may be possible for land managers to manipulate habitat and browsing pressure through carefully planned timber harvest. We are developing methods to relate deer habitat use patterns to regeneration condition and complexity across large landscapes. The preliminary research presented here involved development of methodology to efficiently and effectively model deer habitat use patterns across forested landscapes using fecal pellet groups as an activity index.

Work was conducted in summer 1997 on the 3,078-hectare West Virginia University Forest (WVUF) in north-central West Virginia and the 3,413-hectare Westvaco Wildlife and Ecosystem Research Forest (WERF) in the south-central portion of the state. Fecal pellet group counts were conducted on geolocated 1.72-meter radius circular plots at an intensity of approximately 1 plot per 2.4 hectares. We used variography to investigate spatial dependence in the data. Deposition patterns were modeled using geographic information systems (GIS) technology and a spatial statistics technique known as 2-dimensional ordinary point kriging. Variography revealed spatial contagion in the WVUF data that could be accurately modeled using this methodology. The resulting interpolated probability map is of high potential value in the long-term monitoring of deer activity patterns across this forested landscape. Results on the WERF indicated that sampling intensity was too coarse to allow modeling of strong localized dependence in the data.

INTRODUCTION

Excessive white-tailed deer (*Odocoileus virginianus*) herbivory, or browsing, has been identified as a major impediment to the survival and growth of tree seedlings and herbaceous plants in the northeastern United States (Shafer and others 1961, Tierson and others 1966, Jordan 1967, Alverson and others 1988, Tilghman 1989, Trumbull and others 1989). Browsing can affect forest regeneration by reducing seedling numbers, reducing seedling height, altering species composition, delaying stand establishment, or causing complete regeneration failure (Redding 1987). Of particular concern to forest managers are conversions of woody understories to ferns and grasses (Marquis 1974, Horsely and Marquis 1983) and the elimination of highly valued commercial tree species such as northern red oak (*Quercus rubra*) from the pool of large advance regeneration under mature forest canopies (Marquis and others 1976). Severely impacted forest ecosystems are devoid of most understory plants that are palatable to deer and may exhibit an overall reduction in animal diversity (deCalesta 1994).

Deer herbivory research conducted to date suggests that deer population densities in many forested landscapes should be reduced (Behrend and others 1970, Marquis 1981, Storm and others 1989, Bowersox and others 1993, McCormick 1993). However, the public demand for deer hunting opportunity often outweighs concerns about browsing impacts; state wildlife agencies are consequently reluctant to lower deer densities to suggested levels (Sheffer 1987).

Rather than attempting to control deer densities directly through harvest, it may be possible to manage forested landscapes to reduce browsing pressure and regeneration damage. The manipulation of the spatial distribution of clearcuts has been suggested as one means of accomplishing this objective (Ford and others 1994). Deer extensively forage in recent clearcuts during spring and summer months when succulent leaves and new growth are available (Wentworth and others 1990, Ford and others 1993). In the central and southern Appalachians, the availability of clearcuts has been found to increase carrying capacity of the surrounding forest and reduce browsing pressure on forested understories during the summer months (Johnson and others 1995). Improvement of forage conditions through timber harvest has been suggested as a means of reducing relative deer densities (deCalesta and Stout 1997). Careful planning of the intensity, timing, and location of timber harvest areas may allow managers to draw deer away from forested understories while maintaining the population at a sufficient density to provide adequate hunting opportunity.

Currently, a lack of deer herbivory research conducted at the landscape level constrains the development of forest management recommendations. Most studies have considered localized, or stand-level, effects of deer browsing. Investigators have considered the impacts of deer in contiguous closed-canopy forests (Marquis 1981), in small isolated woodlots (Storm and others 1989, Bowersox and others 1993), in recently clearcut areas (Marquis and

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Grisez 1978), or after specific stand treatments such as thinnings (McCormick 1993). Typically, such research has involved comparisons of areas with varying densities of deer or the use of fencing and barriers to manipulate deer impacts. Work at larger spatial scales (200 to 400-hectare blocks) has been recently initiated in Pennsylvania (Stout and others 1995). However, the more fragmented severely browsed forests of Pennsylvania may not be comparable to the more continuous less intensively browsed forests found elsewhere in the Appalachian region.

In 1997, we initiated a study designed to relate deer habitat use patterns to woody regeneration condition and complexity across landscapes dominated by relatively continuous closed-canopy second-growth hardwood forest. This paper reports the results of the first phase of study in which we developed a method of quantifying landscape-level deer habitat use patterns using fecal pellet group counts as an activity index. Generally, pellet groups are counted on either rectangular belt transects (Bennett and others 1940, Robinette and others 1958, Fuller 1991) or circular plots of various dimensions (Eberhardt and Van Etten 1956, Robinette and others 1958, Van Etten and Bennett 1965). Counts of fecal pellet groups are widely used to census deer populations and to determine habitat use (Neff 1968). Average counts within management units or habitat types are compared as relative indices of deer abundance or are used with defecation rate to estimate population densities. Bennett and others (1940) found that deer defecate close to feeding areas, indicating that high densities of fecal pellets may be associated with high levels of herbivory.

Traditionally, spatial patterns of pellet group deposition are not considered. However, if the spatial dependency among fecal pellet counts made at a large number of discreet points throughout a forested landscape is strong enough to be modeled, the relationship could be used to estimate pellet count densities at unsampled locations. A map, or data layer, of predetermined spatial resolution would result. The field of geology offers techniques, termed geostatistics, that can be used to quantify the inherent spatial dependencies in data and use them to create interpolated data coverages. Although developed to model petroleum and mineral reserves, geostatistics have also been used in the modeling of ecological data (Robertson 1987, Rossi and others 1992, Pelletier and Parma 1994, Villard and Maurer 1996). One of these techniques, *kriging*, has been applied outside of the geological sciences to model insect outbreaks (Kemp and others 1989, Liebhold and others 1991, Hohn and others 1993, Gribko and others 1995), forest biomass and nutrient cycles (Pauly and others 1996), and old-growth forest characteristics (Biondi and others 1994).

The purpose of this study was to determine if fecal pellet group counts made at discreet points could be used to model deer activity patterns across a forested landscape. Specific objectives included: 1) quantifying the spatial dependence inherent in deer fecal group deposition, and 2) determining the suitability of geostatistical techniques, specifically 2-dimensional ordinary point kriging, in mapping white-tailed deer use of forested landscapes.

METHODS

Study Areas

The study was conducted on two sites: the 3,078-hectare West Virginia University Forest (WVUF) in Monongalia and Preston Counties, WV, and the 3,413-hectare Westvaco Wildlife and Ecosystem Research Forest (WERF) located in Randolph County, WV. The WVUF is located on the western-most anticline of the Allegheny Mountains. Elevations range from 318 to 796 meters. The tract is covered by closed-canopy 70-80 year old hardwood forest; no more than 200 hectares have been impacted by timber harvest or major canopy disturbance in the past 20 years. Four forest types as classified by SAF (Eyre 1980) have been identified on the WVUF: yellow-poplar (*Liriodendron tulipifera*)—white oak (*Quercus alba*)—northern red oak, white oak—black oak (*Quercus velutina*)—northern red oak, chestnut oak (*Quercus prinus*), and yellow-poplar. The two types containing yellow-poplar components are found in protected coves, on north and east aspects, and at shaded mid-slope positions. The oak-dominated types are found on the drier, less protected sites; the chestnut oak type is constrained primarily to ridgetops and upper slope positions.

Elevations on the WERF range from 699 to 1176 meters. The topography is dominated by a series of ridges oriented northeast to southwest. Ninety-seven percent of the forested area is comprised of 60 to 70-year-old stands of mature hardwoods. However, the majority of the mature stands have been partially harvested at least once in the past decade and canopy cover is not continuous. The WERF includes 3 forest types in addition to 140 hectares of open or non-forested land. Ninety percent of the forest cover is classified as the sugar maple (*Acer saccharum*)-beech (*Fagus grandifolia*)-yellow birch (*Betula allegheniensis*) SAF type; associated species include white ash (*Fraxinus americana)*, American basswood (*Tilia americana*), black cherry (*Prunus serrotina*), yellow-poplar, and northern red oak. The yellow-poplar—white oak—northern red oak type is found on 195 hectares of protected coves. The remaining 21 hectares are classified as the white oak SAF type; associated species include chestnut oak, scarlet oak (*Quercus coccinea*), black oak, and hickory (*Carya* spp.).

Field Methods

Data was collected in summer 1997. Sample points were established in an approximately square-grid pattern at an intensity of approximately 1 per 2.4 hectares; points were approximately 240 meters apart. A total of 1,400 points were located on the WVUF; 1,445 points were located on the WERF. The points were georeferenced using a hand-held GPS unit and fixed base station which allowed differential correction of the satellite locations. Each point was the center of a 1.72-meter (9.29 square meters) radius circular plot on which fecal pellet surveys were conducted (Robinette and others 1958, Van Etten and Bennett 1965). Pellet groups on each of the plots were counted once from June to July on the WVUF and from August to September on the WERF. All pellets were destroyed by crushing. The counts provided baseline data and were not assumed to represent deer activity during any specific season.

Data Analysis

Topographic variables—It was first necessary to examine the relationships between pellet group densities and possibly confounding variables related to topography. Digital elevation models (DEMs) were available for both experimental forests. The model for the WERF was developed by Westvaco personnel at a resolution of slightly over 15 meters. The 30-meter resolution USGS DEM was used to describe the topography of the WVUF. We calculated percent slope and aspect in degrees azimuth using the elevation data and the raster-based geographic information system (GIS), IDRISI (Eastman 1997). We then standardized the aspect data by subtracting 180 degrees and taking the absolute value of the difference; effectively lumping the data collected on approximately eastern and western aspects (intermediate site quality) while segregating data collected on approximately northern (highest site quality) and southern (lowest site quality) aspects. For each experimental forest, pellet group data were plotted against elevation, slope, and aspect and relationships were examined.

Variography—Spatial dependence on each experimental forest was quantified with the variogram, a graph that illustrates the relationship between the distance separating pairs of data points (*h*) and half the average squared deviation of a regionalized variable ($\gamma(h)$). In this case, the variogram statistic at each given distance *h* represents half the average squared difference between paired fecal pellet group counts separated by that particular distance (Hohn 1988, Isaaks and Srivastava 1989, Liebhold and others 1991, Rossi and others 1992). The variogram statistic is defined as

$$\gamma(h) = \frac{1}{2n}\sum_{k=1}^{n_h}[z(x_k) - z(x_k + h)]^2$$

where n_h is the number of pairs of cells separated by a distance of *h* (expressed in meters in this study), $z(x_k)$ is the observed pellet group count at location x_k, and $z(x_k + h)$ is the observed count at a location *h* meters from x_k. In the absence of spatial dependence, the variogram is constant with distance *h*. In the presence of strong spatial dependence, the difference between $z(x_k)$ and $z(x_k + h)$ is relatively small when *h* is small and increases with separation distance. At some distance, the difference between counts fails to increase further and the variogram levels off. This distance is referred to as the *range* of the variogram. The value of $\gamma(h)$ at the range is called the *sill*. In general, one would expect the variogram to originate at the origin; however, variograms manifest a non-zero y-intercept when some component of the variability is non-spatial or on a smaller scale than sample spacing.

For each experimental forest, VARIOWIN version 2.2 (Pannatier 1996) was used to calculate the variogram statistic in 120-meter lags with a 60-meter tolerance to a maximum separation distance of 1200 meters. The first lag included pairs having a separation distance smaller than the tolerance; in other words, all pairs separated by less than 60 meters. The few pairs in this lag were retained rather than disregarded. The relationship between pairs separated by more than 1200 meters was not considered. Surface plots of the variogram statistics calculated in the east-west direction versus those calculated in the north-south direction were examined for indications of directionality in the data (Figure 1-2). In the presence of strong directional trends, several individual models are required to accurately describe the spatial dependence in the data. In its absence, a single omnidirectional model is sufficient. In neither data set was there discernible directionality.

Omnidirectional variograms for each forest were plotted using VARIOWIN and models were fit iteritively by eye (Hohn 1988). Single and nested linear, exponential, gaussian, and spherical model forms were tested for

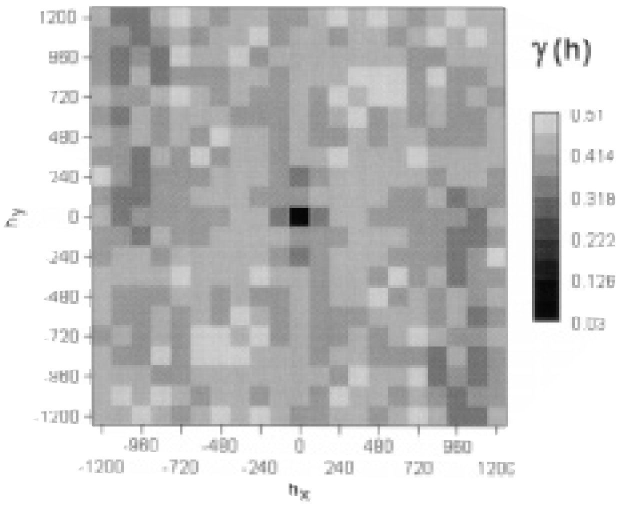

Figure 1—Variogram surface for the West Virginia University Forest (WVUF).

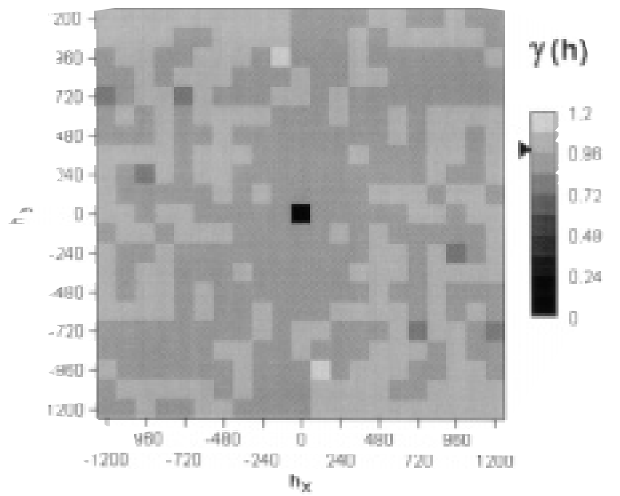

Figure 2—Variogram surface for the Westvaco Wildlife and Ecosystem Research Forest (WERF).

general fit to the shape of the variograms and the models were fine-tuned by alternately adjusting the *sill* and the *range*. Special care was taken to fit the first few lags accurately, so the final models may or may not have exhibited the best overall goodness of fit. In both cases, the origin was forced through zero.

Kriging—Results of the variogram analysis were used to calculate weights employed in the 2-dimensional ordinary point kriging of the pellet count data. Kriged estimates are weighted averages of values at nearby locations

$$z^* = \sum_{j=0}^{n} w_j \cdot z_j$$

where z^* is the interpolated value being estimated, $z = [z_1, z_2, z_3, \ldots, z_n]$ is the vector of values at nearby locations and $w = [w_1, w_2, w_3, \ldots, w_n]$ is the vector of corresponding weights to be used in averaging the values. Because the *kriging* procedure minimizes the variance of the errors, the weight matrix can be estimated as

$$w = C^{-1} \bullet D$$

where C is the (n+1) by (n+1) covariance matrix and all the z values from nearby samples and D is the n+1 vector of covariances of z values between the point being estimated and the nearby samples. The covariances can be computed directly from the variogram as

$$\mathrm{cov}(h) = c - \lambda(h)$$

where c is the variogram sill.

GSLIB version 2.0 was used to krige the data (Deutsch and Journel 1998). Kriged estimates were made at a 30-meter resolution and were based on at last 2, but no more than 36, values collected within 500 meters of each unsampled location. Estimates made near the boundaries of the experimental forests were based necessarily on fewer samples than those made in the interior.

RESULTS

There were no strong identifiable relationships between the topographic variables and pellet group counts on either of the experimental forests. Aspect may have some effect on the WERF as evidenced by somewhat depressed, but statistically insignificant, pellet group counts on the more southerly aspects. It appears that elevation possibly may be related to counts on the WVUF; however, high pellet group densities at elevations over 600 meters were clustered on a portion of the forest that was heavily impacted by overstory mortality in 1990-91. Higher counts in these areas were likely related to browse availability rather than elevation. Linear regressions through these data were all essentially horizontal with R^2 values less than 0.01. No attempt was made to transform the data.

Variogram surface plots revealed no directional dependence in the WVUF data (Figure 1). Weakly elevated dependence was suggested among pellet plots oriented in a northeast to southwest direction on the WERF (Figure 2). However, the trend was not strong enough to necessitate calculation of directional variograms. The majority of the cells in the variogram surface had values of approximately 0.72 - 0.96; had the values been appreciably lower in the northeast-southwest direction (perhaps in the range of 0.24 – 0.48), a pair of directional variograms may have improved our ability to model the data.

Omnidirectional variograms and the final fitted models for both forests are shown in Figures 3-4. The variogram calculated for the WVUF data was fit with the nested exponential model:

$$\gamma(h) = 0.015\left[1 - e^{\frac{-3|h|}{156}}\right] + 0.415\left[1 - e^{\frac{-3|h|}{231}}\right].$$

The omnidirectional variogram for the WERF was fit with the nested model:

$$\gamma(h) = 0.75\left[1.5\frac{|h|}{92} - 0.5\left(\frac{|h|}{92}\right)^3\right] + 0.20\left[1 - e^{\frac{-3|h|}{485.48}}\right] \text{ if } |h| \le 92,$$

else,

$$\gamma(h) = 0.75 + 0.20\left[1 - e^{\frac{-3|h|}{485.48}}\right].$$

The first structure in this model is spherical; the second is exponential.

Maps of the kriged estimates are displayed in Figures 5-6.

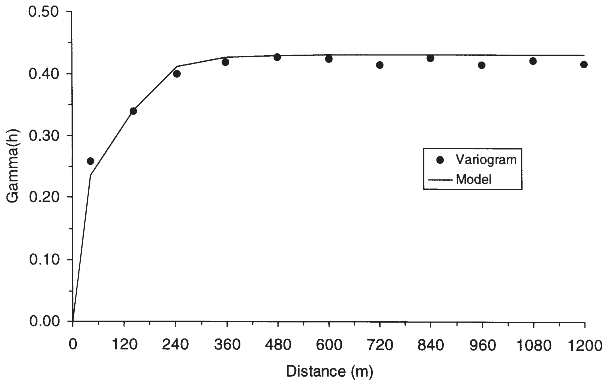

Figure 3—Variogram and fitted model for the WVUF.

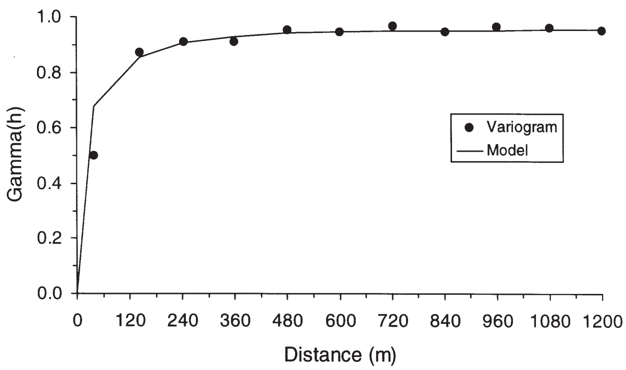

Figure 4—Variogram and fitted model for the WERF.

Figure 5—Kriged estimate map for the WVUF. Cells are 15 by 15 meters.

DISCUSSION

Absence of both strong directional trends and strong associations with topographic variables simplifies the modeling process and suggests that differences in pellet group densities are likely related to differences in habitat quality. On both study sites, a single omnidirectional model can be used to describe spatial contagion in the data. In addition, there is no apparent need to incorporate additional independent variables through the use of more complex geostatistical techniques such as cokriging or kriging with external drift (Goovaerts 1997, Deutsh and Journel 1998). This will allow us to monitor the response of deer populations to timber management activities across the study sites without the added complication of confounding variables.

Because the exact location and orientation of the sample plots were not critical in this initial exploratory study, we used temporary locations with the distance between plots paced, rather than measured. This facilitated more rapid data collection but also resulted in less uniformity in the sample design; some plots were installed in closer proximity to adjacent plots than was actually planned. These data could have been omitted; however, we chose to retain them because they provide important information about spatial dependencies in the data at smaller separation distances. The WVUF data included 132 pairs of plots that were separated by an average distance of 42.31 meters and 5,832 pairs separated by an average distance of 142.76 meters. Numbers of additional pairs ranged from 9,928 at a separation distance of 244.40 meters to 37,576 at the maximum separation distance of 1,200.19 meters. The WERF data included 78 pairs separated by an average distance of 40.25 meters and 5,426 separated by an average distance of 144.03 meters. Numbers of pairs ranged from 9,946 at 244.76 meters to 39,144 at approximately 1,200 meters.

The variogram models for both forests included two nested structures; indicating that on both sites there existed local and regional trends in the data. Both structures used in the construction of the WVUF variogram model were of exponential form. The first had a range of 156 meters and sill of 0.015, whereas the second had a range of 231 meters and sill of 0.415. The model used to describe the variogram of the WERF data was more complex. The first structure had a spherical form that described the steep increase in variability between separation distances of less than approximately 100 meters. The second structure, of exponential form, had a range of 485.48 and sill of 0.20. It is important to note that the fit of this model is heavily

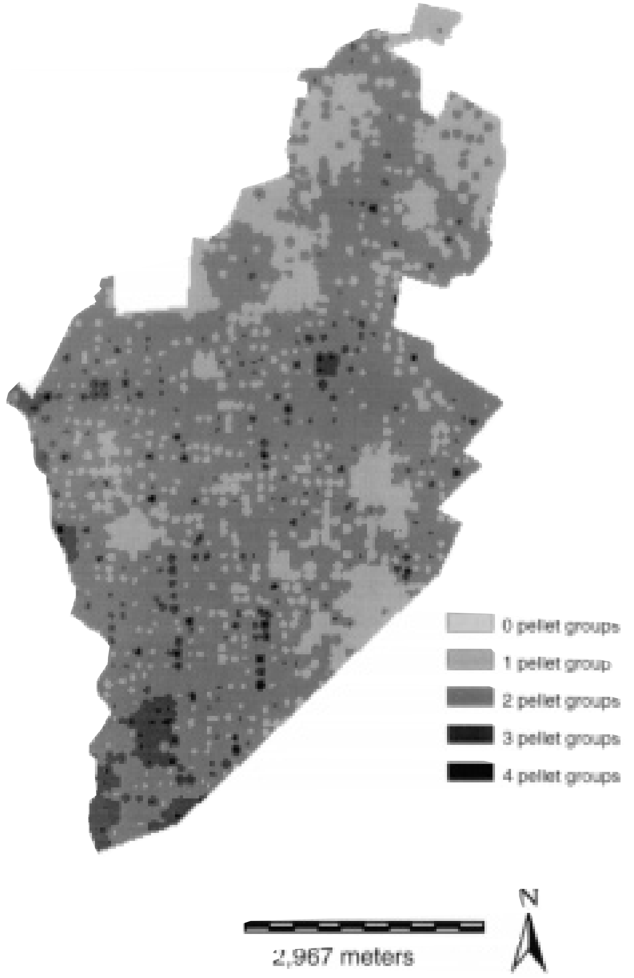

Figure 6—Kriged estimate map for the WERF. Cells are 15 by 15 meters.

influenced by the 78 pairs at very close separation distances.

Overall, pellet group counts made on the WERF were more spatially variable than those made on the WVUF. The combined sill of 0.95 for the WERF data was more than double the 0.43 sill for the WVUF data. This was expected given the differences in the sites. The WVUF is relatively uniform and continuous with harvest units clustered along a rudimentary road system. The majority of the site has not been actively managed for 80 years and little of the total area is in early successional stages. In addition, the forest is heavily hunted and deer populations are not excessive (no quantitative estimate of deer densities are available at this time). In contrast, the WERF has been heavily impacted by both partial timber harvest and dispersed regeneration harvests. The habitat is more variable and more of the total area is covered by young forest. The area is also heavily bisected by a well-developed road system; however, the WERF is gated and hunting is prohibited. Deer populations are consequently much higher on the WERF; the excessive browsing of woody understory vegetation and paucity of herbaceous vegetation indicate that the site is saturated with deer.

The relative lack of sample data at small separation distances combined with the strong localized trend in the WERF data appears in the kriged estimate map as an “egg crate” effect; the overwhelming influence of the most proximal samples caused localized depressions and peaks in the kriged estimates. This problem apparently could be rectified through more intensive sampling. Based on the results of the variography, sampling intensity may have to be more than doubled.

In contrast, the kriged estimates for the WVUF reveal clearly defined areas of heightened deer activity. Less overall variability, a weaker local trend, and a stronger regional trend enabled us to produce an apparently useful predictive map of baseline deer activity patterns. In general, the peaks in estimated pellet counts correspond quite well with known canopy disturbances. For example, peak deer activity occurred in the northeastern lobe of the forest, which was impacted by high overstory mortality in 1990-91 followed by timber salvage operations. The spike of activity at the tip of the northwestern lobe corresponds with a gas pipeline right-of-way. Peaks in the southeastern corner correspond with recent timber harvest operations.

CONCLUSIONS

In general, pellet group data appears to contain enough spatial contagion to allow the production of estimate maps using 2-dimensional ordinary point kriging. Although the estimates have not yet been validated, it appears that spikes in pellet group densities on the WVUF correspond to overstory disturbances and resultant increased availability of woody browse. The results on the WERF were less conclusive and generally indicate that the technique may be less useful in areas of extremely high deer population and variable habitat. However, based on the presence of localized spatial dependence in the small number of samples collected at less than 240 meters, it appears that a follow-up study at higher sampling intensity is warranted before concluding that the technique cannot be used on the WERF or similarly impacted sites.

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REFERENCES

Alverson, W.S.; Waller, D.M.; Solheim, S.L. 1988. Forests too deer: edge effects in northern Wisconsin. Conserv. Biol. 2: 348-358.

Behrend, D.F.; Mattfeld, G.F.; Tierson, W.C.; Wiley, J.E., III. 1970. Deer density control for comprehensive forest management. Journal of Forestry. 68: 695-700.

Bennett, L.J.; English, P.F.; McCain; R. 1940. A study of deer populations by use of pellet-group counts. Journal Wildl. Manage. 4: 398-403.

Biondi, F.; Myers, D.E.; Avery, C.C. 1994. Geostatistically modeling stem size and increment in an old-growth forest. Canadian Journal of Forest Research. 24: 1354-1368.

Bowersox, T.W.; Storm, G.L.; Tzilkowski, W.M. 1993. Developing regeneration in woodlots at Gettysburg National Military Park. In: Proceedings, 9th central hardwood forest conference; 1993 March 8-10. W. Lafayette, IN: Purdue University: 409-417.

deCalesta, D. 1994a. Impact of deer on interior forest songbirds in northwestern Pennsylvania. Journal Wildl. Manage. 58: 711-718.

deCalesta, D.; Stout, S. 1997. Relative deer density and sustainability: a conceptual framework for integrating deer management with ecosystem management. Wildl. Soc. Bull. 25: 252-258.

Deutsch, C.V.; Journel, A.G. 1998. GSLIB: geostatistical software library and user's guide. New York: Oxford University Press. 369 p.

Eastman, J.R. 1997. IDRISI for windows: user's guide. Version 2.0. Worcester, MA: Clark University.

Eberhardt, L.; Van Etten, R.C. 1956. Evaluation of the pellet group count as a deer census method. Journal Wildl. Manage. 20(1): 70-74.

Eyre, F.H., ed. 1980. Forest cover types of the United States and Canada. Washington, DC: Soc. of Amer. For. 148 p.

Ford, W.M.; Johnson, A.S.; Hale, P.E. 1994. Nutritional quality of deer browse in Southern Appalachian clearcuts and mature forests. For. Ecol. and Manage. 67: 149-157.

Ford, W.M.; Johnson, A.S.; Hale, P.E.; Wentworth, J.M. 1993. Availability and use of spring and summer woody browse by deer in clearcut and uncut forests in the Southern Appalachians. Southern Journal of Applied Forestry. 17: 116-119.

Fuller, T.K. 1991. Do pellet counts index white-tailed deer numbers and population change? J. Wildl. Manage. 55(3): 393-396.

Goovaerts, P. 1997. Geostatistics for natural resources evaluation. New York: Oxford University Press. 483 p.

Gribko, L.S.; Liebhold, A.M.; Hohn, M.E. 1995. Model to predict gypsy moth (Lepidoptera: Lymantriidae) defoliation using kriging and logistic regression. Environ. Entomol. 24: 529-537.

Hohn, M.E. 1988. Geostatistics and petroleum geology. New York: Van Nostrand Reinhold.

Hohn, M.E.; Liebhold, A.M.; Gribko, L.S. 1993. A geostatistical model for forecasting the spatial dynamics of defoliation caused by the gypsy moth, *Lymantria dispar* (Lepidoptera: Lymantriidae). Environ. Entomol. 22(5): 1066-1075.

Horsely, S.B.; Marquis, D.A. 1983. Interference by weeds and deer with Allegheny hardwood reproduction. Canadian Journal of Forest Research. 13: 61-69.

Isaaks, E.H.; Srivastava, R.M. 1989. An introduction to applied geostatistics. New York: Oxford University Press. 561 p.

Johnson, A.S.; Hale, P.E.; Ford, W.M. [and others]. 1995. White-tailed deer foraging in relation to successional stage, overstory type and management of Southern Appalachian forests. Am. Midl. Nat. 133: 18-35.

Jordan, J.S. 1967. Deer browsing in northern hardwoods after clearcutting. Res. Pap. NE-57. U.S. Department of Agriculture, Forest Service.

Kemp, W.P.; Kalaris, T.M.; Quimby, W.F. 1989. Rangeland grasshopper (Orthoptera: Acrididae) spatial variability: macroscale population assessment. Journal Econ. Entomol. 82: 1270-1276.

Liebhold, A.M.; Zhang, X.; Hohn, M.E. [and others]. 1991. Geostatistical analysis of gypsy moth (Lepidoptera: Lymantriidae) egg mass populations. Environ. Entomol. 20: 1407-1417.

Marquis, D.A. 1974. The impact of deer browsing on Allegheny hardwood regeneration. Res. Pap. NE-308. U.S. Department of Agriculture, Forest Service.

Marquis, D.A. 1981. The effects of deer browsing on timber production in Allegheny hardwood forests of northwestern Pennsylvania. Res. Pap. NE-475. U.S. Department of Agriculture, Forest Service.

Marquis, D.A.; Eckert, P.L.; Roach, B.A. 1976. Acorn weevils, rodents, and deer all contribute to oak-regeneration difficulties in Pennsylvania. Res. Pap. NE-356. U.S. Department of Agriculture, Forest Service.

Marquis, D.A.; Grisez, T.J. 1978. The effect of deer exclosures on the recovery of vegetation in failed clearcuts on the Allegheny Plateau. Res. Note NE-270. U.S. Department of Agriculture, Forest Service. 5 p.

McCormick, L.H.; Groninger, J.W.; Penrod, K.A.; Ristau, T.E. 1993. Deer exclusion effects on understory development following partial cutting in a Pennsylvania oak stand. In: Proceedings, 9th central hardwood forest conference; 1993 March 8-10. W. Lafayette, IN: Purdue University: 418-427.

Neff, D.J. 1968. The pellet group count technique for big game trend, census, and distribution: a review. Journal Wildl. Manage. 23: 597-614.

Pannatier, Y. 1996. VARIOWIN: software for spatial data analysis in 2D. New York: Springer-Verlag. 91 p.

Pauley, E.F.; Nodvin, S.C.; Nicholas, N.S. [and others]. 1996. Vegetation, biomass, and nitrogen pools in a spruce-fir forest of the Great Smoky Mountains National Park. Bull. Torrey Bot. Club. 123: 318-329.

Pelletier, D.; Parma, A.M. 1994. Spatial distribution of pacific halibut (Hippoglossus stenolepis): an application of geostatistics to longline survey data. Canadian Journal Fish. Aquat. Sci. 51: 1506-1518.

Redding, J. 1987. Impact of deer on forest vegetation and timber production in northern Pennsylvania. In: Proceedings, symposium on deer, forestry, and agriculture: interactions and strategies for management; 1987 June 15-17. Warren, PA: Allegheny Sec. Soc. Amer. For.: 23-32.

Robertson, G.P. 1987. Geostatistics in ecology: interpolating with known variance. Ecology. 68(3): 744-748.

Robinette, W.L.; Ferguson, R.B.; Gashwiler, J.S. 1958. Problems involved in the use of deer pellet group counts. Trans. N. Am. Wildl. Conf. 23: 411-425.

Rossi, R.E.; Mulla, D.J.; Journel, A.G.; Franz, E.H. 1992. Geostatistical tools for modeling and interpreting ecological and spatial dependence. Ecological Monographs. 62: 277-314.

Shafer, E.L.; Grisez, T.J.; Sowa, E. 1961. Results of deer exclosure studies in northeastern Pennsylvania. Res. Note NE-121. U.S. Department of Agriculture, Forest Service.

Sheffer, D.E. 1987. Deer herd management in Pennsylvania. In: Proceedings, symposium on deer, forestry, and agriculture: interactions and strategies for management; 1987 June 15-17. Warren, PA: Allegheny Sec. Soc. Amer. For.: 138-142.

Storm, G.L.; Yahner, R.H.; Cottam, D.F.; Vecello, G.M. 1989. Population status, movements, habitat use, and impact of white-tailed deer at Gettysburg National Historic Site, Pennsylvania. Tech. Rep. NPS/MAR/NRTR - 89/043. U.S. Department of the Interior, National Park Service. 513 p.

Stout, S.L.; DeCalesta, D.S.; Horsley, S.B. [and others]. 1995. Ecosystems management research in hardwood forests dominated by deer. In: Proceedings, 10th central hardwood forest conference; 1995 March 5-8; Morgantown, WV. 577 p.

Tierson, W.C.; Patric, E.F.; Behrend, D.F. 1966. Influence of white-tailed deer on the logged northern hardwood forest. Journal of Forestry. 64: 801-805.

Tilghman, N.G. 1989. Impacts of white-tailed deer on forest regeneration in northwestern Pennsylvania. Journal Wildl. Manage. 53: 24-532.

Trumbull, V.L.; Zielinski, E.J.; Aharrah, E.C. 1989. The impact of deer browsing of the Allegheny forest type. Northern Journal of Applied Forestry. 6: 162-165.

Van Etten, R.C.; Bennett, C.L., Jr. 1965. Some sources of error in using pellet-group counts for censusing deer. Journal Wildl. Manage. 29: 723-729.

Villard, M.A.; Maurer, B.A. 1996. Geostatistics as a tool for examining hypothesized declines in migratory songbirds. Ecology. 77(1): 59-68.

Wentworth, J.M.; Johnson, A.S.; Hale, P.E.; Kammermeyer, K.E. 1990. Seasonal use of clearcuts and food plots by white-tailed deer in the Southern Appalachians. Proc. Annu. Conf. Southeast. Fish and Wildl. Agencies. 44: 215-223.

Silviculture

DEVELOPMENT OF OAK REGENERATION NINE YEARS AFTER SHELTERWOOD CUTTING AND CLEARCUTTING ON THE COASTAL PLAIN OF WEST TENNESSEE

Wayne K. Clatterbuck, Philip Blakley and Paul Yielding[1]

Abstract—Quantity, size and species composition of hardwood reproduction were compared in oak-hickory (*Quercus* spp.-*Carya* spp.) stands following clearcutting and two intensities of shelterwood cutting (leaving 40 and 60 ft^2 of basal area) on Chickasaw State Forest in the Coastal Plain of west Tennessee. Measurements were taken prior to the timber harvest and at 3-year intervals for 9 years after the harvest. The clearcut treatment had the greatest number of oaks per acre, while the shelterwood treatments had a greater proportion of oaks compared to total stems. After 9 years, the total number of stems in each treatment decreased while the percentage of oaks greater than 1-foot tall increased. The development of oak regeneration and the resulting stand dynamics are discussed based on these 9-year regeneration data.

INTRODUCTION

Concerns about the aesthetics of clearcuts have caused a renewed interest in the shelterwood method of regeneration within the even-aged silvicultural system for regenerating oaks. Several authors have prescribed guidelines for regenerating oaks using the shelterwood method (Hannah 1987, Loftis 1983, Loftis 1990, Sander 1979, Sander and Graney 1993). However, Smith (1981) states that "many shelterwood recommendations contain the statement that details are uncertain and more research is needed." The complexity and variability of natural hardwood stands with their multitude of species, differential growth rates, varying site productivities, different stand structures and conditions, silvical characteristics, past disturbances, and previous management regimes are not amenable to a single prescription.

In theory, the shelterwood method should create the microenvironment required by oaks for successful establishment and nurturing early seedling growth. In practice, a fine line exists on how much overstory to leave in promoting oak regeneration. Heavy shelterwood cuts favor the development of fast-growing intolerant species such as yellow-poplar (*Liriodendron tulipifera* L.) rather than oaks. Lighter cuts may encourage more tolerant, less desirable species such as maples (*Acer* spp.), beech (*Fagus grandifolia* Ehrh.), and elms (*Ulmus* spp.) at the expense of oaks (Clatterbuck and Meadows 1993, Hodges 1989).

This study provides a 9-year history of the development of regeneration following three overstory treatments: clearcutting and two intensities of shelterwood cuts leaving 40 and 60 ft^2 of residual basal area. The objective of the research is to compare the regeneration dynamics between treatments in size, number and species composition. Although this is an unreplicated case study, it provides information about the oak regeneration process.

STUDY AREA

The study was conducted on a 46-acre upland hardwood stand at Chickasaw State Forest (CSF) in Hardeman County, TN, located approximately 20 miles south of Jackson in southwestern Tennessee and managed by the Tennessee Department of Agriculture, Division of Forestry. Soils are Typic Hapludults (Smithdale series), formed in loamy Coastal Plain deposits, severely eroded and moderately drained (Ditzler and others 1994). The study area is on a convex, moderately steep (8 to 20 percent), strongly dissected slope. Annual precipitation averages 50 inches, usually evenly distributed in all seasons. Average site index (base age 50) for upland oaks ranges from 75 to 80 feet (Schnur 1937).

The CSF was part of the federal Resettlement Administration purchase of land during the 1930's. The area occupied by this study was formerly a portion of two farms. Old fences indicate that all or part of the area was grazed, cultivated, or both. Fire scars on many of the trees attest to burning before state acquisition.

Ages of the dominant and co-dominant trees range from 60 to 90 years. The predominant species are oaks: white (*Q. alba* L.), northern red (*Q. rubra* L.), scarlet (*Q. coccinea* Muenchh.), black (*Q. velutina* Lam.), and southern red (*Q. falcata* Michx.). Other species include hickories, sweetgum (*Liquidambar styraciflua* L.), yellow-poplar, American beech, dogwood (*Cornus florida* L.) and blackgum (*Nyssa sylvatica* Marsh.).

METHODS

The 46-acre stand was divided into four subunits of approximately equal size, each of which represented the stand as a whole with regard to site quality, aspect, position on slope, and slope gradient. Four treatments were imposed, one for each subunit: clearcut and three shelterwood cuts leaving 40, 50 and 60 ft^2 of residual basal area respectively in dominant and codominant trees of desirable species.

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Table 1—Treatment size, number of plots, initial stocking and summary of timber harvest volumes in 1982 at Chickasaw State Forest, Tennessee

Treatment	Area	Plots	Initial stocking	Average total volume cut	Volume/acre cut
	Acres	*#*	*Ft^2/acre*	*Board feet*[a]	*Bd.ft./acre*
Clearcut	10	23	75	110,700	11,070
40-ft^2 BA	12	23	85	34,780	2,900
60-ft^2 BA	12	26	80	15,200	1,270

[a] International ¼ inch log rule.

Before the actual timber harvest in 1982, 97 permanent, 0.1-acre plots were established over the entire 46 acres (Table 1). Plots were spaced 2.2 chains apart on parallel lines 2.2 chains apart. All stems 1.5 inches dbh and larger were recorded by species, diameter, and merchantable height (height to base of crown). Cut and leave trees were tallied for each plot. Regeneration was measured on a 0.02 acre circular plot at the plot center of the 0.1 acre plot. Height of all woody stems 1 foot and taller to 1.5 inches dbh were counted by species. All woody stems less than 1 foot were also recorded by species.

For the clearcut, all trees greater than 12 inches in diameter at breast height (dbh; 4.5 feet) were commercially harvested. The remaining stems over 1.5 inches dbh were cut and left on the site. Leave trees for the shelterwood cut were selected on the basis of species, form, vigor, and crown position class. Oaks were favored as leave trees, particularly white, northern red, southern red, and black oaks. Hickories were also favored, but were secondary to the oaks. Species such as yellow-poplar, sweetgum, American beech, and blackgum were not favored. However, trees of these species were left when necessary to maintain the required residual basal area. Midstory and understory stems in the shelterwood treatments were not intentionally cut, but may have been disturbed during the removal of overstory trees during the commercial harvest. Volumes harvested for each treatment area are presented in Table 1.

Regeneration data were collected in 1982 before the harvest and at 3-year intervals: 1985, 1988 and 1991. The data for the shelterwood cut with 50 ft^2 of residual basal area are not presented in this paper.

RESULTS

Total Stems per Acre

The clearcut unit had the greatest number of stems per acre, both less than and greater than 1-foot tall, at the time of the harvest (Table 2). Nine years later, the clearcut treatment with more than 4,700 stems/acre, increased by three times the stems over 1-foot tall, while the amount of reproduction less than 1 foot in height decreased.

Table 2—Reproduction present in three treatment areas before and 9 years after harvest, Chickasaw State Forest, Tennessee

Treatment	1982 ≤1 ft.	1982 >1 ft.	1991 ≤1 ft.	1991 >1 ft.
	----------------	*Stems/acre*		----------------
Clearcut	4,845	1,537	2,974	4,783
40-ft^2 BA	3,193	964	5,068	2,011
60-ft^2 BA	3,216	986	3,408	657

Both shelterwood treatments had fewer stems than the clearcut treatment in 1982. The number of stems less than 1 foot in height increased for the 40 ft^2 of residual basal area (BA) treatment and remained essentially the same for the 60-ft^2 BA treatment. The total number of stems greater than 1-foot tall was approximately 2,000 stems/acre, doubling in the 9 years following the harvest in the 40-ft^2 BA treatment, but decreased from 986 to 657 for the 60-ft^2 BA treatment.

The total number of stems per acre greater than one foot in height increased dramatically between the initial measurement in 1982 to the next measurement in 1985 for all three treatments (Figure 1). The amount of reproduction decreased progressively in the clearcut and the 60-ft^2 BA shelterwood treatments during the next two measurement periods in 1988 and 1991. The 40-ft^2 BA treatment continued to increase to a measured high point in 1988, then decreased.

Species Composition

For all three treatments, the other species category, primarily of shade-tolerant species such as red maple (*A. rubrum* L.), sourwood (*Oxydendrum arboreum* (L.) DC.), blackgum, dogwood, serviceberry (*Amelanchier arborea* (Michx.f.) Fern.), sassafras (*Sassafras albidum* (Nutt.) Nees), hophornbeam (*Ostrya virginiana* (Mill.) K. Koch), and the intolerant black cherry (*Prunus serotina* Ehrh.),

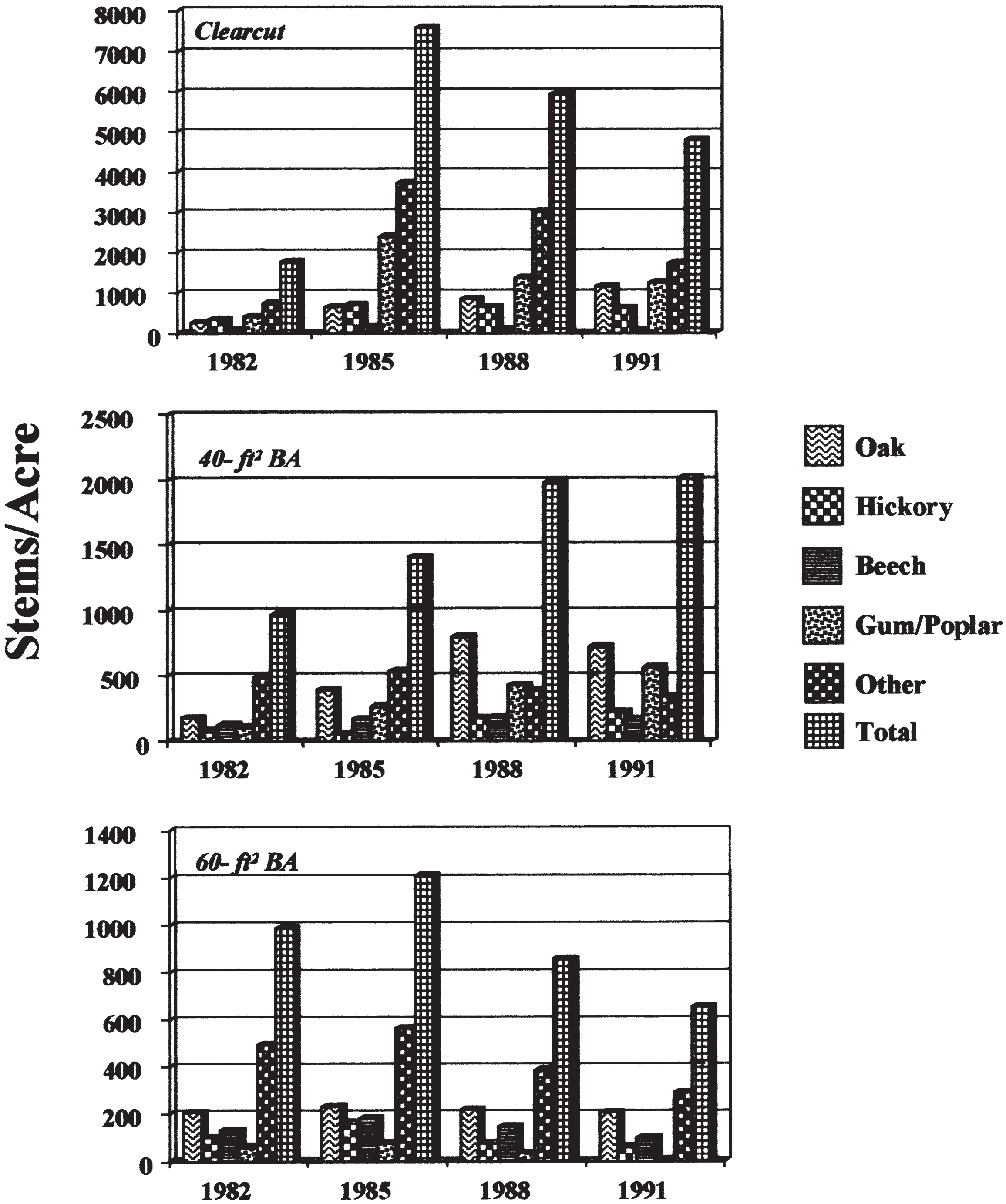

Figure 1—Stems per acre of reproduction greater than 1-foot tall for each species group by the clearcut and two shelterwood treatments for four measurements from 1982-1991, Chickasaw State Forest, Tennessee.

composed the majority of the reproduction greater than 1-foot tall before the harvest (Table 3). Oak species ranged from 14 to 21 percent of the stems, hickory from 8 to 18 percent, beech 3 to 13 percent and gum/poplar 6 to 23 percent. After 9 years, the percentage of oaks increased for every treatment. The gum/poplar percentage component increased for the clearcut and the 40-ft^2 BA treatment, but decreased for the 60-ft^2 BA treatment. Hickory remained steady after 9 years for both shelterwood treatments, but decreased for the clearcut treatment. Beech was a minor component in the clearcut treatment, but maintained or increased its presence in the shelterwood treatments. After 9 years, the other species category was still prominent in the clearcut and the 60-ft^2 BA shelterwood treatments, but decreased by more than half in the 40-ft^2 BA treatment.

The change in the amount of reproduction greater than 1 foot in height by species and by treatment for each

Table 3—Species composition of stems greater than 1 foot in height before and 9 years after timber harvest by treatment, Chickasaw State Forest, Tennessee

	1982			1991		
Species	Clearcut	40-ft^2 BA	60-ft^2 BA	Clearcut	40-ft^2 BA	60-ft^2 BA
	Percent			*Percent*		
Oaks[a]	14	18	21	24	36	31
Hickories	18	8	10	13	11	9
Beech	3	13	13	1	8	15
Gum/poplar[b]	23	11	6	26	28	1
Other[c]	42	50	50	36	17	44

[a] White, black, northern red, southern red, scarlet oaks.
[b] Yellow-poplar and sweetgum.
[c] Red maple, blackgum, black cherry, sassafras, dogwood, elms, sourwood, serviceberry, hophornbeam.

measurement period is shown in Figure 1. The number of stems per acre for each species category increased for each treatment from 1982 to 1985 except for the hickory component in the 40-ft^2 BA treatment that decreased slightly. The amount of reproduction for the gum/poplar and other species components decreased from 1985 to 1991, the oak component increased, and the hickory component remained essentially the same. Beech was not a significant species in the clearcut treatment.

In contrast, for the 40-ft^2 BA shelterwood treatment, the number of stems per acre for oaks increased from 1982 to 1988 and decreased slightly in 1991 (Figure 1). Other species increased initially, then progressively decreased in 1988 and 1991. Gum/poplar increased steadily from 106 stems/acre in 1982 to 563 stems/acre in 1991. Beech remained between 125 and 185 stems/acre for the duration of the study. Hickory slowly increased from 54 stems/acre in 1985 to 221 stems/acre in 1991.

The amount of reproduction per acre increased for each species from 1982 to 1985 in the 60-ft^2 BA treatment (Figure 1). The number of stems for each species decreased slowly after 1985. The gum/poplar component had the fewest number of stems of any species category.

Height Growth

The initial mean height measurements in 1982 for each species were similar for each of the treatment areas (Table 4). Heights for the 9 years from 1982 to 1991 increased the most in the clearcut treatment with other species (primarily dogwood and blackgum) and gum/poplar as the tallest components, followed by oaks and hickories.

Table 4—Mean heights of regeneration greater than 1 foot in height before and 9 years after harvest by treatment, Chickasaw State Forest, Tennessee

	1982			1991		
Species	Clearcut	40 ft^2 BA	60 ft^2 BA	Clearcut	40 ft^2 BA	60 ft^2 BA
	Feet			*Feet (range)*		
Oaks[a]	1.8	2.0	2.0	6.8 (1-17)	2.4 (1-6)	1.8 (1-8)
Hickories	4.2	3.1	2.8	4.0 (2-15)	3.2 (1-4)	2.3 (1-5)
Beech	4.6	5.0	3.9	—	5.8 (1-11)	3.9 (1-7)
Gum/poplar[b]	2.6	1.9	2.7	9.2 (2-15)	5.2 (1-10)	—
Other[c]	4.2	3.4	3.6	8.5 (2-22)	4.6 (1-10)	4.5 (1-14)

[a] White, black, northern red, southern red, scarlet oaks.
[b] Yellow-poplar and sweetgum.
[c] Red maple, blackgum, black cherry, sassafras, dogwood, elms, sourwood, serviceberry, hophornbeam.

Height growth was less for each species in the 40-ft^2 BA shelterwood treatment. Beech, gum/poplar, and other species were the tallest reproduction (Table 4). Hickory and oaks remained the same with little increase. The 60-ft^2 BA treatment had the least height growth. The mean height for all species decreased. Other species and the beech components were the most prominent reproduction. Gum/poplar was not a component in this treatment.

DISCUSSION

Clearcut

The total number of stems per acre increased from more than 1,500 in 1982 to more than 7,500 stems/acre in 1985, a five-fold increase (Figure 1). Most of this increase occurred in the gum/poplar component and sprouting from severed stems between 1.5 to 12 inches dbh in the other species category (dogwood, red maple, and blackgum). Clearcutting tends to initially favor the establishment and growth of shade-intolerant, fast-growing, light-seeded species such as yellow-poplar and sweetgum and sprouting stems with established root systems. The number of stems decreased from the 1985 measurement through the 1988 and 1991 measurements indicating that the site was fully-occupied and that a sorting of stems was occurring. The larger, more competitive stems were beginning to express dominance to form the upper canopy, while the less competitive stems were being relegated to the subcanopy or dying. The poplar/gum component was the most aggressive and tallest species in the overstory with the mostly shade-tolerant, other species category beginning to slip into the lower canopy. The number of oaks per acre increased steadily during each measurement period from less than 250 in 1982 to almost 1,150 stems in 1991 (Figure 1), but their mean height was shorter than the gum/poplar component (Table 4).

40-Ft^2 Basal Area

Compared to the clearcut treatment with more than 7,500 stems/acre at its peak in 1985, the highest amount of total reproduction per acre for the 40-ft^2 basal area treatment was at just over 2,000 stems/acre in 1991 (Figure 1). The 40-ft^2 per acre residual basal area cut provided space and sunlight to favor growth of species present in the lower canopy layers. The rate of increase for the total number of stems per acres decreased between the 1988 and 1991 measurement periods, indicating the gaps in the upper canopy were beginning to close, thus limiting the amount and intensity of sunlight received by lower canopy stems. The proportion of oaks and gum/poplar increased during the measurement periods for this shelterwood cut, while the other species category decreased after a peak in 1985. However, mean heights for oaks were substantially less than those of gum/poplar, beech and other species (Table 4).

60-Ft^2 Basal Area

The total number of stems per acre for the 60-Ft^2 BA treatment increased from the initial measurement in 1982 to a peak in 1985, then decreased through 1991 (Figure 1). Compared to the clearcut and 40-ft^2 BA treatments with maximums of approximately 7,500 and 2,000 stems/acre respectively, the 60-ft^2 BA reached a peak of slightly more than 1,200 stems in 1985. This shelterwood treatment had fewer trees cut in the overstory (or more trees retained) and canopy gaps were smaller allowing the canopy to close more quickly than the 40-ft^2 BA treatment, thus limiting continued growth and development of reproduction. The 60-ft^2 BA treatment after 9 years favored the development of shade-tolerant species in the understory such as beech, red maple, blackgum, dogwood and elms. The gum/poplar component averaged 7 stems per acre and heights less than 1 foot after 9 years. Oaks were present at 200+ stems/acre, but their mean heights were less than 2 feet after 9 years. Mean heights for all species categories for the 60-ft^2 BA treatment were greatly diminished compared to the other treatments.

Oak Regeneration

Oaks are present in different numbers and proportions for each treatment (Table 5). The clearcut treatment has the greatest number and size of oaks (Table 4), but the lowest proportion of stems of the three treatments. Most mature oak stands did not start as fully-stocked stands. Usually oaks are a component of mixed stands in their infancy; during stand development they emerge as the dominant or codominant species (Clatterbuck and Hodges 1988; Oliver 1980a). During the stand initiation stage (Oliver 1980b), oaks are present, but often are inconspicuous in the jungle of woody and non-woody vegetation found after clearcutting. However, some oak stems will persist and stratify above other vegetation forming a dominant canopy. Research in other areas suggest as few as 50 to 100 well-spaced oak stems per acre are needed, if those oaks are assured of becoming upper canopy trees (Clatterbuck and Hodges 1988, Oliver 1980a).

Although this study follows for 9 years the development of oak regeneration in relation to other species, the research is inconclusive whether the oaks present will become dominant or codominant trees. The data indicate that even though the oaks are smaller, on the average than other species, there are individuals (Table 4) that will be dominant with time. However, the fast growth of yellow-poplar will surpass many of the oaks (O'Hara 1986). The question then remains how many oaks are needed and how many oaks will actually succeed to become mature trees.

Table 5—Oak regeneration greater than 1-foot tall by treatment in 1991, Chickasaw State Forest, Tennessee

	Stems/acre	Proportion of total stems
	Number	*Percent*
Clearcut	1,128	24
40 ft^2 BA	680	36
60 ft^2 BA	188	31

With the 40-ft^2 BA shelterwood cutting, most species in the lower canopy benefited from the increased light duration and intensities. The cutting was heavy enough to allow species to grow and develop throughout the 9 years. Canopy closure was just beginning after 9 years. Oaks composed 36 percent of the total stems per acre (Table 5) and a greater number of stems than the gum/poplar component (Figure 1), but were less than half the average height of yellow-poplar. This treatment favored an increase in height of gum/poplar and beech. When released from the overstory, the gum/poplar and beech are positioned to become the dominant canopy. Oaks, though greater in number, are not in a position to grow with the taller species. The tallest oak individual sampled in the reproduction plots and the only individual over 6 feet tall measured 6.4 feet. Although few stems from the other species category of shade-tolerant species will be part of the overstory, most of these stems will probably be relegated to the midstory or succumb.

Loftis (1983, 1990) has successfully used the shelterwood method in the southern Appalachian mountains to increase the size of oak regeneration in relation to other species before release from the overstory. The method used was not to create gaps in the overstory, but to cut trees in the midstory to allow sunlight to filter through the overstory and not be intercepted by an additional canopy layer. This study used a different approach by not treating the midstory in the shelterwood treatments but by creating gaps in the overstory. Either method should be satisfactory if the microenvironment (primarily light conditions) created will favor the oaks at the expense of the shade-tolerant species or the faster-growing intolerant species (Hodges 1989).

The 60-ft^2 BA shelterwood treatment had the lowest mean heights for each of the species categories (Table 4). Most species groups increased in quantity from 1982-1985, then decreased as the canopy closed from the growth of the residual overstory (Figure 1). Oaks composed 31 percent of the reproduction after 9 years (Table 5), but only averaged 1.8 feet tall. The gum/poplar component was non-existent, while the more tolerant beech and other species categories were the tallest and composed a greater proportion of stems. This treatment promoted the more tolerant species at the expense of the oaks and intolerant species.

CONCLUSIONS

Both the clearcut and the 40-ft^2 BA shelterwood treatment appear to provide conditions conducive to promoting the development and growth of oak regeneration. The clearcut treatment had the greatest number of oaks, while the shelterwood had the greatest proportion of oaks in relation to the total number of stems. In both treatments, the oaks will continue to compete with the faster-growing gum/poplar component for overstory prominence. Even if oaks are not the primary component of the overstory, they will be a part of the overstory. The shade cast by the overstory in the light shelterwood cutting (60-ft^2 BA) promoted the shade-tolerant species and not the oaks.

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REFERENCES

Clatterbuck, W.K.; Meadows, J.D. 1993. Regenerating oaks in the bottomlands. In: Loftis, David L.; McGee, Charles E., eds. Oak regeneration: serious problems, practical recommendations: Symposium proceedings; 1992 September 8-10; Knoxville, TN. Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 184-195.

Clatterbuck, Wayne K.; Hodges, John D. 1988. Development of cherrybark oak and sweetgum in mixed, even-aged bottomland stands in central Mississippi, U.S.A. Canadian Journal of Forest Research. 18(1): 12-18.

Ditzler, Craig A.; Thomas, David W.; Cody, Richard M. [and others]. 1994. Soil survey of Chester County, Tennessee. U.S. Department of Agriculture, Soil Conservation Service. 164 p.

Hannah, P.R. 1987. Regeneration methods for oaks. Northern Journal of Applied Forestry. 4(2): 97-101.

Hodges, John D. 1989. Regeneration of bottomland oaks. Forest Farmer. 49(1): 10-11.

Loftis, David L. 1983. Regenerating Southern Appalachian mixed hardwood stands with the shelterwood method. Southern Journal of Applied Forestry. 7(4): 212-217.

Loftis, David L. 1990. A shelterwood method for regenerating red oak in the Southern Appalachians. Forest Science. 36(4): 917-929.

O'Hara, Kevin L. 1986. Development patterns of residual oaks and oak and yellow-poplar regeneration after release in upland hardwood stands. Southern Journal of Applied Forestry. 10: 244-248.

Oliver, Chadwick Dearing. 1980a. Even-aged development of mixed-species stands. Journal of Forestry. 78: 201-203.

Oliver, Chadwick Dearing. 1980b. Forest development in North America following major disturbance. Forest Ecology and Management. 3: 153-168.

Sander, Ivan L. 1979. Regenerating oaks with the shelterwood system. In: Holt, Harvey A.; Fischer, Burnell C., eds. Proceedings: regenerating oaks in upland forests: the 1979 John S. Wright forestry conference; 1979 February 22-23; West Lafayette, IN. Purdue University, Department of Forestry and Natural Resources; Indiana Cooperative Extension Service: 54-60.

Sander, Ivan L.; Graney, David L. 1993. Regenerating oaks in the Central States. In: Loftis, David L.; McGee, Charles E. eds. Oak regeneration: serious problems, practical recommendations: Symposium proceedings; 1992 September 8-10; Knoxville, TN. Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 174-183.

Schnur, G. Luther. 1937. Yield, stand, and volume tables for even-aged upland oak forests. Agric. Tech. Bull. 560. Washington, DC: U.S. Department of Agriculture. 87 p.

Smith, H. Clay. 1981. Normal hardwood silvicultural and management systems. In: Proceedings of the national silvicultural workshop; 1981 June 1-5; Roanoke, VA. Washington, DC: U.S. Department of Agriculture, Forest Service, Division of Timber Management: 66-76.

RELEASING SHELTERED NORTHERN RED OAK DURING THE EARLY STEM EXCLUSION STAGE

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Abstract—The utility of releasing sheltered northern red oak was examined in mesic hardwood stands in north central West Virginia. Different levels of release were applied in the spring of 1996 - six growing seasons after planting 2-0 seedlings that were protected with 5 ft corrugated plastic shelters. The planting was done in a 7.77 acre forest opening that developed abundant natural regeneration, but lacked a self-sustaining oak component. The most intensive release consisted of cutting all stems within a 5 ft radius of the sheltered tree (n = 20). A less intensive release consisted of the removal of only stems that were significantly overtopping the sheltered tree (n = 20). A control group in which the trees were not released was also incorporated into the study (n = 19). Trees at the beginning of the study either were codominant or intermediate in crown class. Total height of northern red oak prior to treatment averaged 9.7 ft and did not differ among treatments. Height growth after two growing seasons was statistically different among treatments (p = 0.0151). Average height growth was 4.09 ft and 3.96 ft for the minimal release and the control group, respectively, and 2.78 ft for the full release. Height of the competing vegetation after two growing seasons also differed by treatment (p = 0.0045) and was 16.10 ft in the minimal release, 17.88 ft in the control, and 20.80 ft in the full release. There was also strong statistical evidence that crown class distribution differed among treatments after two growing seasons (p = 0.0032). In the control group, 26 percent of the trees were newly classified as overtopped in two growing seasons. In the full release, 5 percent of the trees were overtopped and none were classified as overtopped in the minimal release. Considering the height variation between the desired tree and the competing vegetation, preliminary results indicate release operations that leave a moderate level of woody competition around trees like red oak with weak epinastic control, may prove to be the most effective at retaining a favorable competitive status of the desired tree.

INTRODUCTION

The retention of oak species in mixed-oak forest ecosystems throughout the eastern and central United States has been an enduring problem for the last four decades (e.g. Carvell and Tyron 1961, Johnson 1993, Lorimer 1984, McGee 1975, Weitzman and Trimble 1957). Throughout this period, substantial reductions in the oak component of mixed species stands followed most harvesting and regeneration efforts. Leading Noss and others (1995) to conclude that high-quality oak-hickory forests had declined significantly in parts of the central and southern Appalachians. Furthermore, the problem is not avoided in older second-growth and old-growth forests excluded from harvesting. Such protected areas are currently undergoing changes in species composition toward greater abundance of shade-tolerant species and a reduction in oaks and other mid-seral species (Abrams and Downs 1990, Parker and others 1985, Smith and Miller 1987).

Restoration of the oak component following harvesting using natural regeneration methods continue to be evaluated (Loftis 1990, Schlesinger and others 1993, Schuler and Miller 1995) and the use of prescribed fire to improve oak competitiveness has recently received more attention (Keyser and others 1996, Kruger and Reich 1997). However, such methods may require a period of 10 to 20 years, or significantly longer, to develop sufficient oak regeneration. This time period may deter acceptance of such practices in forests characterized by short ownership tenure. An alternative regeneration technique being evaluated is the use of plastic tree shelters to protect planted or natural oak seedlings during the early stages of stand development after overstory removal (Lantagne and others 1990, Schuler and Miller 1996, Smith 1993).

Much has been reported on the operational use of tree shelters during the first few years after installation to enhance seedling establishment and early growth (Brissette 1996). Tree shelters have been shown to increase height growth, root growth, and total biomass of northern red oak seedlings (*Quercus rubra*) for several years after planting in new forest openings and in old fields (Lantagne 1996, Ponder 1996, Schultz and Thompson 1996). However, accelerated growth rates of sheltered trees return to normal after the tree's crown emerges from the shelter (Schuler and Miller 1996). As the effect of the shelter diminishes on northern red oak growth rates, sympatric species often exhibit accelerated rates of height growth as competition for growing space intensifies during the early stem exclusion stage. Black cherry (*Prunus serotina*), yellow-poplar (*Liriodendron tulipifera*), sweet birch (*Betula lenta*) and other species in the central Appalachians often reach 60 ft in total height 20 years after a major disturbance (Miller and others 1995). A vigorously growing codominant oak is expected to be about 40 ft tall in the same time period (Schnur 1937). In general, red oak height growth will lag behind black cherry and yellow-poplar on good to excellent growing sites (Lamson and Smith 1978, Smith 1983). To offset this discrepancy in height growth, silvicultural treatments may be useful to sustain the planted oaks until they are firmly established as codominant trees in the developing stand.

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The extent, number, and timing of needed treatments is unknown. In this paper, we evaluate the utility of releasing sheltered northern red oak in mesic hardwood stands. In doing so, we examined different levels of cleaning and assessed the effects on the crop tree, the competing vegetation and their competitive interaction.

METHODS

Study Area

The study took place on the Fernow Experimental Forest (39.03°N, 79.67°W) near Parsons, West Virginia. The area is referred to as the Allegheny Mountains Section of the Central Appalachian Broadleaf Forest (M221B) as designated by the U.S. Department of Agriculture, Forest Service National Hierarchical Framework of Ecological Units (McNab and Avers 1994). The draft landtype association has been nominally referred to as the Allegheny Front Sideslopes (M221Ba10) and represents over 99,000 acres within the Monongahela National Forest alone (DeMeo and others 1995). The potential natural vegetation of this area is referred to as mixed mesophytic (Braun 1950). Overstory species composition is often quite varied and may include over 20 different species within a spatial scale of roughly 10 or more acres. Common species include northern red oak, sugar maple (*Acer saccharum*), basswood (*Tilia americana*), yellow-poplar, black cherry, and sweet birch. The local climate is characterized by an annual precipitation of 55 to 58 inches (Pan and others 1997), a mean temperature of 61.5°F from April through September and 35.1°F from October through March, resulting in 120 to 140 frost-free days. Distinguishing topographic features of this landtype association include sideslopes of plateau blocks ranging from 2200 to 3500 ft in elevation.

This study was part of a larger study designed to develop an operational method for establishing northern red oak and other difficult to regenerate hardwood seedlings. The portion of the study reported here was initiated in a new forest opening in the spring of 1990. The study area was prepared by clearcutting the overstory during the 1989 growing season in 7.7 acre research compartment nominally referred to as Fork Mountain Gate. Sawlog-size material (11.0 inches dbh and larger) was skidded from the site, and all other stems (1.0 inch dbh and larger) were felled and left in place. Merchantable material removed from the site averaged 19,000 board feet•acre^{-1} (International 1/4 inch). The study area is classified as an excellent growing site equivalent to a northern red oak site index of 80. The initial study included treatment combinations involving 5-ft corrugated plastic tree shelters and transplanted northern red oak seedlings. Partial two-year results were reported by Smith (1993) and referenced as Site 4. Oak seedlings in all treatments involving shelters were significantly taller than those in unsheltered treatments.

Release Test Procedures

Different levels of release were applied in the spring of 1996 - six growing seasons after the initial outplanting. All saplings selected were protected by shelters from the beginning of the study. The shelters were still in place after the most recent remeasurement in the spring of 1998 without any notable decrease in structural integrity. The most intensive release consisted of cutting all stems within a 5-ft radius of the sheltered tree. A less intensive release consisted of the removal of only stems that were significantly overtopping the sheltered tree. A control group in which the trees were not released was also incorporated into the study. Trees at the beginning of the study either were codominant or intermediate in crown class. Dominant trees that were developing in the absence of significant competition from woody vegetation were not included. Total height of northern red oak prior to treatment averaged 9.7 ft and did not differ among treatments ($p = 0.433$) according to analysis of variance results. Three levels of thinning were replicated 20 times for each thinning level and 19 times for the control group. The assignment of treatments to individual trees was randomized. Therefore the design was an unbalanced completely randomized one-way analysis of variance.

Data Analysis

A fixed-effects model was assumed for all statistical analysis. Following data acquisition, the data were assembled and tested for model adequacy using graphical and statistical techniques two years after experimental implementation. The Shapiro-Wilkes statistic and p-value were generated using the SAS univariate procedure for each dependent variable of interest. The results did not indicate the error component deviated from normality for crop tree height growth ($p = 0.8481$). However, the normality assumption did not hold when the height of competing vegetation was used as the response variable in the model ($p = 0.0398$). Therefore, a transformation of the response variable was employed using the natural logarithm which yielded acceptable results with respect to the normality assumption ($p = 0.7473$).

The Brown-Forsythe test was used to evaluate the equal variance assumption related to height growth ($p = 0.9516$) and the log transformed height of competing vegetation ($p = 0.5105$). The associated large p-values do not provide any statistical evidence that the variance associated with either variable differed for the level of thinning. Graphical analysis of residuals corroborated these conclusions. Based on these findings, we established that the model format was adequately describing the response to the treatments and proceeded with tests of significance for both crop tree height growth and height of the competing vegetation. The Duncan's multiple range test was used to further break down the response to treatments when significant differences were found. This test controls the comparison-wise error rate, not the experiment-wise error. So the actual probability of incurring a type I error among all comparisons is greater than the stated alpha. Finally, the chi-square statistic was used to assess the treatment effects on crown class distribution two years after experimental implementation.

RESULTS

Height growth two years after release was significantly related to the degree of release ($p = 0.0151$). To interpret these findings we conducted a Duncan's multiple range test ($\alpha = 0.05$) on height growth response. The results identified two distinct responses and associated the minimal release procedure, hereafter referred to as a codominant release, and the no release as one group and the 5-ft release as another. The codominant release and the no release responses were superior to the 5-foot radial release in terms of height growth (Table 1). Similar to height growth, the average total height of released trees among treatments stratified in accordance with differences in height growth (Table 1, Fig. 1), although treatment means were not significantly different ($p = 0.4603$).

Not clear from the height growth response was the effect of the release treatment on the height of the competing vegetation. This requires consideration because it is the height of the released tree relative to the height of the competing vegetation that determines the potential vigor and survival of the released tree. The log-transformed height of the competing vegetation did not differ by release method ($p = 0.1583$) prior to treatment and averaged 12.57 feet. After two growing seasons, the log-transformed height

Table 1—Treatment means for two-year northern red oak height growth response as a function of release procedure

Treatment	2-yr height growth	Total height	N
	feet[a]	*feet*	
Codominant release	4.09a	13.73	20
No release	3.96a	13.25	19
5-foot release	2.78b	12.83	20

[a] Means with the same letter are not significantly different ($\alpha = 0.05$).

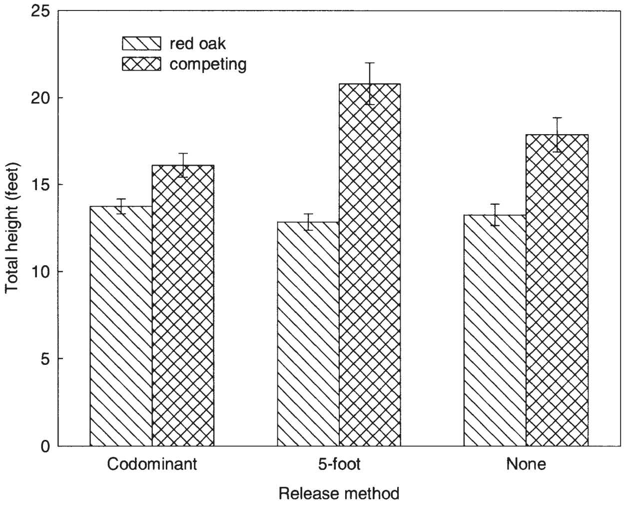

Figure 1—Mean total heights and standard errors of previously sheltered northern red oak and the natural regeneration two years after initiating release procedures and eight years after stand reinitiation.

of the competing vegetation differentiated by treatment ($p = 0.0060$). Transformation of the dependent variable was necessary both prior to and after treatment because the residuals from the general linear model were not normally distributed ($p = 0.0398$). The Duncan's multiple range test identified the codominant release and the 5-ft release as distinct responses (Table 2). The 5-ft release had a positive effect on the height of the competing vegetation relative to no release, while the codominant release had a negative effect. The mean total height of the competing vegetation with respect to the 5-ft release was more than 4 feet taller than the competing vegetation of the codominant release. Based on the results presented in Table 1, this difference equates to two or more years of red oak height growth. Such a discrepancy in total height could lead to substantial differences in northern red oak survival during the early stages of stand development.

The crown class distribution of the released oak trees further characterizes the relationship between the height of the competing vegetation and the height of released trees. In 1996 prior to treatment, crown class distribution did not differ by treatment (chi-square = 1.061, $p = 0.588$). Prior to treatment, 61 percent of the trees were classified as codominant and 39 percent of the trees were classified as intermediate. After two years, crown class distribution had changed among treatments (chi-square = 11.457, $p = 0.022$). The trees that were not released declined substantially in terms of competitive status. Only 37 percent of the unreleased trees retained codominant classification, while the same percentage was classified as intermediate, and 26 percent were newly classified as overtopped (Table 3). This illustrates the rate of which conditions can change during this stage of stand development. In only two growing seasons, one of every four unreleased northern red oak trees became overtopped. It is unlikely that an overtopped red oak will regain a more competitive crown position and high rates of mortality are anticipated for such trees.

In contrast to the unreleased red oak trees, crown class distribution improved for the group of trees that received the codominant release. No trees in this category were classified as overtopped either prior to or after two growing

Table 2—Treatment means for height of the competing vegetation two years after implementation as a function of release procedure

Treatment	Duncan group[a]	Total height	Total height
		log-feet	*feet*
5-foot release	A	3.00280	20.14
No release	AB	2.85793	17.43
Codominant release	B	2.76181	15.83

[a] Means with the same letter are not significantly different ($\alpha=0.05$).

Table 3—Change in northern red oak crown class two years after release by treatment category. Table values are frequency and column percent in parentheses

	Treatment			
Crown class	Initial	Codominant release	5-foot release	No release
Codominant	36	16	13	7
	(61)	(80)	(65)	(37)
Intermediate	23	4	6	7
	(39)	(20)	(30)	(37)
Overtopped	0	0	1	5
	(0)	(5)	(5)	(26)

seasons and the ratio of codominant to intermediate trees changed from roughly 60/40 in the spring of 1996 to 80/20 after two additional growing seasons. This is to be expected because any and all overtopping trees were cut. However, this procedure has also resulted in greater height growth relative to the full 5-ft release. It is this combination of factors that led to a more favorable crown class distribution.

The 5-ft release resulted in one tree becoming overtopped and had little effect on the remaining crown class distribution (Table 3). However, the disparity between the height of the released tree and the height of the competing vegetation was greatest in this category (Fig. 1). A continuation of this trend will likely result in a substantial, and perhaps an abrupt, decline in the competitive status of these trees.

When release treatments began, competing vegetation comprised 16 species and did not differ by treatment category (chi-square = 31.975, p = 0.369). Sweet birch was the principal species with 37 percent of the total number of dominant competing stems. This species is an aggressive competitor during early stand development and has little commercial value in the central Appalachian region. The eventual species composition of third-growth stands dominated by this species is uncertain. Other major competitors included yellow-poplar (10 percent), pin cherry (*P. pennsylvanica*) (10 percent), and black cherry (7 percent). Only one sampled northern red oak representing about 2 percent of the total dominant competing stems was recorded in 1996 at the onset of release efforts.

Two years after initial release, species composition of the competing vegetation did not yet differ by treatment, but some evidence suggests a trend may be developing toward more stratification (chi-square = 31.458, p = 0.087). We anticipate that species with slower juvenile growth rates will dominate the competing vegetation of the codominant release method in the future because faster growing species will have been selectively removed. Species richness in 1998 declined to 11 species with sweet birch continuing to dominate the young stand (36 percent), while yellow-poplar increased to 20 percent and pin cherry increased to 22 percent over all treatment categories. Black cherry declined slightly to about 5 percent of the total and northern red oak was no longer represented in the sample.

DISCUSSION

Variations in stand density generally are assumed to have little effect on individual tree height growth (Smith 1986). Height growth is so closely associated with the productive potential of a site that the concept of site index, height of dominant and codominant trees at a convenient base age, is based on an understanding that height growth is relatively independent of stocking level. However, the height growth of trees used in site index equations are predicated on trees that have retained dominant or codominant status throughout their development. The use of dominant or codominant trees implies competition for above ground growing space is one factor that determines growth characteristics. This is evident when trees are isolated or are grown at low stocking levels. In particular, species with weak epinastic control (i.e., relatively minimal influence by the tree's terminal bud over the length and orientation of lateral branches) that are grown at low stocking levels might exhibit reductions in height growth (Oliver and Larson 1996). Weak epinastic control is a trait of northern red oak and many other angiosperms native to the eastern United States (Kramer and Kozlowski 1979).

A decline in height growth associated with extreme reductions in stand level density was documented for oak and yellow-poplar saplings in southeastern Ohio by Allen and Marquis (1970). They concluded that short-term height growth was maximized at about 70 percent stocking based on experimental manipulation of density. Codominant oaks thinned to 70 percent of full stocking grew 1.5 ft annually which was three times the growth rate of trees grown at 20 percent stocking. Concomitant yellow-poplar height growth at the same site was also optimized at the same stocking level. As such, total release of an individual oak (e.g. as in thinning to a 5-ft radius) may inadvertently provide a partial release to a bordering yellow-poplar. The unintentional consequence of such a cleaning would be to slow oak height growth and increase height growth of the competition. This may partially explain the results reported here regarding the accelerated height growth of the competing vegetation in the full release treatment (Fig. 1).

Some investigators have also concluded the effects of a full release on hardwood saplings decreases height growth, whereas a partial release can improve height growth. A full release of suppressed sapling-sized hickories in Ohio and Indiana generally slowed height growth for the first three years after release relative to unreleased trees (Nixon and others 1983). The authors speculated that release cutting around hardwood saplings would have been beneficial if the crown release had not been complete. In an extensive study conducted in Connecticut, Ward (1995) reported significant height growth depression of released codominant black (*Q. velutina*) and scarlet (*Q. coccinea*) oaks four years after treatment. Moreover, fully released dominant northern red oak also exhibited less height growth than partially released individuals. But response to release can also change over time and variable responses have been reported. In a seven-year-old even-aged hardwood stand in West Virginia, fully released red oak grew slower in height than unreleased trees for the first two years but differences were not evident after five years (Trimble 1974). Similar results were noted for yellow-poplar (Johnson and others 1997). Others have found that crop tree release did not affect height growth in young Appalachian hardwood stands (Smith and Lamson 1983, Wendel and Lamson 1987).

The results of this study and the work of others previously noted suggests release thinning in very young stands has the potential to be a useful silvicultural tool. It is apparent that partial release thinning has stimulated short-term juvenile height growth in some cases. It is not yet apparent if long-term survival and competitive status of mesic site oaks can be maintained or improved by release thinning in young stands. Existing recommendations indicate it is better to delay stand manipulations until competitive pressures have selected the most vigorous individuals. For example, to select the best quality timber trees for crop tree release, Perkey and others (1993) recommend waiting until the trees are at least 25 ft tall. However, to influence species composition, release work may need to begin before codominant trees reach this stage of development. This is especially true when objectives for the stand include the retention of oak that is often not abundant relative to other species. The need for work in very young stands when oak perpetuation is an objective is illustrated by the sharp decline in crown classification of unreleased oaks reported in this study (Table 3). Moreover, in a practical sense, the release of previously sheltered oaks protects the existing investment in tree shelters.

Oak regeneration problems continue to plague forest managers throughout the eastern and central United States. Forest stands that included a significant oak component during both old-growth and second-growth stages, are often characterized by a greatly diminished oak component following second-growth harvesting. Silviculturists continue to explore ways to develop abundant understory oak to promote oak regeneration following harvesting activities. However, harvesting and the regeneration of new stands continues unabated while prescriptions for abundant natural oak regeneration remain unresolved. As such, to meet common timber, wildlife, and diversity goals that include the retention of oak, forest managers will need to develop techniques for increasing the survival rate of the scarce oak stems common in many young third-growth forests. Release thinning in very young stands may be beneficial in that respect. Release thinning of previously sheltered northern red oak is essential to maintain competitive status of these trees on mesic sites. Future research needs to focus on the long-term survival of released trees and the degree, timing, number, and cost of releases necessary to achieve oak retention relative to site characteristics and the competing vegetation.

MANAGEMENT IMPLICATIONS

Preliminary results indicate that releasing previously sheltered northern red oak can be beneficial to retaining this species during the early stages of stand development. The following suggestions are offered as guidelines for implementing release procedures on small to medium-sized operations:

- Wait as long as possible to do the release work but not until the desired oak trees are overtopped by competing vegetation, usually five to six years after planting on good sites.

- Retain all trees not overtopping the desired tree. This will induce the released tree to sustain rapid height growth and maintain strong epinastic control.

- Schedule annual release work for individual stands for the best results. Individual trees receiving a codominant release require very little treatment time. With initial planting density on a 25-foot basis, it is reasonable for a two person crew to inspect and release, if required, 600 to 1,000 trees per day. Simple hand tools are sufficient for doing the release work. Do not plant and shelter more northern red oak than can be released in the central Appalachians as release work will be vital to their survival.

- Target faster growing, short-lived species with lower commercial value for removal and retain slower growing species, if possible. Altering the species composition immediately surrounding the desired oak by favoring slower growing species may lessen the need for repeated release efforts.

- Release work should be done during the dormant season to take advantage of better visibility. When releasing sheltered oaks, it is easier to find the desired trees when the shelters are retained on the tree, even though the shelter may be providing no protection to the tree. Occasionally, manually splitting the tree shelter is necessary because diameter growth becomes restricted by a shelter that has not degraded sufficiently.

- Scheduling early stand release work for sheltered oaks also facilitates selecting highly desirable natural regeneration for similar cultural efforts. If such trees are included, flag the tree so that it can be more easily relocated during the following years.

- On good to excellent growing sites in the central Appalachians, it may require a period of 10 to 15 years (e.g., from 5 to 15 years after stand reinitiation) of release work before long-term survival of oak throughout stand development will be achieved. The frequency of required release work will decline as tree sizes increase.

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REFERENCES

Abrams, M.D.; Downs, J.A. 1990. Successional replacement of old-growth white oak by mixed mesophytic hardwoods in southwestern Pennsylvania. Canadian Journal of Forest Research. 20: 1864-1870.

Allen, R.H.; Marquis, D.A. 1970. Effect of thinning on height and diameter growth of oak and yellow-poplar saplings. Res. Pap. NE-173. U.S. Department of Agriculture, Forest Service.

Braun, E.L. 1950. Deciduous forests of Eastern North America. Philadelphia: Blakiston Co.

Brissette, J.C., ed. 1996. Proceedings of the tree shelter conference; 1995 June 20-22; Harrisburg, PA. Gen. Tech. Rep. NE-221. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 80 p.

Carvell, K.L.; Tryon, E.H. 1961. The effect of environmental factors on the abundance of oak regeneration beneath mature oak stands. Forest Science. 7: 98-105.

DeMeo, T.; Tracy, L.; Wright, L. 1995. Landtype associations of the Monongahela National Forest. U.S. Department of Agriculture, Forest Service, Monongahela National Forest. 1: 250,000; UTM map projection.

Johnson, J.J.; Bollig, J.J.; Rathfon, R.A. 1997. Growth response of young yellow-poplar to release and fertilization. Southern Journal of Applied Forestry. 21(4): 175-179.

Johnson, P.S. 1993. Perspectives on the ecology and silviculture of oak-dominated forests in the Central and Eastern States. Gen. Tech. Rep. NC-153. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 28 p.

Keyser, P.D.; Brose, P.; Van Lear, D.; Burtner, K.M. 1996. Enhancing oak regeneration with fire in shelterwood stands: preliminary trials. Transactions of the North American wildlife and natural resources conference. Washington, DC: Wildlife Management Institute: 215-219.

Kramer, P.J.; Kozlowski, T.T. 1979. Physiology of woody plants. San Diego, CA: Academic Press. 811 p.

Kruger, E.L.; Reich, P.B. 1997. Responses of hardwood regeneration to fire in mesic forest openings. I. Post-fire community dynamics. Canadian Journal of Forest Research. 27: 1822-1831.

Lamson, N.I.; Smith, H.C. 1978. Response to crop tree release: sugar maple, red oak, black cherry, and yellow-poplar saplings in a 9-year-old stand. Res. Pap. NE-354. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 8 p.

Lantagne, D.O. 1996. Effects of tree shelters on planted red oaks in Michigan. In: Brisette, John, ed. Proceedings, tree shelter conference; 1995 June 20-22; Harrisburg, PA. Gen. Tech. Rep. NE-221. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 24-28.

Lantagne, D.O.; Ramm, C.W.; Dickman, D.I. 1990. Tree shelters increase heights of planted oaks in a Michigan clearcut. Northern Journal of Applied Forestry. 7: 24-26.

Loftis, D.L. 1990. Predicting post-harvest performance of advanced red oak reproduction in the Southern Appalachians. Forest Science. 36(4): 908-916.

Lorimer, C.G. 1984. Development of the red maple understory in northeastern oak forests. Forest Science. 30: 3-22.

McGee, C.E. 1975. Regeneration alternatives in mixed oak stands. Res. Pap. SE-125. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 8 p.

McNab, W.H.; Avers, P.E. 1994. Ecological subregions of the United States: section descriptions. Publ. WO-WSA5. U.S. Department of Agriculture, Forest Service.

Miller, G.W.; Schuler, T.M.; Smith H.C. 1995. Method for applying group selection in central Appalachian hardwoods. Res. Pap. NE-696. Radnor, PA: U.S. Dept. of Agriculture, Forest Service, Northeastern Forest Experiment Station. 11 p.

Nixon, C.M.; McClain, M.W.; Landes, R.K. [and others]. 1983. Response of suppressed hickories to release cutting. Wildlife Society Bulletin. 11(1): 42-46.

Noss, R.F.; LaRoe, E.T.; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Bio. Rep. 28. Washington, DC: National Biological Services.

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

Pan, C.; Tajchman, S.J.; Kochenderfer, J.N. 1997. Dendroclimatological analysis of major forest species of the central Appalachians. Forest Ecology and Management. 98: 77-87.

Parker, G.R.; Leopold, D.J.; Eichenberger, J.K. 1985. Tree dynamics in an old-growth, deciduous forest. Forest Ecology and Management. 11: 31-57.

Perkey, A.W.; Wilkins, B.L.; Smith, H.C. 1993. Crop tree management in eastern hardwoods. Tech. Pap. NA-TP. U.S. Department of Agriculture, Forest Service: 19-93.

Ponder, F. 1996. Tree shelter effects on stem and root biomass of planted hardwoods. In: Brisette, John, ed. Proceedings, tree shelter conference; 1995 June 20-22; Harrisburg, PA. Gen. Tech. Rep. NE-221. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 19-23.

Schlesinger, R.C.; Sander, I.L.; Davidson, K.R. 1993. Oak regeneration potential increased by shelterwood treatments. Northern Journal of Applied Forestry. 10: 149-153.

Schnur, G.L. 1937. Yield, stand, and volume tables for even-aged upland oak forests. Tech. Bull. 560. U.S. Department of Agriculture. 85 p.

Schuler, T.M.; Miller, G.W. 1995. Shelterwood treatments fail to establish oak reproduction on mesic forest sites in West Virginia. In: Gottschalk, Kurt W.; Fosbroke, Sandra L.C., eds. Proceedings, 10th central hardwood conference; 1995 March 5-8; Morgantown, WV. Gen. Tech. Rep. NE-197. Northeastern Forest Experiment Station: 375-388.

Schuler, T.M.; Miller, G.W. 1996. Guidelines for using tree shelters to regenerate northern red oak. In: Brisette, John, ed. Proceedings—tree shelter conference; 1995 June 20-22; Harrisburg, PA. Gen. Tech. Rep. NE-221. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 37-45.

Schultz, R.C.; Thompson, J.R. 1996. Tree shelters for plantation establishment of bareroot red oak and black walnut in 5 Midwestern States. In: Brisette, John, ed. Proceedings, tree shelter conference; 1995 June 20-22; Harrisburg, PA. Gen. Tech. Rep. NE-221. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 19-23.

Smith, D.M. 1986. The practice of silviculture. New York: John Wiley. 527 p.

Smith, H.C. 1983. Growth of Appalachian hardwoods kept free to grow from 2 to 12 years after clearcutting. Res. Pap. NE-528. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 6 p.

Smith, H.C. 1993. Development of red oak seedlings using plastic shelters on hardwood sites in West Virginia. Res. Pap. NE-672. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 7 p.

Smith, H.C.; Lamson, N.I. 1983. Precommercial crop-tree release increases diameter growth of Appalachian hardwood saplings. Res. Pap. NE-354. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 7 p.

Smith, H.C.; Miller, G.W. 1987. Managing Appalachian hardwood stands using four regeneration practices—34 year results. Northern Journal of Applied Forestry. 4: 180-185.

Trimble, G.R. 1974. Response to crop-tree release by 7-year-old stems of red maple stump sprouts and northern red oak advance reproduction. Res. Pap. NE-303. U.S. Department of Agriculture, Forest Service.

Ward, J.S. 1995. Intensity of precommercial crop-tree release increases diameter and crown growth in upland hardwoods. In: Gottschalk, Kurt W.; Fosbroke, Sandra L.C., eds. Proceedings, 10th central hardwood conference; 1995 March 5-8; Morgantown, WV. Gen. Tech. Rep. NE-197. Northeastern Forest Experiment Station: 388-398.

Weitzman, S.; Trimble, G.R. 1957. Some natural factors that govern the management of oaks. Pap. 88. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 40 p.

Wendel, G.W.; Lamson, N.I. 1987. Effects of herbicide release on the growth of 8- to 12-year-old hardwood crop trees. Res. Pap. NE-594. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 8 p.

UNDERPLANTED NORTHERN RED OAK 17 YEARS AFTER THINNING AND UNDERSTORY CONTROL AND 8 YEARS FOLLOWING OVERSTORY REMOVAL

Ron Rathfon and Wayne Werne[1]

Abstract—Thinning-from-below and understory vegetation control treatments were applied to an oak-dominated stand in 1981 and 1982. Northern red oak seedlings (*Quercus rubra* L.) were planted in the understory to determine whether artificial regeneration was a viable option in ensuring that oak remain a prominent component of the succeeding stand. The overstory was removed in a harvest in 1991. Eight years following overstory removal, the thinning plus understory control treatment combination had many more underplanted red oaks in a competitive position than either treatment alone. The underplanted red oaks made up 40 percent of the total oak regeneration in the thinning plus understory control treatment.

METHODS

The study was established on a south-facing, 6 to 20 percent slope site in the unglaciated region of southern Indiana. White Oak Site Index was 70 feet. Oak (white, red, black and scarlet) comprised 75 percent of the sawtimber basal area of the original stand. In 1981, a small feller-buncher was used to thin-from-below (THIN) one-half of the area of each of four two-acre blocks creating one-acre harvest units. In September 1982, an understory vegetation control treatment (VEGCON) was applied to a one-half acre area within each THIN treatment plot by mist-blowing herbicides (Wright and others 1985).

Northern red oak seedlings were planted on a 9.5 foot by 9.5 foot grid for a total of 96 seedlings per VEGCON split plot, or 432 seedlings per acre. Various types of planting stock were tested for suitability in underplantings (Wright and others 1985).

The overstory was removed in a harvest in early 1991. Underplanted red oaks were remeasured in 1998. Natural regeneration was also assessed. A 1-inch dbh tree was considered to be the minimum size for viable, competitive growing stock in 1998.

RESULTS

The THIN treatment produced far fewer underplanted red oaks meeting minimum size requirements (greater or equal to 4.5 feet in height) at the time of overstory removal than recommended by Sanders and others (1976) to ensure red oak prominence in the final stand (Table 1). VEGCON produced over 2-1/2 times more underplanted red oaks per acre meeting the minimum size criteria than did THIN. While THIN+VEGCON had over 5-1/2 times the number of

Table 1—Survival, mean height, and stocking of underplanted red oak just prior to overstory removal and 8 years after overstory removal

	1990			1998				
						Viable trees[a]		
Treatment	Survival	Mean height	Viable trees	Survival	Mean height	1-in d.b.h. class	2-in d.b.h. class	Total stocking
	Percent	*Feet*	*Stems/ac*	*Percent*	*Feet*	--------- *Stems per acre* ---------		
Untreated	46.6	1.32	0.0	-	-	-	-	-
VEGCON	58.3	2.63	101.1	30.7	10.1	38.3	5.6	43.9
THIN	30.2	2.48	38.6	19.8	9.2	20.3	2.3	22.6
THIN+VEGCON	52.9	3.25	216.7	38.5	13.4	72.1	27.0	99.1

[a] Stems greater than or equal to 4.5 feet in height.

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criteria-meeting trees than did THIN, it still did not meet the minimum stocking required to result in an oak dominated stand (Sanders and others 1976). Eight years following overstory removal, 99.1 underplanted red oaks per acre remained in a competitive position (greater or equal to 1 inch dbh) in THIN+VEGCON (Table 1). THIN+VEGCON had almost 4-1/2 times more competitive underplanted red oak stems than that of THIN and over twice that of VEGCON.

Natural oak regeneration (red, black and white oak) was a significant component of the stand. It comprised 9.4 percent, 18.9 percent and 12.1 percent of the total 1-inch-dbh-and-greater stocking for VEGCON, THIN and THIN+VEGCON, respectively (Table 2). The combined natural and underplanted oak in THIN+VEGCON accounted for one-third of all stems over 2 inches dbh. Forty percent of the total oak stocking in THIN+VEGCON was from underplanted red oaks.

REFERENCES

Sanders, IL; Johnson, P.S.; Watt, R.F. 1976. A guide for evaluating the adequacy of oak advance reproduction. Gen. Tech. Rep. NC-23. U.S. Department of Agriculture, Forest Service. 7 p.

Wright, G.M.; Pope, P.E. ; Fischer, B.C. [and others]. 1985. Chemical weed control to establish natural and artificial regeneration in a mechanically thinned upland hardwood stand. Proceedings third biennial southern silvicultural conference; Atlanta. Gen. Tech. Rep. SO-54. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 266-272.

Table 2—Stocking by species, treatment, and d.b.h. class 8 years after overstory removal

	VEGCON		THIN		THIN+VEGCON	
	D.b.h. class		D.b.h. class		D.b.h. class	
Species	1 in.	2 in.	1 in.	2 in.	1 in.	2 in.
	Stems per acre					
Oaks[a]	144.4	28.9	182.9	38.5	166.9	83.5
Tulip-poplar	447.7	86.6	0.0	9.6	70.6	32.1
Black cherry	43.3	67.4	57.8	67.4	32.1	70.5
Ash	9.6	0.0	19.3	9.6	19.3	6.4
Hickory	9.6	0.0	173.3	9.6	70.6	0.0
Sugar maple	115.5	24.1	163.7	19.3	218.3	6.4
Misc.	370.7	28.9	260.0	28.9	423.7	51.3

[a] Natural regeneration plus underplanted red oak.

INFLUENCE OF CUTTING METHODS ON 12-YEAR-OLD HARDWOOD REGENERATION IN CONNECTICUT

Jeffrey S. Ward and George R. Stephens[1]

Abstract—Many upland oak forests are approaching economic and biological maturity. This has led to concern over species composition following stand regeneration. A case study of hardwood regeneration was established in 1984 in central Connecticut to study the effects of cutting method on regeneration composition. The six harvest treatment plots (4-6 acres) were on medium quality sites (SI=60). Harvest treatments were: shelterwood, diameter limit, coppice with standards, commercial clearcut, silvicultural clearcut, and uncut control. Regeneration was sampled by species and height class prior to harvest, and 5 and 12 years post-harvest using 16-28 1/300 acre plots per treatment.

Preharvest inventories found fewer than 50 oaks/acre ≥ 5 feet tall, much less than the 433/ac recommended for the western branch of the Central Hardwood region. However, oak regeneration accounted for 13 percent (150 oak/acre) of codominant and dominant trees 12 years after harvest on the silvicultural clearcut plot. This strongly suggests that there should be regional standards for evaluating oak regeneration. All treatments, except diameter limit, increased regeneration density. Shelterwood, but not other partial cuts, increased the number of oaks ≥ 5 feet tall. Five years after the shelterwood cut red maple and black birch densities had also increased by 88 percent and 1500 percent, respectively. Clearly, this will increase the competition oaks experience following final harvest. Partial cutting should not be prescribed when management goals include obtaining oak regeneration in future stands.

INTRODUCTION

There are 14 million acres of mature oak forest in the Northeast, and 14 million in the North Central states (Powell and others 1993). The present forest is a mixture of land continuously forested, but repeatedly cut, and land abandoned from agriculture during the 19th or 20th centuries. Many forests were established early in the 20th century following harvests for fuelwood and charcoal production. In Connecticut, sawtimber stand area has increased from 17 percent of the 1.8 million acres of commercial forest land in 1952 to 63 percent in 1985 (Dickson and McAfee 1988). Seedling and sapling stands account for only 10 percent of forested area. The unbalanced age class and economic maturity of the forest, combined with favorable stumpage prices, have recently led to increased harvest and regeneration activity.

As many forests are approaching biological maturity, concern over forest regeneration has increased. Commercially less valuable red maple (*Acer rubrum*) and birch (*Betula lenta* and *B. alleghaniensis*) now account for half of all stems in the sapling size class in Connecticut (Dickson and McAfee 1988). The lack of adequate oak regeneration is a well-recognized problem and challenge for foresters throughout the oak-hickory region (Holt and Fischer 1979, Lorimer 1989, Loftis and McGee 1993). Nearly two-thirds of the pole and sawtimber stands in Connecticut are classified as oak-hickory (Dickson and McAfee 1988). Even-aged systems are recommended for regenerating oak forests, except on low quality sites (Roach and Gingrich 1968, Sander 1977, Hibbs and Bentley 1983, Johnson 1993). There is a consensus that oak regeneration should be present before final harvest in mature stand if the succeeding stand is to contain abundant oak. However, the minimum size and amount of oak advance regeneration (seedlings and saplings established before final overstory removal), and ability of various cutting methods to provide the regeneration, varies among studies. Prescriptions or guidelines for regenerating oak are largely derived from research in the western portion of the central hardwood forest. Their applicability to the eastern branch of the central hardwood forest, specifically southern New England, remains untested.

This research was established to provide information on hardwood regeneration in the eastern section of the central hardwood forest. The general objectives of this research are: 1) determine whether the minimum size requirements for oak regeneration developed in the Midwest are too stringent for southern New England, and 2) determine the effect of different treatments (i.e., cutting methods) on regeneration composition by species group. The specific objective of this paper was to examine how regeneration differed among treatments 12-years after cutting.

METHODS

Research plots were established in 1984 in North Madison, Connecticut to examine the effect of various cutting methods on hardwood regeneration, especially oak. Six cutting method plots (4-6 acres each) were established on medium quality sites. Cutting methods were silvicultural clearcut, commercial clearcut, coppice with standards,

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diameter limit, shelterwood, and uncut control. These results should be treated as a case study due to the lack of replication. Specific treatment guidelines were:

Silvicultural clearcut—All residual trees > 2 inches dbh were cut after sawtimber and cordwood harvests;

Commercial clearcut—All merchantable trees ≥ 11 inches dbh were harvested;

Coppice with standards—Approximately 55 “crop trees” per acre were selected from among desirable poles and sawtimber. All other trees ≥ 5 inches dbh were harvested. At 10 to 15-yr intervals 5 crop trees per acre will be harvested and 5 new crop trees will be selected;

Diameter limit—the diameter limit was 16 inches for desirable species (oaks, ash, sugar maple) and 14 inches dbh for other species and low quality desirable species;

Shelterwood—stands were marked to remove 50 percent of basal area and 50 percent of upper canopy (dominant and codominant trees) by removing undesirable species and favoring retention of oak; and

Control—No cutting

The coppice with standards treatment was the first step of converting a relatively, even-aged stand into a multi-aged stand. Therefore, we selected crop tree by diameter classes rather than by age. With time we hope to convert this stand to a traditional coppice with standards stand. The coppice with standards is designed to favor selected crop trees for the production of high value timber or veneer logs while periodically removing all other merchantable material at regular intervals. This system, originally developed in Europe to provide fuel and waddle for tenant farmers and timber for estate owners, may be useful for small accessible tracts were a market for fuelwood exists. About 40-50 standards, potential crop trees, are selected on each acre from existing pole and sawtimber. Preference is given to desirable species, single stems, and excellent form. Eight to ten new standards are selected 10-20 year intervals. Then one fifth of crop trees are harvested along with all other stems larger than 5 inches dbh. Harvesting intervals are flexible to accommodate market conditions and growth.

Poles (4.5-10.5 inches dbh) and sawtimber (> 10.5 inches dbh) were inventoried prior to harvest using 4-7 prism plots (10 factor) per treatment. Prism point centers were randomly placed and then permanently marked using treated stakes. The following data were recorded for all trees ≥ 4.5 inches dbh: species, diameter, and crown class. For this paper, regeneration is defined as stems < 4.5 inches dbh. Regeneration was sampled prior to harvest, and 5 and 12 years post-harvest using 4 1/300 acre plots per each prism point, each 33 ft from the prism plot center in a cardinal direction (16-28 plots/treatment). Regeneration plot centers were also permanently located using treated stakes. For each inventory the following data were recorded for all stems: species, height (<1 ft, 1-1.9 ft, … 8-8.9 ft, ≥ 9 ft), and crown class. All regeneration, including 1-year-old seedlings, was tallied.

To simplify analysis and discussion, the 26 species found on the plots were collapsed into 5 species groups: Oak—*Quercus alba, Quercus coccinea, Quercus prinus, Quercus rubra, Quercus velutina*; Maple—*Acer rubrum, Acer saccharum*; Birch—*Betula alleghaniensis, Betula lenta*; Other—*Carya cordiformis, Carya glabra, Carya tomentosa, Fagus grandifolia, Fraxinus americana, Juglans cinerea, Liriodendron tulipifera, Nyssa sylvatica, Prunus serotina, Sassafras albidum*; and Minor—*Amelanchier arborea, Betula populifolia, Carpinus caroliniana, Castanea dentata, Cornus florida, Juniperus virginiana, Ostrya virginiana*.

RESULTS AND DISCUSSION

Prior to harvest, seedling density (< 5 feet tall) was high on all treatments, 15-29,000/acre (Table 1). Maple and oak were found on nearly every regeneration sample plot, 99 and 96 percent respectively. Birch was found on only 58 percent of the 1/300 acre plots. Maple dominated the seedling size class with an average of nearly 10,000/acre. Oak seedlings were quite numerous, 4,700/acre. Although oak seedling densities were high, there were few oak saplings (≥ 5 feet tall, Table 2). Only 25 of the 144 plots (17 percent) had an oak sapling. This is far below the 60 percent stocking recommended by Sander (1977). It should be noted our estimate of stocking was optimistic because

Table 1—Density (stems/acre) of seedlings (< 5 feet tall) prior to harvest by cutting method and species group in North Madison, CT

	Oak	Maple	Birch	Other	Minor	Total
Silvicultural clearcut	4489	9707	1093	3139	1693	20121
Shelterwood	2368	16404	2604	6000	1757	29132
Diameter limit	3814	13939	868	3171	1232	23025
Commercial clearcut	6840	7920	1185	1200	1245	18390
Coppice with standards	6638	3694	844	3319	300	14794
Uncut	4100	6663	4963	4325	675	20725
Mean	4700	9721	1926	3526	1150	21031

Table 2—Density (stems/acre) of saplings (≥ 5 feet tall) prior to harvest by cutting method and species group in North Madison, CT

	Oak	Maple	Birch	Other	Minor	Total
Silvicultural clearcut	11	311	129	107	418	975
Shelterwood	0	107	43	32	171	354
Diameter limit	11	300	11	129	214	664
Commercial clearcut	105	180	165	30	195	675
Coppice with standards	113	131	150	19	94	506
Uncut	13	250	163	113	225	763

we used a larger sampling plot (1/300 acre) than indicated by Sander (1/735 acre) and we assumed that 100 percent of pole and sawtimber oaks would resprout.

Five years after harvest, distinct differences had developed among the cutting methods (Table 3). Overall sapling (≥ 5 feet tall) and oak sapling density was higher on treatments that removed pole sized material, i.e., shelterwood and silvicultural clearcut. Oak sapling density increased on both the shelterwood and silvicultural clearcut plots, but decreased on the diameter limit, commercial clearcut, and coppice with standards plots. There was a much greater concurrent increase in maple and minor species density on the silvicultural clearcut and birch and minor species on the shelterwood plot. Sapling density had actually decreased on the other plots.

A clearer picture had emerged 12 years after cutting (Table 4). Oak sapling density remained high on the silvicultural clearcut plot after canopy closure. Although oak only accounted for 13 percent of upper canopy stems (Table 5), the 150 oaks/acre in the codominant and dominant crown class should be more than adequate to meet management goals. The mystery is where did the other upper canopy oaks come from? Only 2 of the 7 plots with an oak in the upper canopy at age 12 had had an oak at least 5 feet tall prior to the harvest. Three of the other 5 plots did have an oak 2-5 ft tall. On the remaining 2 plots, all oak regeneration was less than one foot tall prior to harvest.

Table 3—Density (stems/acre) of saplings (≥ 5 feet tall) 5 years after harvest by cutting method and species group in North Madison, CT

	Oak	Maple	Birch	Other	Minor	Total
Silvicultural clearcut	257	1296	171	300	1050	3075
Shelterwood	96	236	589	43	600	1564
Diameter limit	21	214	0	107	225	568
Commercial clearcut	90	105	105	30	255	585
Coppice with standards	19	75	113	0	19	225
Uncut	13	263	163	75	163	675

Table 4—Density (stems/acre) of saplings (≥ 5 feet tall) 12 years after harvest by cutting method and species group in North Madison, CT

	Oak	Maple	Birch	Other	Minor	Total
Silvicultural clearcut	889	804	450	900	1468	4511
Shelterwood	86	268	1511	107	696	2668
Diameter limit	21	257	461	129	268	1136
Commercial clearcut	105	60	195	60	285	705
Coppice with standards	19	19	56	0	19	113
Uncut	0	188	150	38	288	663

Table 5—Density (stems/acre) 12 years after a silvicultural clearcut by crown class and species group in North Madison, CT

	Oak	Maple	Birch	Other	Minor	Total
Dominant/codominant	150	332	107	193	386	1168
Intermediate	171	161	75	107	343	857
Suppressed (≥ 5 ft tall)	568	311	268	600	739	2486
Suppressed (1-4 ft tall)	461	54	96	793	193	1596
Suppressed (< 1 ft tall)	1018	568	32	1500	2057	5175

Thus, it appears that minimum size standards to obtain oak in the upper canopy at crown closure in the Northeast may be smaller than for the central (Sander 1977) and southeastern states (Loftis 1990).

The initial gain in oak sapling density on the shelterwood plot was not lost, but was being overwhelmed by a three-fold increase in birch sapling density through 12 years after cutting (Table 4). Shelterwood success has been spotty in other studies (Sander and Graney 1993), especially where deer herds are large (Smith 1993). Recommendations for regenerating oak involve two or three shelterwood cuts extended over a 10-30 year period (Hibbs and Bentley 1983, Martin and Hix 1988, Loftis 1993). Small diameter maple and birch readily sprout after cutting and respond well to release (Ward and Stephens 1993, 1996). Therefore, it is likely that shelterwood cuts extended over a long period will increase the proportion of birch and maple in the next stand.

This case study provides strong evidence that a new standard should be developed for evaluating oak regeneration in the Northeast. Furthermore, when increasing oak regeneration is part of the management objectives, partial cutting from above (cutting the largest trees) is counter-productive. Care should also be taken to not delay overstory removal with partial cutting from below. Assuming the relative competitive status is correlated with height; a long interval between the initial cut and the final harvest increases the competitive status of maple and birch relative to oak.

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REFERENCES

Dickson, D.R.; McAfee, C.L. 1988. Forest statistics for Connecticut—1972 and 1985. Resour. Bull. NE-105. U.S. Department of Agriculture, Forest Service.

Hibbs, D.E.; Bentley, W.R. 1983. A management guide for oak in New England. Storrs, CT: University of Connecticut, Cooperative Extension Service, College of Agriculture and Natural Resources.

Holt, H.A.; Fischer, B.C. 1979. Regenerating oaks in upland hardwood forests. The 1979 John S. Wright forestry conference. W. Lafayette, IN: Purdue University.

Johnson, P.S. 1993. Perspectives on the ecology and silviculture of oak-dominated forests in the Central and Eastern States. Gen. Tech. Rep. NC-153. U.S. Department of Agriculture, Forest Service.

Loftis, D.L. 1990. Predicting post-harvest performance of advance red oak reproduction in the Southern Appalachians. Forest Science. 36: 908-916.

Loftis, D.L.; McGee, C.E. 1993. Oak regeneration: serious problems, practical recommendations. Symposium proceedings. Gen. Tech. Rep. SE-84. U.S. Department of Agriculture, Forest Service.

Lorimer, C.G. 1989. The oak regeneration problem: new evidence on causes and possible solutions. Forest Resour. Analysis 8. Madison, WI: University of Wisconsin.

Martin, A.J.; Hix, D.M. 1988. Regeneration development in an upland hardwood stand following a shelterwood harvest. Northern Journal of Applied Forestry. 5: 46-49.

Neter, J.; Wasserman, W. 1974. Applied linear statistical models. Homewood, IL: Richard D. Irwin, Inc. 842 p.

Oliver, B.C.; Larson, C.D. 1990. Forest stand dynamics. New York: McGraw-Hill.

Powell, D.S.; Faulkner, J.L.; Darr, D.R. [and others]. 1993. Forest resources of the United States, 1992. Gen. Tech. Rep. RM-234. U.S. Department of Agriculture, Forest Service.

Roach, B.A.; Gingrich, S.F. 1968. Even-aged silviculture for upland central hardwoods. Agric. Handb. 355. U.S. Department of Agriculture, Forest Service.

Sander, I.L. 1977. Manager's handbook for oaks in the North Central States. Gen. Tech. Rep. NC-37. U.S. Department of Agriculture, Forest Service.

Sander, I.L.; Graney, D.L. 1993. Regenerating oaks in the Central States. In: Oak regeneration: serious problems, practical recommendations. Gen. Tech. Rep. SE-84. U.S. Department of Agriculture, Forest Service: 174-183

Sander, I.L.; Johnson, P.S.; Watt, R.F. 1976. A guide for evaluating the adequacy of oak advance reproduction. Gen. Tech. Rep. NC-23. U.S. Department of Agriculture, Forest Service.

Smith, H.C. 1993. Regenerating oaks in the central Appalachians. In: Oak regeneration: serious problems, practical recommendations. Gen. Tech. Rep. SE-84. U.S. Department of Agriculture, Forest Service: 211-221.

Waddell, K.L.; Oswald, D.D.; Powell, D.S. 1989. Forest statistics of the United States, 1987. Resour. Bull. PNW-RB-168. U.S. Department of Agriculture, Forest Service.

Ward, J.S.; Stephens, G.R. 1993. Influence of crown class, diameter, and sprout rank on red maple (*Acer rubrum* L.) development during forest succession in Connecticut. In: Proceedings, 9th central hardwood forestry conference. Gen. Tech. Bull. NC-161. U.S. Department of Agriculture, Forest Service: 342-352.

Ward, J.S.; Stephens, G.R. 1996. Influence of crown class and survival and development of *Betula lenta* in Connecticut, USA. Canadian Journal of Forest Research 26: 277-288.

METHODS TO IMPROVE ESTABLISHMENT AND GROWTH OF BOTTOMLAND HARDWOOD ARTIFICIAL REGENERATION

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Abstract—With ongoing attempts to reforest both cut-over and abandoned agricultural land in the lower Mississippi alluvial plain, it has become evident that there exists a need for an efficient regeneration system that makes biological and economic sense. Also, there is a need to address how to minimize competition from invading weeds, to deter predation by small mammals, and to achieve adequate tree establishment. This study was designed as a randomized complete block experiment with treatments arranged as factors (3 species X 2 levels of protection X 4 weed control treatments) with three replications, to assess efficacy of seedling protection and weed control to improve seedling growth and survival. The study was conducted on a cleared area in the Delta Experimental Forest, Stoneville, MS. Three tree species, Nuttall oak (*Quercus nuttallii* Palmer), green ash (*Fraxinus pennsylvanica* Marsh.), and persimmon (*Diospyros virginia* L.) were planted as 1-0 bareroot seedlings in March 1997. Each treatment plot had 25 seedlings, spaced at 0.75 meters X 0.75 meters. Shelter protection was installed on half of the seedlings. Shelters were 1 meter tall, 15 centimeter diameter plastic tree shelters. Each shelter treatment (with or without shelter) received one of four weed control treatments: mechanical mowing (gas-powered weed cutter), fabric mat (woven, black polypropylene material), chemical herbicide (Oust, sulfometuron-methyl at 210 grams per hectare), or undisturbed (control). Response of shelters and weed control treatments on seedling survival, height and diameter were followed for one growing season. Seedlings in shelters had greater survival (98 percent) than seedlings without shelters (93 percent). For all three species, height growth was significantly greater for sheltered seedlings (43 centimeters) compared to unsheltered seedlings (15 centimeters). For the unsheltered seedlings, fabric mat weed control increased survival relative to chemical weed control. All seedlings had significantly greater height and diameter growth under the fabric mat weed control compared to growth under the other treatments except for unsheltered oak seedlings.

INTRODUCTION

In the lower Mississippi alluvial plain (LMAP), some land cleared for soybean production is being converted back to bottomland hardwood forests under the Wetland Reserve Program (WRP). Large-scale reforestation of former agricultural lands faces many challenges. Newly planted trees are subjected to harsh site conditions, including heavy clay soils, herbaceous competition, herbivory, drought and flooding. Proper matching of species to soil and site conditions is necessary to successfully establish seedlings. Species commonly used in reforestation under WRP include Nuttall oak (*Quercus nuttallii* Palmer), willow oak (*Quercus phellos* L.), water oak (*Quercus nigra* L.), green ash (*Fraxinus pennsylvanica* Marsh.), cottonwood (*Populus deltoides* Marsh.) and persimmon (*Diospyros virginia* L.). Natural invasion on these sites is usually minimal, as few seed sources exist (Allen 1990). Those species that do invade usually include sweetgum (*Liquidambar styraciflua* L.), green ash and sugarberry (*Celtis laevigata* Willd.). Therefore, the most common reforestation strategy in the LMAP is to introduce hardmast species and rely on wind and water dispersal of light seeded species. With this strategy, fields may or may not be prepared by disking, oaks are established by planting 1-0 bareroot seedlings or direct seeding, and post-planting weed control is typically not used (Haynes and others 1995, Stanturf and others 1998).

Hardwoods have been established successfully on many sites. Krinard and Kennedy (1987) observed 69-97 percent survival rates two years after planting hardwood seedlings on Sharkey clay soil. In their study, Nuttall oak seedling survival averaged 85 percent. Wittwer (1991) recorded a survival rate of 78 percent three years after planting bottomland oaks in eastern Oklahoma. Savage and others (1989) reported a 64 percent survival rate for seedlings on reforested bottomlands in Louisiana, while Schweitzer and others (1997) reported 63 percent survival one year after planting oak seedlings on a former farmed wetland dominated by heavy clay soils. Despite these successes, operational reforestation under WRP has proven difficult. A recent survey of reforested former agricultural lands in west-central Mississippi enrolled in the WRP found that only 23 percent of the land planted with 1-0 bareroot seedlings had at least 100 trees per acre after three growing seasons (Personal communication. 1998. Callie Jo Schweitzer, Research Forest Ecologist, Southern Research Station, P.O. Box 227, Stoneville, MS 38776). The higher survival reported in the studies cited above were on smaller tracts, while the average tract size in the 1992 WRP survey was approximately 210 acres. Nevertheless, Allen (1990) evaluated oak plantations established by USDI Fish & Wildlife Service personnel on refuges in west-central Mississippi. Similar establishment techniques were used on these tracts as the WRP tracts. Seven out of ten stands Allen assessed had over 200 trees per acre.

In addition to animal browsing stress, seedlings must also compete with invading weeds. In areas where climatic

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conditions tend toward droughty, weeds compete for available water. A fabric weed barrier, more commonly used in western states for shelterbelt establishment, may aid in short-term moisture retention while mitigating the effects of herbaceous competition. However, damage from deer and small rodents may still pose a threat to the newly planted seedlings.

The benefits of tree shelters have been well documented, mostly in cut-over sites in more northern climates. Seedling survival is increased by minimizing losses from animal damage and by stimulating early height growth that can result in earlier crown closure (Strobi and Wagner 1995). Tree shelters may increase the competitiveness of slower growing, desirable species on bottomland sites.

We investigated the impact of tree shelters and four different weed control treatments on the survival and growth of three bottomland hardwood species. Our objective was to evaluate whether weed control, with or without tree shelters, could increase survival and growth of bottomland species.

METHODS

The study was conducted on the Delta Experimental Forest, approximately 7.24 kilometers north of Stoneville, MS. The site is dominated by Alligator clay soils (very fine, montmorillonitic, acid, thermic, Vertic Haplaquepts). Alligator soils are poorly drained and developed in sediments deposited by the Mississippi River. These montmorillonitic clays have high shrink-swell capacity and are common in bottomland forests and land offered for reforestation in this area. In Delta Experimental Forest, an area of approximately 1.62 hectares was cleared in 1967 and has been maintained in grass by bushhogging. The study area was surveyed and prepared for planting by double disking in the fall of 1996.

In February 1997, three blocks of 24 plots were delineated on the study site. Each treatment plot (4.27 X 4.27 meters) contained 25 planting spots equally spaced at 0.75 X 0.75 meters, and plots were surrounded by a 0.61 meter buffer. Treatment plots were marked with wooden corner stakes and all planting spots were marked with a flag prior to planting and weed control treatment. Seedlings were hand planted using planting shovels on March 12, 1997. Nuttall oak, green ash and persimmon were chosen to study because of their compatability on bottomland sites, and their widespread use in reforestation efforts in the LMAP. The bareroot stock (1-0) used in this experiment was purchased from a local nursery.

Shelter protection was installed on half of the seedlings immediately after planting. Shelters were 1 meter tall, 15 centimeter diameter, commercially available opaque, plastic tree shelters, each secured with plastic ties attached to a wooden stake. Seedlings also received one of four weed control treatments: mechanical, fabric mat, herbicide or none. Mechanical mowing was implemented manually using a gas-powered weed cutter, applied every other week from March until December 1997. The chemical herbicide Oust (sulfometuron-methyl) was applied using a back-pack sprayer on March 21, 1997 at 210 grams per hectare. The fabric mat (woven, black polypropylene material) was placed on plots in 0.91 meter wide strips and secured in place using ground staples. Fabric mat strips overlapped slightly at each seedling row, allowing for the seedling (and seedlings in shelters) to be completely surrounded by mat. No weed control was performed on the control or undisturbed plots after the initial disking for site preparation.

All seedlings were measured immediately after planting and again in January 1998. Height was measured with a meter stick to the nearest centimeter, and diameter was measured with a digital caliper to the nearest millimeter. In July 1997, square meter plots for vegetation sampling were placed in three plots for each weed control treatment. A seedling in the selected plot was randomly selected, and the square meter frame was place next to that seedling. All vegetation was removed, dried and weighed.

Data were analyzed using SAS software (SAS Institute 1988) according to a randomized complete block design with three replications and a factorial treatment arrangement. First-year height and diameter growth were computed by subtracting initial values from final values. Survival rates in decimal fractions were transformed with inverse sine transformation as they covered a wide range of values (Steel and Torrie 1980). Analysis of variance (ANOVA) and Duncan's new multiple range test were used to test for significant differences among treatment means. One-way ANOVA was used to compare survival and growth of seedlings with and without shelters. A three-way ANOVA was used to analyze the effects of weed control, species and blocks on variation in mean seedling growth and survival. Significant differences were reported at the 0.05 percent level.

RESULTS

Seedling Survival and Growth

Survival of all species was exceptionally high. Of the total 1800 seedlings planted, only 81 seedlings died. Ash seedlings had significantly higher survival (588/600, 98 percent) than persimmon (550/600, 92 percent), while oak survival (581/600, 96.8 percent) did not differ from that of ash or persimmon. Unsheltered seedlings showed significantly lower survival (841/900, 93 percent) than sheltered seedlings (878/900, 97.6 percent).

Unsheltered ash seedlings had significantly greater survival than persimmon; oak survival was not significantly different than ash or persimmon (Figure 1). Sheltered oak and ash seedlings had significantly greater survival than sheltered persimmon seedlings (Figure 1).

Sheltered seedlings grew significantly taller than seedlings without shelters (height growth with shelter=43.3 centimeters, n=878, without shelter=14.9 centimeters, n=841). However, diameter growth of unsheltered seedlings was greater than diameter growth of sheltered seedlings (diameter growth without shelter=3.9 millimeters, n=841, with shelter=3.3 millimeters, n=878).

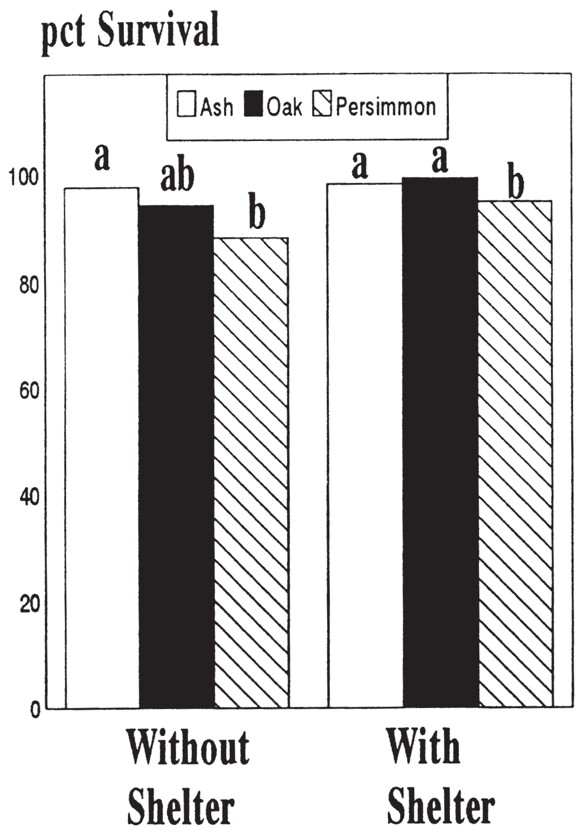

Figure 1—Percent survival comparisons among three bottomland hardwood species with tree shelters and without tree shelters. Means in each shelter grouping followed by different letters are significantly different at the 0.05 level.

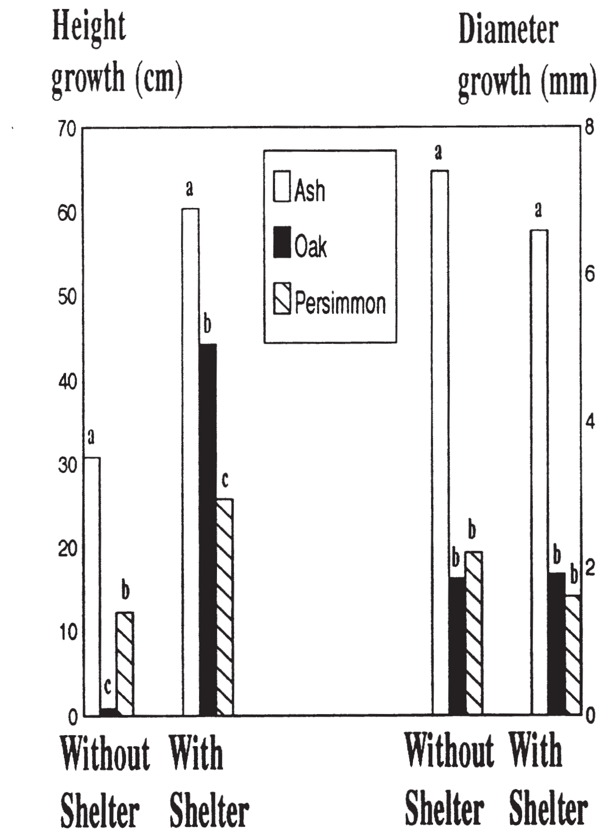

Figure 2—First year height and diameter growth (initial minus final) for three bottomland hardwood species. Means in each shelter grouping followed by different letters are significantly different at the 0.05 level.

Height growth was significantly different among all three unsheltered species, with ash > persimmon > oak (Figure 2). Unsheltered ash seedlings also had significantly greater diameter growth compared to persimmon and oak (Figure 2).

Oak seedlings showed favorable growth in shelters, increasing height growth by 5400 percent. Ash seedlings still grew the tallest, showing 96 percent increase with shelters. For seedlings with shelters, ash diameter growth was significantly greater than oak or persimmon (Figure 2).

However, height growth was significantly greater for sheltered seedlings compared to unsheltered seedlings for each species, under all four weed control treatments. In general, diameter growth was greater for unsheltered seedlings across all weed control treatments. Under the herbicide treatment, unsheltered ash, oak and persimmon seedlings had significantly greater diameter growth than sheltered. Unsheltered ash and persimmon under mechanical weed control also had significantly greater diameter growth than sheltered, and unsheltered persimmon under no weed control had significantly greater diameter growth compared to sheltered persimmon under no weed control.

Weed Control Treatment

For all seedlings without shelters, those under fabric mat weed control had significantly greater survival than herbicide treatment. Survival of seedlings in the mechanical and no weed control treatments was not significantly different from those in the fabric mat treatment (Figure 3). For any one species without shelters, survival was not significantly different among weed control treatments (Figure 4). Sheltered seedlings under both fabric mat and no weed control treatments had significantly greater survival than seedlings treated with herbicide (Figure 3). Results were mixed for survival of sheltered seedlings, by species and weed control combinations. Sheltered seedlings in the mechanical weed control treatment had survival rates that were not significant from the other three treatments (Figure 3). Sheltered ash seedlings under herbicide treatment had significantly lower survival than those in mechanical, fabric mat or no weed control treatments (Figure 5). There were no significant differences in survival among the four weed control treatments for sheltered oak seedlings. For sheltered persimmon, fabric mat treatment resulted in significantly greater survival than mechanical or herbicide; survival under no weed control treatment did not vary significantly from any of the other treatments.

Seedlings with shelters and without shelters had significantly greater height and diameter growth under the fabric mat weed control treatment except for unsheltered oak seedlings (Table 1). Unsheltered oaks with no weed control had significantly greater height growth than those seedlings grown with fabric mat as weed control. Height

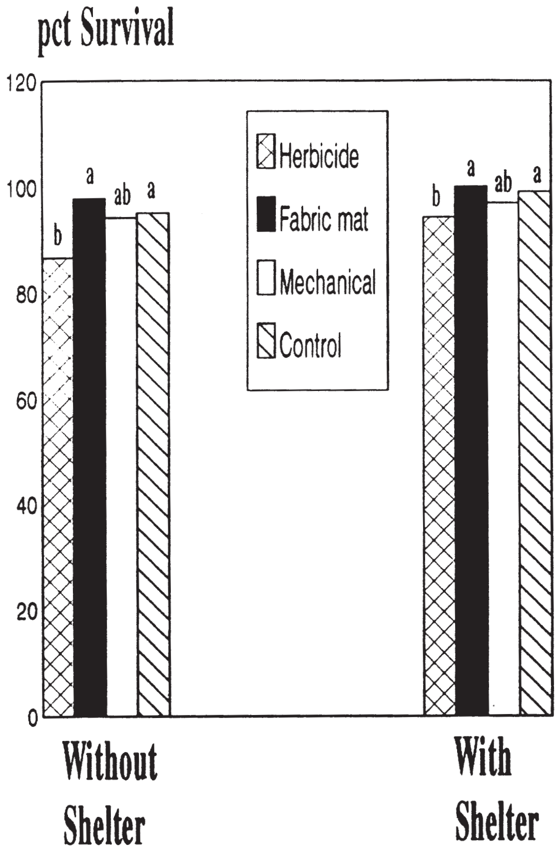

Figure 3—First year survival of all seedlings under four weed control treatments. Means in each shelter grouping followed by different letters are significantly different at the 0.05 level.

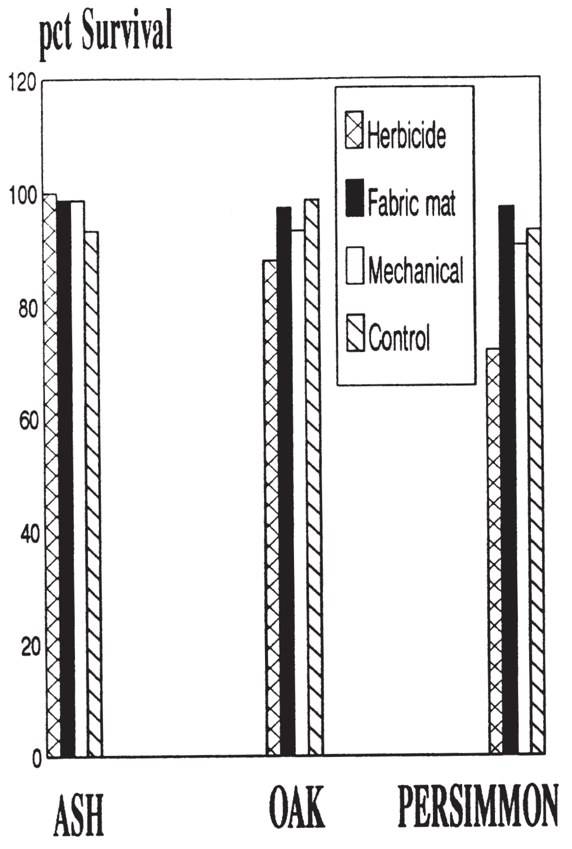

Figure 4—First year survival of three bottomland hardwood species without tree shelters. There were no significant differences in survival means among the four weed control treatments.

Table 1—Results of weed control treatment on height and diameter growth (final - initial) of ash, oak and persimmon seedlings, with and without shelters. Different letters within a row indicate significant difference at the 0.05 level. (Negative values indicate dieback and resprouting)

		Herb	Mat	Mech.	None
		------------	*Cm*	------------	
Height					
Ash	With shelter	38.6c	95.3a	55.4b	50.3b
Ash	W/out shelter	23.7c	43.1a	33.7b	22.2c
Oak	With shelter	13.4c	42.1a	23.5b	23.9b
Oak	W/out shelter	1.0ab	-1.3b	0.2ab	3.4a
Per.	With shelter	24.3c	59.7a	46.7b	44.2b
Per.	W/out shelter	3.5c	20.3a	11.3b	11.5b
Diameter					
Ash	With shelter	3.8c	12.1a	4.7b	5.2b
Ash	W/out shelter	5.9c	10.9a	7.3b	5.4c
Oak	With shelter	1.1c	3.1a	1.9b	1.7b
Oak	W/out shelter	1.8a	1.7a	2.1a	1.8a
Per.	With shelter	1.2b	3.0a	1.0b	1.3b
Per.	W/out shelter	1.7b	3.3a	1.8b	1.9

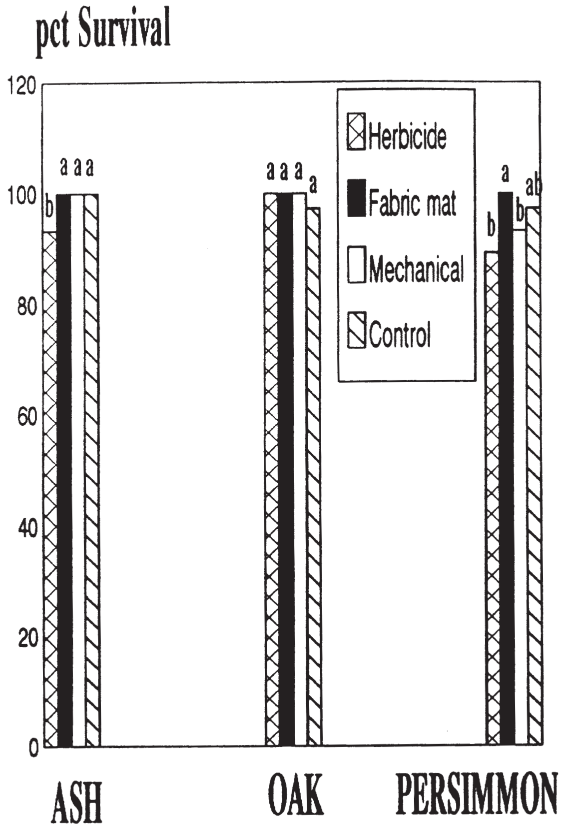

Figure 5—First year survival of three bottomland hardwood species with tree shelters. Means in each species grouping followed by different letters are significantly different at the 0.05 level.

growth for seedlings in the herbicide and mechanical treatments did not differ from either no weed control or fabric mat treatment. Unsheltered oak diameter growth was not different among the four weed control treatments. The herbicide treatment resulted in the smallest height and diameter growth rates, compared to fabric mat, no weed control, and mechanical treatments. This was significant across all species, for seedlings with and without shelters, except for persimmon seedlings with shelters and oak seedlings without shelters. Diameter growth of persimmon seedlings, with shelters, in the herbicide treatment differed significantly only from the fabric mat treatment. Oak seedlings without shelters had the greatest height growth in the no weed control treatment, which was significantly greater than height growth in the fabric mat. There was no difference in diameter growth among the four weed control treatments for oak seedlings without shelters.

DISCUSSION

Survival of all seedlings was high. Successful establishment may be attributed to proper species selection for the study site, adequate site preparation, and quality control on seedling handling and planting.

Shelter Effects

Seedlings with shelters had significantly higher survival than seedlings without shelters. In concurrence with literature reports (Tuley 1983, Frearson and Weiss 1987, Lantagne and others 1990) shelters increased survival of the three species we studied.

In addition to improving survival, shelters also enhanced seedling height growth. However, diameter growth was depressed by the shelters. Ponder (1995) reviewed several studies which showed tree shelters accelerated height growth of young trees. These studies have shown that it is not unusual for diameter growth to be less with shelters than without them during the first year or two of growth. Gillespie and other (1996) observed etiolation of sheltered trees. They noted that shifts in light quality or quantity may contribute to this, as well as stems allocating more carbon to height growth and less to caliper or diameter growth with mechanical support coming from the shelter. Lantagne and others (1990) suggested that the modified growing environment created by shelters results in the reallocation of growth from roots, stem diameter and branches to the main stem. The effect of shelters on root growth was not examined in this study. Many of the seedlings in the present study have emerged from the shelters with good apical dominance which should increase their chance of growing into dominant and codominant crown positions.

Weed Control

The fabric mat used to control weeds had a significant effect on the survival and growth of the seedlings. In reforestation, mulch mats appear to suppress competing vegetation primarily by blocking light necessary for photosynthesis, and to a lesser extent, by mechanically impeding growth (Clarkson and Frazier 1957). To control weeds effectively, mulches must be applied early, remain intact, and be large enough. The fabric mulch mat used in this study was applied prior to weed establishment on a disked field. We did encounter some difficulties installing the mat. Ground staples were used to secure the corners, and additional staples had to be placed 0.5 m apart to hold the mat in place. In addition to the staples, we also placed clay pots on all the corners to aid in keeping the mat from blowing around. Installment was difficult in the heavy clay soil, as the mat became stuck with mud before it could be spread out evenly. The cost of the mat may also prohibit its use in large-scale reforestation efforts.

The fabric mat treatments had significantly lower weed biomass compared to the other weed control treatments (Table 2). Although the herbicide treatment reduced weeds compared to no weed control treatment, height and diameters of all seedlings studied were reduced compared to the other three weed control treatments. Kennedy (1981) used disking, mowing and no weed control on several bottomland species to examine survival and growth under different cultural treatments. After four growing seasons, he found that there was no difference in seedling survival, height or diameter for seedlings that were mowed versus those in the control. Disking did improve survival and growth. Kennedy (1981) concluded that competition still exists whether weeds are allowed to grow and cut back by mowing or to grow continuously as in the control, therefore mowing was not an acceptable substitute for disking.

Table 2—Total weed biomass collected from three plots in each weed control treatment. Different letters within weight column indicates significant difference at 0.05

Weed control treatment	Total weight	n	Dominant vegetation
	G		
None	605.38a	9	Bundle weed[a], vines, grass
Herb	404.42b	9	Bundle weed
Mech.	298.32b	9	Grass
Mat	167.83c	9	Grass, bundle weed

Means in each shelter grouping followed by different letters are significantly different at the 0.05 level.
[a] Bundle weed (*Desmanthus illinoensis* Michaux).

CONCLUSIONS

The results of this study indicate that a potential exists for the use of tree shelters in bottomland hardwood reforestation. Herbivory was not a major stress in this study, although browsing damage has been reported by others who have artificially reforested in this area. Therefore, protection benefits of tree shelters need to be more thoroughly tested under heavy animal pressure. The accelerated height growth of sheltered seedlings after one year compared to the unsheltered seedlings showed promise that those seedlings will rise above the competing vegetation and herbivores. This increase in height growth may also be beneficial in areas that receive late season flooding, common on bottomland sites. Seedlings that are above the flood water levels have a greater chance of survival.

It is premature to prescribe a method for regenerating bottomland hardwood species based on the first growing season. Based on first growing season results, however, data from this study show that an application of sulfometuron-methyl herbicide reduced the amount of competing vegetation, but appeared to suppress seedling survival and growth. Best survival and growth, and the lowest weed invasion, was found with the fabric mat. Although both the shelters and the fabric mat gave positive survival and growth results, cost may be a prohibitive factor in their large-scale use.

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REFERENCES

Allen, J.A. 1990. Establishment and growth of bottomland oak plantations on the Yazoo National Wildlife Refuge Complex. Southern Journal of Applied Forestry. 14(4): 206-210.

Clarkson, V.A.; Frazier, W.A. 1957. Plastic mulches for horticultural crops. Station Bull. 562. Corvallis, OR: Oregon State College, Agriculture Experiment Station. 12 p.

Frearson, K; Weiss, N.D. 1987. Improved growth rates within tree shelters. Quarterly Journal of Forestry. 81(3): 184-187.

Gillespie, A.R.; Rathfon, R.; Myers, R.K. 1996. Rehabilitating a young northern red oak planting with tree shelters. Northern Journal of Applied Forestry. 13(1): 24-39.

Haynes, R.J.; Bridges, R.J.; Gard, S.W. [and others]. 1995. Bottomland forest reestablishment efforts of the U.S. Fish & Wildlife Service: Southeast Region. In: Fischenich, J.C.; Lloyd, C.W.; Palermo, M.R., eds. Proceedings national wetlands engineering workshop. Tech. Rep. WPR-RE-8. Vicksburg, MS: Waterways Experiment Station: 322-334.

Kennedy, H.E. 1981. Foliar nutrient concentrations and hardwood growth influenced by cultural treatments. Plant and Soil. 63: 307-316.

Krinard, R.M.; Kennedy, H.E. 1987. Fifteen-year growth of six planted hardwood species on Sharkey clay soil. For. Res. Note SO-336. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 4 p.

Lantagne, D.O.; Ramm, C.W.; Dickman, D.F. 1990. Tree shelters increase heights of planted oaks in a Michigan clearcut. Northern Journal of Applied Forestry. 7: 24-26.

Ponder, F. 1995. Shoot and root growth of northern red oak planted in forest openings and protected by treeshelters. Northern Journal of Applied Forestry. 12(1): 36-42.

SAS Institute. 1988. PC-SAS user's guide. Statistics. Version 6.04 ed. Cary, NC: SAS Institute.

Savage, L.; Pritchett, D.W.; DePoe, C.E. 1989. Reforestation of a cleared bottomland hardwood area in northeast Louisiana. Restoration and Management Notes. 7(2): 88.

Schweitzer, C.J.; Stanturf, J.A.; Shepard, J.P. [and others]. 1997. Large-scale comparison of reforestation techniques common to the Lower Mississippi Alluvial Valley: first-year results. In: Pallardy, S.G.; Cecich, R.A.; Garrett, H.G.; Johnson, R.S., eds. Proceedings, 11th central hardwood forest conference. Gen. Tech. Rep. NC-188. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 410 p.

Stanturf, J.A.; Schweitzer, C.J.; Gardiner, E.S. 1998. Afforestation of marginal agricultural land in the Lower Mississippi River Alluvial Valley, U.S.A. Silva Fennica. 32(2): 281-297.

Steel, R.G.D.; Torrie, J.H. 1980. Principles and procedures of statistics: a biometrical approach. New York: McGraw-Hill. 633 p.

Strobi, S.; Wagner, R.G. 1995. Early results with translucent tree shelters in southern Ontario. In: Brissette, J.C., ed. Proceedings of the tree shelter conference; 1995 June 20-22; Harrisburg, PA. Gen. Tech. Rep. NE-122. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 13-18.

Tuley, G. 1983. Shelters improve the growth of young trees in forest. Quarterly Journal of Forestry. 77: 77-87.

Wittwer, R.F. 1991. Direct-seeding of bottomland oaks in Oklahoma. Southern Journal of Applied Forestry. 15(1): 17-22.

THINNING EFFECTS ON BASAL AREA GROWTH OF RED MAPLE (*ACER RUBRUM* L.)

H. Alexis Londo, Terry R. Strong, Helga Soares, David D. Reed[1]

Abstract—Red maple (*Acer rubrum* L.) is one of the most abundant and wide spread tree species in North America. Due to the characteristic poor form and wood quality of red maple, it is often categorized as a soft maple and most commonly merchandised as pulp. Because of the increasing interest and occurrence of red maple, a study analysis was conducted to test several silvicultural thinning treatments on red maple growth in even-aged stands located in Upper Michigan and northern Wisconsin by examining stand dynamics over a 14 year study period.

The effects of the silvicultural treatments of 40, 60, 80, 100, 120, and 140 ft^2 residual basal area per acre respectively, were examined. Plots were remeasured every 5 years. The increase in stand parameters such as basal area, average percent DBH (diameter at breast height), stand density and stand volume were calculated and analyzed. Diameter and volume growth of the 40 ft^2 were shown to increase the most over time.

INTRODUCTION

Red maple (*Acer rubrum* L.) is one of the most abundant and wide spread tree species in North America, with a contiguous range from Newfoundland to the southern tip of Florida, extending to southeastern Manitoba and East Texas (Burns and Honkala 1990). This species thrives on a wide range of sites and is becoming evermore present as a primary producer in forest communities (Crow 1978). Red maple follows only sugar maple (*Acer saccharum* Marsh.) in terms of growing stock volume on Michigan's timberland, with an annual net increase of over 114 million ft^3 of growing stock in Michigan alone (Leatherberry and Spencer 1996). Red maple has become an important resource on over 1 million acres in Upper Michigan and Northern Wisconsin (Erdmann and others 1981).

Red Maple is often categorized as a soft maple due to its characteristically poor form and wood quality and is most commonly merchandised as pulp for the production of high quality paper products. On good sites, red maple may grow fast with good form and quality, producing a better grade of saw logs. Good quality red maple may be substituted for hard maple in furniture production, thereby raising its value.

Stumpage prices since 1971 for red maple have increased at a higher rate than those of sugar maple (Niese and others 1995). Management efforts for this species, however, are tenuous due to the lack of growth and yield information for even-aged management (Erdmann and others 1981). The objective of this study is to analyze the effectiveness of several levels of residual basal area regimes in red maple stands located in Upper Michigan and northern Wisconsin by examining stand dynamics over a 14 year study.

METHODOLOGY

Permanent study sites were established in fully stocked, even-aged, undisturbed stands growing under uniform conditions with 75 percent or more of the overstory composition in red maple (Erdmann and others 1981). These sites were chosen in even-aged red maple stands on good, medium and poor sites in northern Wisconsin and Michigan. Two or more plots, each consisting of two 1/4 acre (1000 m^2) subplots, were established at each study site. Trees greater than 4.0 in. (10 cm) were numbered, species recorded and dbh measured in 1980. After the growing season of 1981 and before the growing season of 1982, plots at each of the study sites received one of seven thinning treatments: no thinning, or reducing basal area to 40, 60, 80, 100, 120, and 140 ft^2/ac.

Plots were thinned from below to the desired basal area to create a stand of uniformly spaced, defect free, dominant and codominant red maple trees. Smaller suppressed and intermediate red maple trees were removed. Pole size species other than red maple trees were removed to increase the stocking of red maple. Stocking levels were then checked and corrections were made where possible.

Remeasurements of the study sites were conducted in 1985, 1990, and 1996. Observations due to ingrowth, defined as trees reaching 4.0 inches dbh after establishment, were added and remeasured during subsequent measurement periods. To create a post thinning profile of the sites, the 1980 data set was used excluding any ingrowth trees and those recorded as cut in the 1985 measurement data.

Basal area calculations were made using the dbh measurements for each tree and these calculations were summarized by species for each plot. Total stem outside

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bark volume calculations for red maple trees were made using the equation of Crow and Erdman 1984. All calculations were then averaged for plots contained in each basal area category.

RESULTS AND DISCUSSION

A total of 53 sites were selected and 226 plots established. Each treatment contained from 6 to 66 plots (table 1). Initial basal area measurements varied from 65.4 ft^2/ac to 238.1 ft^2/ac over all the plots with an average of 136.8 ft^2/ac. In 1982, basal area values decreased due to the application of the thinning treatments. Using a F-test of the general linear hypothesis, thinnings were determined to have an affect on the basal area of the stands ($p < 0.05$) over all the subsequent measurement dates. By 1996, basal area measurements for sites thinned to 40 ft^2/ac and the control sites were the only treatments to surpass their initial (1980) basal area measurements. All other sites increased for each of the remeasurements (fig. 1). A Student-Newman-Kuels (SNK) test of the means (Montgomery 1984) indicated that, for each of the measurement cycles, thinning operations produced different basal area values from the control and each other ($p > 0.05$), except for the 140 ft^2/ac thinning treatment which did not differ from the control ($p > 0.05$). The ordering of the means remains constant for all the measurement cycles until 1996, with basal area values being statistically different ($p > 0.05$) and ordered by thinning intensity. There was an interaction between thinning treatments and time, confirming that the treatments resulted in different basal area growth rates over time. Following the 1996 measurement, plots thinned to 40 ft^2/ac had basal area values larger than those thinned to 60 ft^2/ac ($p > 0.05$).

The average percent dbh growth for each stand following treatment over the study period was computed. These values illustrate the response of dbh of each of the treatments (fig. 2). Plots where basal area was reduced to 40 ft^2/ac and 60 ft^2/ac showed the greatest increases in percent dbh growth. All other treatments exhibited an increase in average percent dbh growth ($p > 0.05$) for each of the measurement cycles. One exception occurred in 1996, when areas thinned to 140 ft^2/ac declined slightly from the 1990 average diameter.

While percent diameter growth increased through time for each of the study sites, trees per acre (TPA), in most cases, did not increase ($p > 0.05$) after the thinning applications were applied in 1982 (Figure 3). Where basal area was reduced to 40 ft^2/ac, the number of trees increased over the subsequent measurement cycles. Plots incurring the intermediate thinning applications, 60 ft^2/ac, 80 ft^2/ac and 100 ft^2/ac, exhibited slight increases in trees per acre in the years following thinning. When light thinnings (120 ft^2/ac and 140 ft^2/ac) were applied, trees per acre decreased slightly by the last measurement year, as was the case for the unthinned (control) plots.

Figures 2 and 3 indicate that thinnings applied to red maple stands increase diameter growth for individual trees. With steady or decreasing trees per acre, the increase in basal area growth is distributed among the individual residual trees. This indicates thinning from below operations in red maple stands increased larger stem size, sawtimber yield. Heavier thinnings (residual ≤ 100 ft^2/ac) result in ingrowth, recruitment of smaller stems into the merchantable size classes over time. The effects of the thinning treatments on volume were investigated (fig. 4). These results when compared with the thinning effects on basal area prove to be very similar. Statistical evaluation of thinning effects on volume production also proved to be analogous to the basal area results.

Table 1—Number of study plots and initial conditions by treatment

Treatment	No. of observations	Initial basal area values: Mean	Minimum	Maximum
		Ft²/ac		
Control	22	141.6	111.3	193.2
40 ft^2/ac	6	93.5	72.5	102.4
60 ft^2/ac	42	119.2	65.4	146.2
80 ft^2/ac	66	136.3	87.7	238.1
100 ft^2/ac	56	144.3	105.8	192.6
120 ft^2/ac	28	149.2	117.4	195.6
140 ft^2/ac	6	163.6	130.0	194.4
Total	226	136.8	65.4	238.1

CONCLUSIONS

Rates of increase in basal area growth are greatest in the most heavily thinned stands. These stands also exhibited the highest gain in trees per acre. Actual growth per tree is minimal in stands with intermediate levels of thinning. All treatments except the 40 ft^2/ac treatment, reduced trees per acre to between 215 and 255 individuals. These stocking densities remained constant with little or no ingrowth for the remainder of the study. Target basal area values for these treatments were from 60 ft^2/ac to 140 ft^2/ac. This implies largest trees occur where BA's are the highest. This is a result of the reduction of stand density from below to attain target basal area.

At last measurement, 14 years after treatment, the basal area growth rates were starting to decrease in treatments with higher target basal areas. Sites reduced to 140 ft^2/ac and 100 ft^2/ac increased very little in basal area from 1990 to 1996. The stands where smaller stand densities of 40 ft^2/ac, 60 ft^2/ac and 80 ft^2/ac were retained, basal area is still increasing at a steady and similar rate over time.

Reducing basal area proves to be effective in increasing the growth rates of red maple, however caution should be taken not to open the site too much. Thinning to 40 ft^2/ac produced improved growth rates, but also dramatically increased trees per acre due to regeneration and

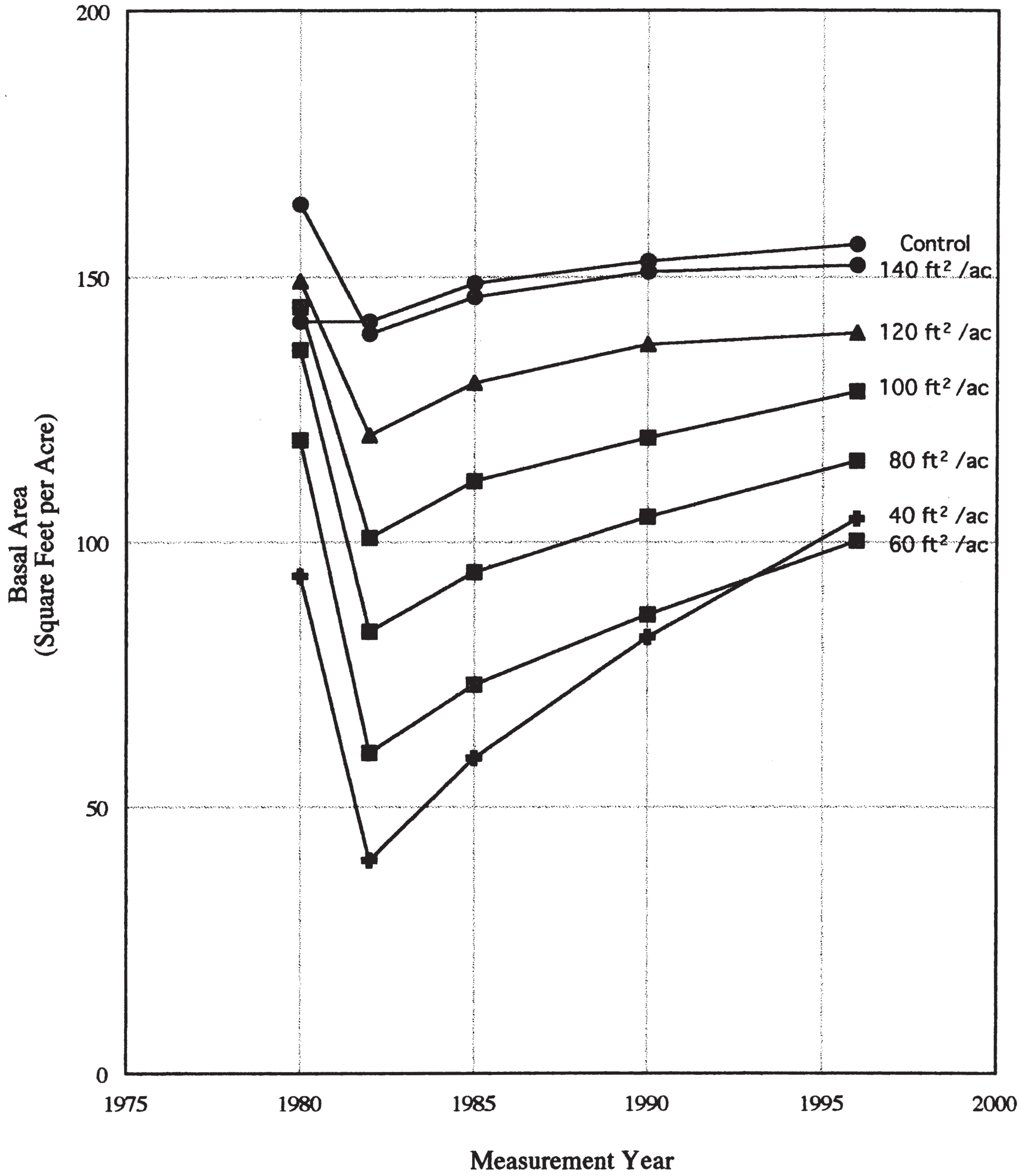

Figure 1—Average basal area (ft^2/ac) given for each thinning treatment and measurement date.

recruitment. Better growth per tree may be attained by reducing trees per acre to between 215 and 235, while keeping basal area between 60 ft^2/ac and 80 ft^2/ac.

These results expand and substantiate the work of Trimble (1974) in which he concluded red maple diameter growth increased after thinning. Consistent with these results, similar work by Voorhis (1990) with yellow poplar (*Liriodendron tulipifera* L.) and paper birch (*Betula papyrifera* Marsh.) showed significant increases in diameter growth after light and heavy thinnings. However, when Voorhis (1990) investigated sugar maple in this same study, diameter growth responses did not increase.

Lamson (1988) investigated cleaning (eliminating trees of similar age but less desirable species or form than that of the target trees) in 7- and 12-year-old stands of basswood (*Tilia americana* L.), red maple, black cherry (*Prunus serotina* Ehrh.) and northern red oak (*Quercus rubra* L.). His research indicated diameter growth increased most

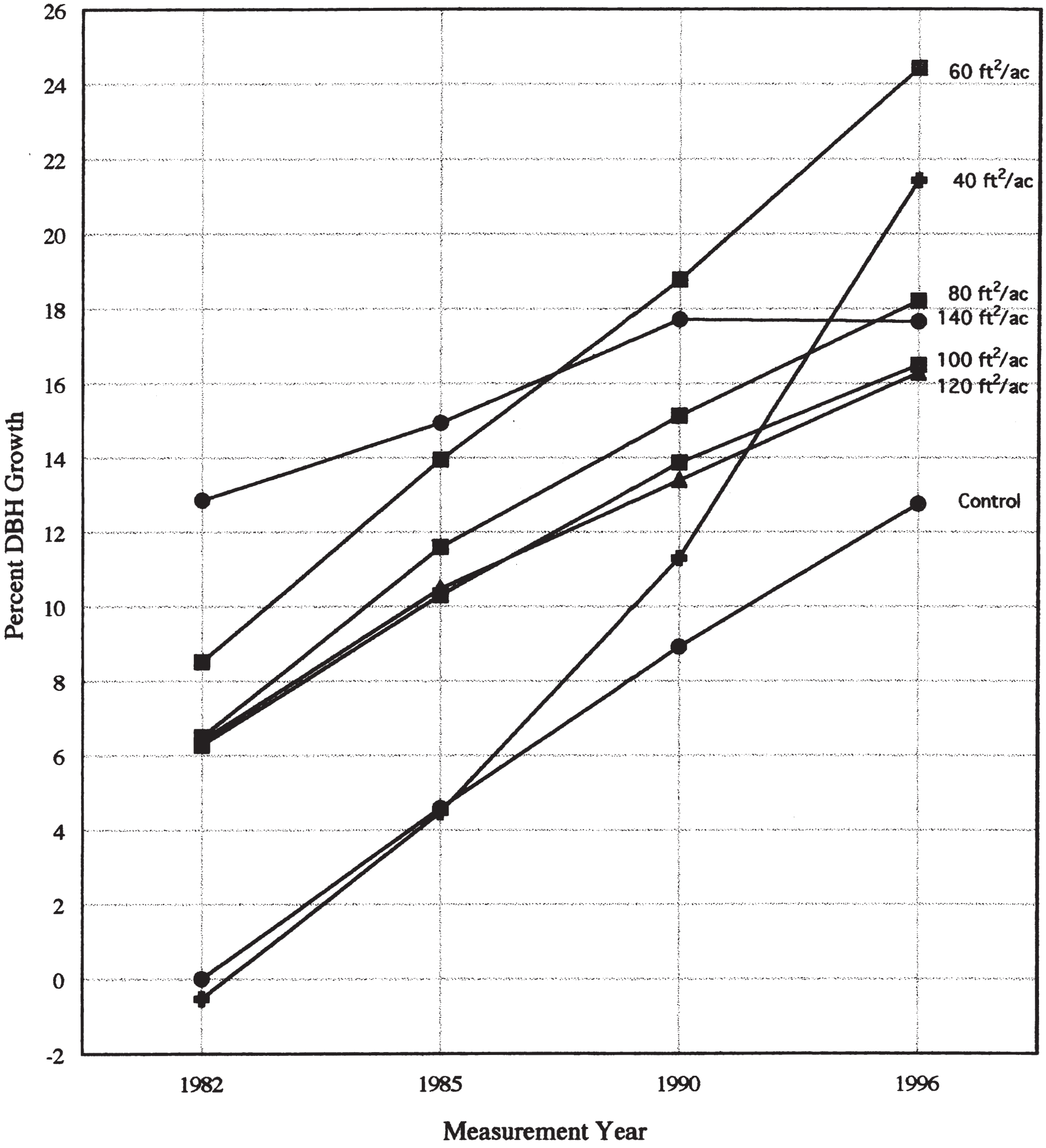

Figure 2—Percent growth of each thinning treatment and measurement cycle.

during the first 5 years after treatment. In this study, this period of increased growth has been extended.

Further research is needed to determine which time is best for these thinnings to occur. The duration of these effects and their influence on the pruning habits, growth patterns and height development of individual trees also needs to be investigated. Finally, research to on the interactions of red maple with the northern hardwood forest species and how relative stand densities influence the volume increment of merchantable wood is another area worthy of research (Nowak 1996).

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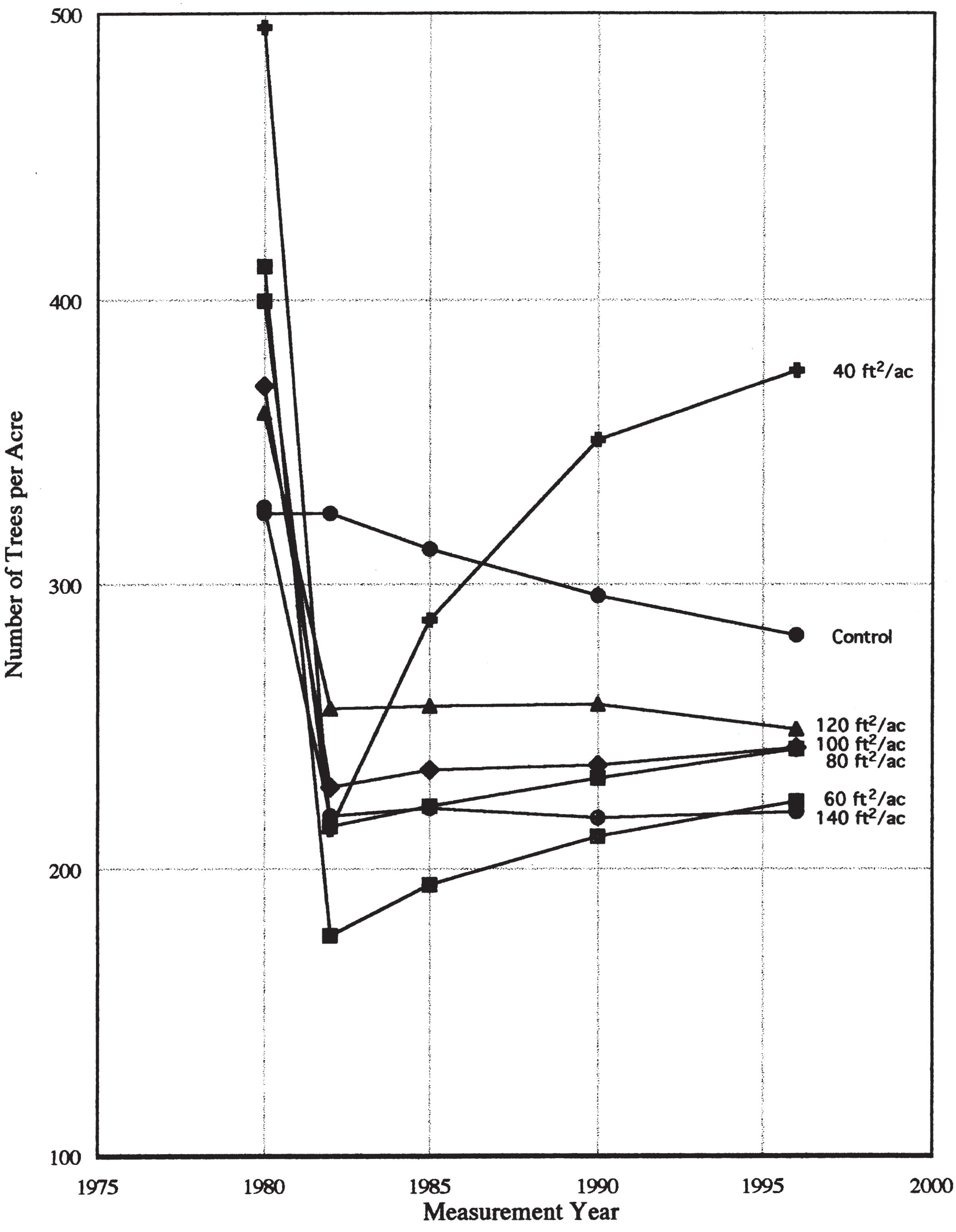

Figure 3—Trees per acre (TPA) of each thinning treatment and measurement cycle.

REFERENCES

Burns, R.M.; Honkala, B.H. 1990. Silvics of North America. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service. 871 p. Vol. 2.

Crow, T.R. 1978. Biomass and production in three contiguous forests in northern Wisconsin. Ecological Society of America. 59(2): 265-273.

Erdmann, G.; Crow, T.; Rauscher, H.R. 1981. Managed even-aged red maple growth and yield study. In: Study proposal from North Central Forest Experiment Station FS-NC-1102-NH-80-02.

Lamson, N.I. 1988. Precommercial thinning and pruning of Appalachian stump sprouts - 10-year results. Southern Journal of Applied Forestry. 12: 23-27.

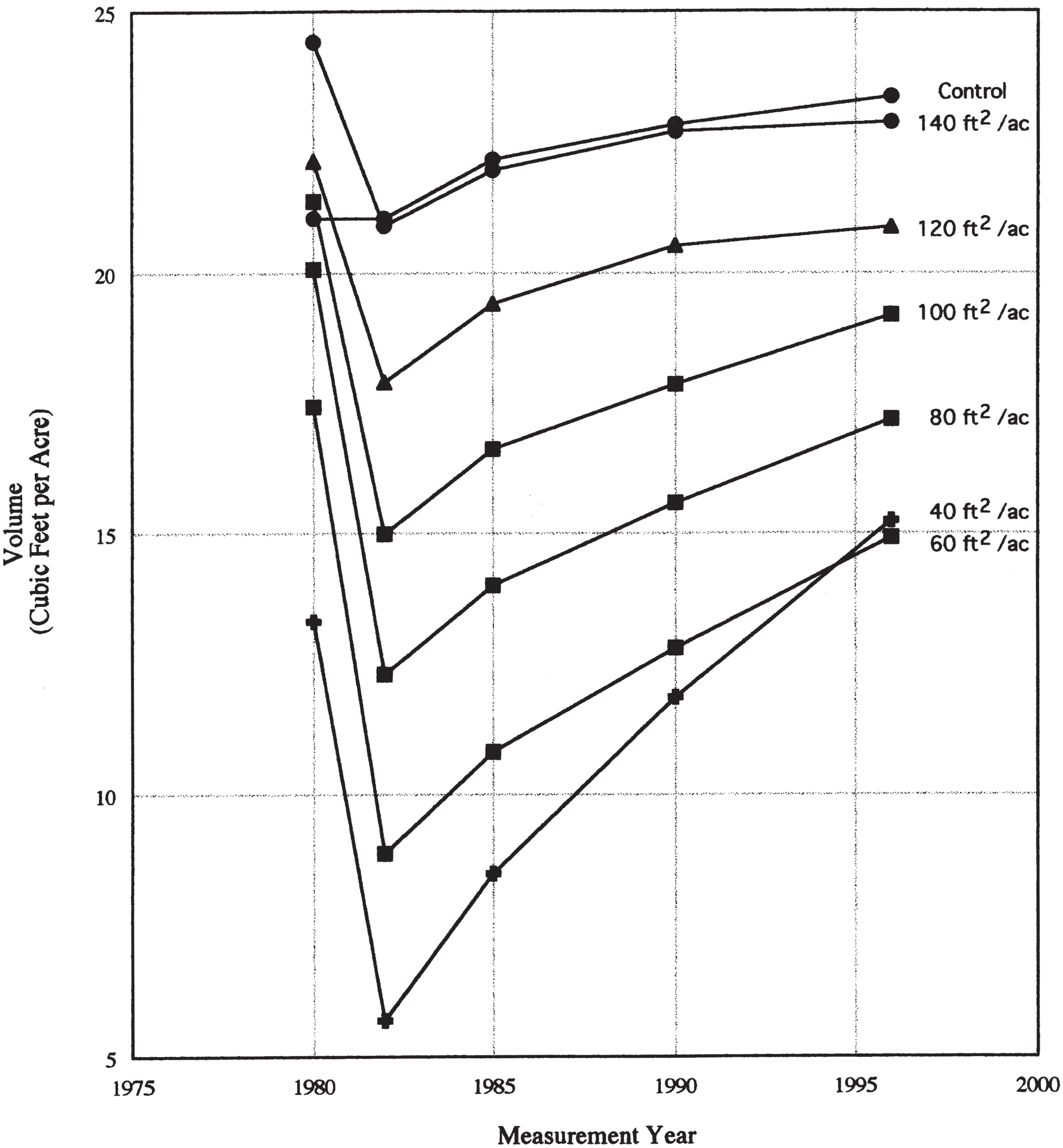

Figure 4—Volume per hectare (ft^2) for each thinning treatment and measurement cycle.

Leatherberry, E.C.; Spence, J.S., Jr. 1996. Michigan forest statistics, 1993. Res. Bull. NC-170. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station.

Montgomery, D.C. 1984. Design and analysis of experiments. New York: John Wiley. 504 p.

Neise, J.M.; Strong, T.F.; Erdmann, G.G. 1995. Forty years of alternative management practices in second-growth, pole-size northern hardwoods. II. Economic evaluation. Canadian Journal of Forest Research. 25: 1180-1188.

Nowak, C.A. 1996. Wood volume increment in thinned 50- to 55-year-old, mixed species Allegheny hardwoods. Canadian Journal of Forest Research. 26: 819-835.

Trimble, G.R. 1974. Response to crop tree release by 7-year-old stems of red maple stump sprouts and northern red oak advance reproduction. Res. Pap. NE-303. U.S. Department of Agriculture, Forest Service.

Voorhis, N.G. 1990. Precommercial crop-tree thinning in a mixed northern hardwood stand. Res. Pap. NE-640. U.S. Department of Agriculture, Forest Service.

RESIDUAL STAND QUALITY FOLLOWING IMPLEMENTATION OF UNEVEN-AGED SILVICULTURE IN EVEN-AGED OAK-HICKORY FORESTS IN THE BOSTON MOUNTAINS OF ARKANSAS

Martin A. Spetich, David L. Graney and Paul A. Murphy[1]

Abstract—A test of group-selection and single-tree selection was installed in 80-year-old even-aged oak-hickory stands in the Boston Mountains of northern Arkansas. Twenty-four 11-acre plots were installed in well stocked stands representing north or east and south or west aspects. Stands between group openings were cut to residual basal areas of 65 and 85 ft^2 per acre using free thinning or structural control. Tree quality in residual stands was evaluated using U.S. Forest Service tree grades for factory lumber and Grosenbaugh tree classes. Trees 11.6 in. and larger in dbh were considered sawtimber and included in the analysis. The effects of density, cutting method, and aspect on tree grade were evaluated using 2,225 sawtimber-sized trees. Results indicate no difference among treatments due to the short time interval since cutting. However, 53 percent of sawtimber either were or have the potential to develop into high quality trees. A residual basal area of 65 ft^2 or less is more likely to effectively increase tree quality and control species composition in the Boston Mountains than an 85 ft^2 target basal area. Overall, this study indicates that there is excellent potential to improve stand tree quality in the Boston Mountains of northern Arkansas using uneven-aged silviculture.

INTRODUCTION

Public concern over the dramatic visual impact of clearcutting has stimulated interest in alternative forest management systems for upland oak forests in the Midsouth. To address these concerns alternatives should avoid the negative visual impacts of clearcutting and must provide the biological conditions necessary for regenerating and maintaining the oaks and other valuable hardwood species. Uneven-aged methods have been suggested as an alternative. Uneven-aged cutting methods are designed to create and maintain at least three age classes within the stand. In single-tree selection, all trees marked for cutting are selected for removal based upon their individual merit. But in group selection, the trees removed in regeneration cuts are selected as a group or aggregate, not as individuals; however, the trees removed between the group openings are selected on individual merit. Group selection can be considered a variant of single-tree selection. For instance, periodic cuts are required in both methods to (1) establish and develop reproduction, (2) improve stand structure and quality, and (3) control residual stocking for sustained yield. They differ in how the periodic cuts are made and their effect on species composition. Recent papers (Miller and others 1995, Murphy and others 1993) have provided an excellent description of both methods and discussed the advantages and disadvantages of each.

Although the feasibility of these uneven-aged cutting methods with upland oak types are being investigated, most research emphasis has been on regeneration and stand structure development (Graney and Murphy 1997, Loewenstein and others 1995). However, no research has concentrated on the long-term effect of the cutting methods on tree quality.

A study has been installed to evaluate effectiveness of group-selection and single-tree selection methods in mature even-aged oak-hickory stands on dry mesic and mesic upland sites in the Boston Mountains of northern Arkansas (Graney and Murphy 1997). The specific objectives are:

(1) To test the feasibility of using group-selection and single-tree selection methods to convert even-aged oak-hickory stands to uneven-aged ones.

(2) To test two methods of regulation and two density levels in combination with group selection.

(3) To compare growth and yield of stands that are managed and regenerated under group selection, two methods of regulation, and two density levels.

As part of objective 3, log grades and tree quality classes (tree quality classes defined using Grosenbaugh 1955 tree classes) were assigned to each sawtimber-sized tree on the growth and yield plots to assess the change in tree quality over time. This also addresses the feasibility of using the Grosenbaugh tree classes on trees in oak-hickory forests of the Boston Mountains. In this paper we describe the preharvest conditions of the sawtimber and any effect of the initial harvest treatments on the residual sawtimber component.

METHODS

Study Region

The Boston Mountains are the highest and most southern member of the Ozark Plateau physiographic province (fig. 1). They form a band 30 to 40 miles wide and 200 miles long from northcentral Arkansas westward into eastern Oklahoma. Elevations range from about 900 ft in the valley bottoms to

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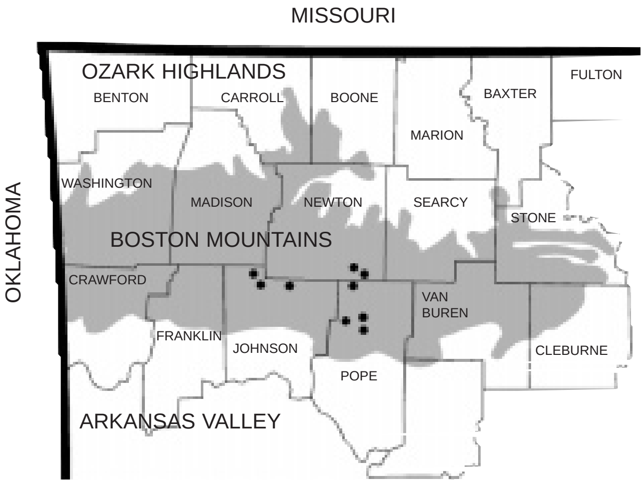

Figure 1—Location of study areas.

2,500 ft at the highest point. The plateau is sharply dissected, and most ridges are flat to gently rolling and are generally less than 0.5 mile wide. Mountainsides consist of alternating steep simple slopes and gently sloping benches.

Soils on mountaintops and slopes usually have shallow to medium depth and are represented by medium-textured members of the Hartsells, Linker, and Enders series (Typic Hapludults). They are derived from sandstone or shale residuum, and their productivity is medium to low. In contrast, soils on mountain benches are deep, well-drained members of the Nella and Leesburg series (Typic Paleudults). They developed from sandstone and shale colluvium, and their productivity is medium to high. Rocks in the area are alternating horizontal beds of Pennsylvanian shales and sandstones. Annual precipitation averages 46 to 48 in., and March, April, and May are the wettest months. Extended summer dry periods are common, and autumn is usually dry. The frost-free period is normally 180 to 200 days long.

Study Description

The regulation study design is a split-plot factorial layout, replicated three times, with aspect as the main plot treatment and residual stand density and regulation as sub-plot treatments. The aspects are northeast and southwest, residual densities are 65 and 85 ft^2/ac, and the regulation methods are free thinning and structural control. Although the study is a straightforward test of group selection with two residual stand treatments (i.e., free thinning versus structural control), it can also be used to evaluate single-tree selection by analyzing the structural control treatment.

Stand Density Treatments

The two residual density levels are 65 and 85 ft^2/acre of basal area in trees 5.6 in. and larger. The overstory density treatments were applied on the plot and buffer areas outside the group opening. The 65 ft^2 density level approximates the B-level of stocking for upland oak stands and should be near the optimum for stand and crop tree growth (Gingrich 1967). The 85 ft^2 density represents about 70-75 percent stocking and is appropriate for a first thinning in older stands that have relatively high stocking levels (Sander 1977).

Plot Location and Layout

The study was installed on the Ozark National Forest in well-stocked hardwood sawtimber stands with no history of previous cutting for at least 50 years. Twenty-four 11-acre plots were installed in nine forest stands on the Buffalo, Bayou, and Pleasant Hill Ranger Districts (fig. 1). Study plots were located on north/east and south/west facing mountain slopes and benches in oak-hickory stands representative of the sites and stand conditions that are designated for uneven-aged management by the Ozark National Forest. These plots were replicated by National Forest Districts with 8 plots established on each District. Harvesting was done by each National Forest District using standard timber sales. Logging at all study locations utilized chainsaw felling and tree-length skidding by standard rubber-tired skidders.

Sample plots consist of a 7.2-acre net plot plus a 66-ft buffer for a total of 11 acres (fig. 2). In addition, each 7.2-acre net plot was subdivided into twelve 0.6-acre subplots. Of the twelve subplots on each net plot, three subplots located on one end of the plot (numbers 1-3 or 10-12) were randomly selected for a separate competition control study (fig. 2) (Graney and Murphy 1997). The remaining 9 subplots were used in the regulation study.

Measurements

A complete preharvest tally of all overstory trees, dbh 5.6 in. and larger, was taken by species, tree class [per Grosenbaugh 1955 (see footnote, table 2)], and 1-in. dbh classes. This tally was used to apply the treatments.

In the postharvest phase, overstory growth and yield were measured on a series of 0.2-acre circular plots located in the center of each 0.6-acre subplot (fig. 2). On each 0.2 acre plot all overstory trees 5.6 in. dbh and greater were numbered and mapped by azimuth and distance from plot center. The following information was collected:

(1) diameter to nearest 0.1 in.,

(2) total height for a sample of trees in each 1-in. dbh class,

(3) log grade for the butt log of all sawtimber trees,

(4) damage to crowns and boles resulting from logging,

(5) tree quality by Grosenbaugh tree class, and

(6) age of selected dominants or codominants.

The 16-foot butt log on sawlog trees (trees larger than 11.6 in. dbh) was graded using the U.S. Forest Service log grading system (Hanks 1976). We will monitor the effects of treatments on stem quality at 5-year intervals. The plots will be cut every 10 years.

Regulation Techniques

The regulation techniques applied to the residual stand are (1) area regulation with free thinning and (2) area

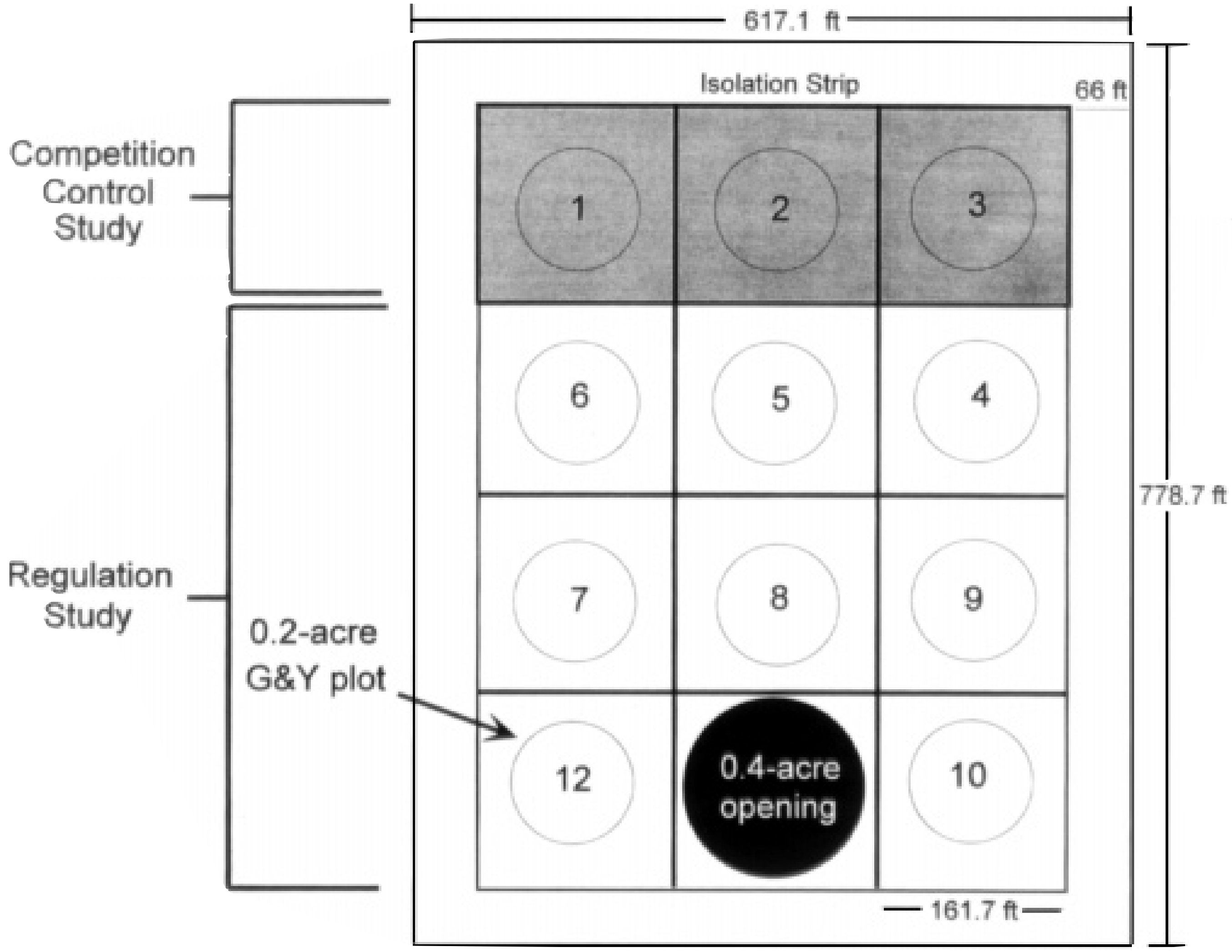

Figure 2—Layout of the plot, subplots, 0.2-acre growth plots, and group opening.

regulation with structural control. Group selection will be used with area control. Assuming an 80-year conversion period and a 10-year cutting cycle, one-ninth of the area would be regenerated each cutting cycle.

Opening size was approximately two times the average height in the dominant trees in the adjoining stand. This resulted in an average opening size of 0.4 acres (range 0.25 to 0.47 acres) in typical stand conditions. Selection of the initial group opening subplots were based on the presence of large reproduction or saplings of desirable species, sprouting potential of desirable species in the small poletimber class, and overstory stocking. A subsequent group opening will be created every 10 years. Group opening diameter and subplot dimensions restrictions precluded complete overstory removal in group opening subplots. Non-opening areas of group opening subplots were cut to the residual stand basal area target.

In the first regulation method, the residual stand (the area not in group openings) was cut to the target density by free thinning. Trees were removed in the following priority: (1) larger cull and defective trees, (2) competing trees of poor form and quality, and (3) intermediate and suppressed trees of lower quality and value. The primary objective was to improve residual stand quality and vigor.

In the second method, the residual stand was thinned to a target stand structure. The target stand structure has a minimum dbh of 5.6 in., a maximum dbh of 18.5 in., a *q* of 1.3 (assuming 2-in. diameter classes), and residual densities of 65 and 85 ft^2/ac. Law and Lorimer (1989) suggested *q* values of 1.3, 1.5, or 1.7 for 2-inch classes in upland hardwoods, and Smith and Lamson (1982) recommended a 1.3 q-value for sawtimber production and higher *q*'s for smaller-product objectives. Because our objective is quality sawtimber production, we chose a *q* of 1.3. We selected the maximum dbh to produce a grade one butt log, also in accordance with a quality sawtimber objective.

When marking to achieve the residual structure for areas outside openings, we divided the trees into four size classes—small poletimber (6-8 in. dbh), large poletimber (9-11 in. dbh), small sawtimber (12-15 in. dbh), and large sawtimber (>15 in. dbh). We calculated the target residual basal area for each class and marked the stand to conform to the target. However, when marking in the poletimber classes, we discriminated against the noncommercial and low-value species and did not always achieve the target structure for the small poletimber class. In these cases, we left more stocking in better-quality trees in the larger classes. We did, however, leave some low vigor oak stems in the small diameter classes to evaluate survival and growth. The main goal was to leave the residual basal area in the best quality trees available on the plot.

Statistical Tests

To test for differences in the proportion of grade 1 trees as the result of the treatments, a split-plot analysis was performed with the arc-sine square root transformation of proportion of grade 1 trees as the dependent variable, aspect as the main plot, and density and regulation as factors with three replications. None of the variables were significant. The results were probably confounded by 4 of the south aspect plots having site indices similar to north aspects.

RESULTS AND DISCUSSION

Preharvest Conditions

The total overstory basal area of trees ≥ 5.6 inches dbh was remarkably uniform across all plots, ranging from 92 to 114 ft^2/acre for north aspects and from 91 to 112 ft^2/acre for plots on the south aspects. The mean stand age and range for north aspects was 79 years and 71 to 93 years respectively. Stands on south aspects were slightly younger with a mean of 74 years and a range from 68 to 81 years. Red or white oak site index on north aspects ranged from 62 to 72 ft (base age, 50 years) and 55 to 69 ft for south aspects. Mean site index was 67 ft and 62 ft for north and south aspects, respectively.

Sawtimber basal area was slightly greater on north aspects (table 1). The basal area in desirable species was 90 percent for both aspects, and the oaks comprise 80 percent of the basal area. There were some minor differences in species mix by aspect. Basal area for red oaks and hickory was greater on north aspects, while white oak basal area averaged more on south aspects.

The tree class “grower” (Grosenbaugh 1955) designates trees that are the objective of management for quality timber. Table 2 shows that these crop trees are a much larger proportion of the stand on north aspects. This larger proportion occurs partly because the red oaks are found more often on north aspects, and red oaks tend to have a larger proportion of growers than white oaks. The incidence of culls and high-risk trees (riskers/killers/culls) occurs with equal proportions on both aspects.

Postharvest Conditions

The treatments affected residual species composition. The more desirable species were retained, and the other

Table 1—Preharvest species composition by basal area and aspect for sawtimber trees (d.b.h.>11.5 inches)

Species groups[a]	North aspect[b]	South aspect
------------ Basal area (*square feet per acre*) ------------		
Hickory-shortleaf pine	5.3	2.9
Other overstory[c]	6.6	5.4
Ash-cherry-walnut	1.4	1.0
White oaks	19.4	24.0
Red oaks	32.4	23.3
All species	65.1	56.6

[a] Species preferred for management: white oaks, red oaks, ash, cherry, walnut, hickory, shortleaf pine.
[b] Means are based on twelve 7.2 acre plots for each aspect.
[c] Other overstory = basswood, beech, blackgum, cucumber tree, sugar maple, sweetgum.

groups were discriminated against when making the cut. Therefore, there was a larger reduction in the proportion of other overstory species groups (table 3). We now have about 96 percent of the residual sawtimber basal area in desirable species. The 85 ft^2 basal area treatments for both structural control and free thinning did not give as much freedom in molding species composition as the 65 ft^2 treatment, because less basal area was removed. There was also no apparent difference in species composition between the two regulation methods.

A major objective was to improve residual stand quality; therefore, culls and lower quality trees had the highest priority for removal regardless of stem size. In table 4, the largest reduction in basal area occurred in culls and the lower quality classes. Culls were reduced from 10 to 13 percent in preharvest conditions to 0 to 2 percent in the residual stands. Any culls that were left occurred in the 85 ft^2/acre treatment to meet this residual basal area target. The reduction in basal area was least in the grower and sleeper categories, which are the best potential crop trees. The proportion of basal area in these trees was increased.

Tree Grade Distribution

The 16-ft butt logs of all sawtimber-sized trees on the 0.2-acre growth and yield plots were evaluated using the U.S. Forest Service tree grading system (Hanks 1976). The total number of sawtimber-sized trees was 2,225, about a third of the sawtimber on the study areas. About 40 percent of the residual sawtimber trees were graded as 1 or 2. More of the grade 1 and 2 trees were located on north aspects (table 5). More of the grade 3 trees were located on south aspects, while the number of grade 4 trees were evenly distributed on both aspects.

There are greater differences in tree grades 1 and 2 on north and south aspects if site index is considered. Four of the south aspect plots had site indexes more comparable to north slopes. These 4 plots confounded the differences between aspects and accounted for 60 percent of the

Table 2—Preharvest structure by Grosenbaugh tree class and aspect for sawtimber trees (diameter greater than 11.5 inches)

Grosenbaugh tree class[a]	North aspect[b]	South aspect
	Basal area (*square feet per acre*)	
Grower	24.5	14.8
Sleeper	5.7	8.2
Cipher/topper/slower	28.1	26.4
Risker/killer/cull	6.8	7.2
All species	65.1	56.6

[a] Adapted from Grosenbaugh (1955):

<u>Grower</u>: A merchantable tree that is vigorous and has no serious defects that would affect growth or desirability of the tree as potential sawtimber growing stock. A grower should also have the potential of developing a grade 1 butt log and have an expectancy of at least 0.90 of living until the next cutting cycle. Some people call such trees “crop trees” or “good growing stock.”

<u>Cipher</u>: A merchantable tree whose expectancy of living for the next cutting cycle is at least 0.90, but does not meet the qualifications of a grower because of slow growth or undesirable characteristics, and is not competing with desirable reproduction or saplings. This tree can either be “financially mature” or may have limitations that disqualify it as a grower.

<u>Sleeper</u>: A cipher which has the potential to become a grower if it is released by removing competing trees.

<u>Topper</u>: A merchantable tree similar to a cipher but overtopping desirable reproduction or saplings.

<u>Slower</u>: The least potentially productive of several merchantable trees (but not riskers or killers—see below) competing in inadequate growing space. It should be cut in thinning.

<u>Risker</u>: A merchantable tree whose life expectancy for the next cutting cycle is less than 0.90. It should be cut to salvage potential loss through mortality.

<u>Killer</u>: A merchantable tree infested with contagious pathogens.

<u>Cull</u>: A tree that is merchantable size but not salable because of defect or other factors.

[b] Means are based on twelve 7.2 acre plots for each aspect

grade 1 and 2 trees on south aspects. The remaining south aspect plots were composed mainly of grade 3 and 4 trees.

Regulation method (free thinning versus structural control) had no apparent effect on tree grade distribution (table 5). One of the principal objectives in both free thinning and structural control was to improve the residual stand by cutting the worst trees and leaving the best ones. Thus, the results from both types of cuts were very similar. In structural control, we discriminated against poorly formed stems, low-value species, and noncommercial stems. Therefore, the residual stand structure was not always attained, but the residual stand was of better quality.

Residual basal area apparently did not affect the distribution of tree grade (table 5). Although the higher

Table 3—Postharvest species composition by aspect for sawtimber trees (diameter greater than 11.5 inches)

Species groups[a]	North aspect[b]	South aspect
	Basal area (*square feet per acre*)	
Hickory-shortleaf pine	3.9	2.1
Other overstory	1.3	1.6
Ash-cherry-walnut	0.9	0.6
White oaks	15.5	18.6
Red oaks	24.5	16.3
All species	46.1	39.2

[a] See footnote a in table 1 for species list.
[b] Means are based on twelve 7.2 acre plots for each aspect.

Table 4—Postharvest Grosenbaugh tree class distribution by aspect for sawtimber trees (diameter greater than 11.5 inches)

Grosenbaugh tree class[a]	North aspect[b]	South aspect
	Basal area (*square feet per acre*)	
Grower	23.8	14.3
Sleeper	5.7	8.1
Cipher/topper/slower	16.3	15.9
Risker/killer/cull	0.3	0.9
All species	46.1	39.2

[a] See footnote a in table 2.
[b] Means are based on twelve 7.2 acre plots for each aspect.

basal area treatment allowed more trees to be retained, the relative proportion of tree grades was not affected. The difference in the proportion of grade 1 trees is probably due to the marking of the structural control plots where the target structure could not be attained in the smaller diameters. Therefore, more of the larger trees were retained to satisfy density requirements.

While the proportion of quality trees was the same for the 65 ft^2 and 85 ft^2 density treatments, the economic operability of the cut and potential effects on future stand and tree quality development will vary. The 85 ft^2 target permitted removal of approximately 16 ft^2 of basal area per ac, mostly in cull and low quality trees. Although higher quality trees will be removed in future 10-year cycle cuts, basal area removed will be less than 20 ft^2 per acre and will be further reduced by mortality expected in the higher density mature stands (Graney and Murphy 1994). Increasing the numbers of grade 1 and grade 2 trees in current stands will depend on maintaining or increasing growth of pole- and small sawtimber-size stems in the higher quality (grower and sleeper) tree classes. Quality

Table 5—Postharvest distribution of sawtimber trees by tree grade, aspect, regulation method and residual basal area

Tree grade	Aspect		Regeneration method		Basal area		Overall mean
	North	South	Free thinning	Structural control	65 ft^2	85 ft^2	
	Percent						
Grade 1	16	11	13	15	11	16	14
Grade 2	29	22	25	26	24	26	25
Grade 3	32	43	36	37	39	36	37
Grade 4	23	24	26	22	26	22	24

stems in 70- to 80-year-old Boston Mountain hardwood stands respond with increased diameter growth following intermediate cutting to medium or lower stand densities (Graney and Murphy 1994). Cutting residual stands to 65 ft^2 allows more flexibility in the removal of lower quality stems and the crown release of potentially higher quality stems in the grower and sleeper tree classes.

Growers are trees that have attained or will probably attain a grade 1 butt log during their lifetime; these trees are the crop trees of management. Of the total number of trees classified as growers, 73 percent were classified as grade 1 or 2 (table 6). The primary reason that some growers are not grade 1 is that they do not yet meet the size requirements but have the potential to do so, given time. As they grow, however, they will eventually satisfy the criteria for grade 1 logs. Sleepers are trees that have good stem quality and could develop into growers if some remedial management is done, such as thinning. Trees classified as sleepers in our study were usually in the large poletimber or small sawtimber classes and could not attain log grade 1 or 2 because they did not meet the size requirements. These trees are unlikely to grow into a size class large enough to meet log grade 1 or 2 in the absence of management. Fifty-three percent of the sawtimber trees were classified as growers or sleepers, which indicates that the potential for high quality sawtimber of upland oak stands in the Boston Mountains is excellent.

CONCLUSIONS

Although 39 percent of the sawtimber trees in the study qualify for grade 1 and grade 2 trees, the 60 percent that are now growers and sleepers indicates that the potential for increase in tree quality is good. These trees are now too small to qualify for the higher grades. It will take time and management before these smaller trees reach the merchantability standards for grade 1 and 2 logs. A major reason for the relatively low proportion of higher grade stems in the 70- to 80-year-old Boston Mountain oak-hickory stands is residual stand diameter. Of the residual stand component, only 50 percent met the minimum diameter requirements for grade 1 or grade 2 logs. However, in the remaining stocking of the small sawtimber class, more than 60 percent are in the grower and sleeper classes and should develop into grade 1 and grade 2 trees as they grow into the larger sawtimber-sized classes.

One reason for the lack of differences among treatments is likely the short time interval since cutting. The high site indices for benches on some southwest facing slopes also may be masking any immediate differences. Aspect alone does not adequately separate high site index sites from low site index sites, as evidenced by the similarity of some of the south aspect plots to north aspect plots. In the Boston Mountains, productive oak sites are also associated with the deep, well drained colluvial soils commonly found on concave, gently sloping inner bench positions that are typical of upper mountain slopes at all aspects (Graney 1977).

Table 6—Postharvest distribution of sawtimber trees by Grosenbaugh tree class and tree grade

Tree grade	Grower	Sleeper	Other classes	All classes
	Percent			
Grade 1	36	0	1	14
Grade 2	37	4	25	25
Grade 3	25	42	44	37
Grade 4	2	54	30	24

The residual basal area of 85 ft^2 is likely too high to be used as an effective management tool and is not likely to produce an economically feasible sale. In the 85 ft^2 treatment the harvest consisted mainly of low quality and cull material. The residual stand of 65 ft^2 produced a harvest of merchantable material and shows promise in increasing tree quality. A residual stand of less than 65 ft^2 of basal area may enhance these effects if carefully applied. Overall, this study indicates that there is excellent potential to improve stand quality in the Boston Mountains of northern Arkansas.

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This study originated under the late Paul Murphy, principal mensurationist and project leader with the Southern Research Station in Monticello, AR. Paul's contributions to the art and science of forest growth and yield will be long remembered. His colleagues will miss his keen insights, his professionalism, and his friendship. We also thank two anonymous reviewers for helpful comments on this manuscript.

REFERENCES

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

Graney, David L. 1977. Site index predictions for red oaks and white oak in the Boston Mountains of Arkansas. Res. Pap. SO-139. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 9 p.

Graney, David L.; Murphy, Paul A. 1994. Growth and yield of thinned upland oak stands in the Boswton Mountains of Arkansas. Southern Journal of Applied Forestry. 18(1): 10-14.

Graney, David L.; Murphy, Paul A. 1997. An evaluation of uneven-aged cutting methods in even-aged stands in the Boston Mountains of Arkansas. In: Pallardy, S.G.; Cecich, R.A.; Garrett, H.E.; Johnson, P.S., eds. Proceedings, 11th central hardwood forest conference; 1997 March 23-26; Columbia, MO. Gen. Tech. Rep. NC-188. St. Paul, MN: U.S. Department of Agriculture, North Central Forest Experiment Station: 130-146.

Grosenbaugh, Lewis R. 1955. Better diagnosis and prescription in southern forest management. Occas. Pap. 145. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 27 p.

Hanks, Leland F. 1976. Hardwood tree grades for factory lumber. Res. Pap. NE-333. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 81 p.

Law, Jay R.; Lorimer, Craig G. 1989. Managing uneven-aged stands. In: Clark, F. Bryan; Hutchinson, Jay G., eds. Central hardwood notes. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station: 6.08(1)-6.08(6).

Loewenstein, Edward F.; Garrett, Harold E.; Johnson, Paul S.; Dwyer, John P. 1995. Changes in a Missouri Ozark oak-hickory forest during 40 years of uneven-aged management. In: Gottschalk, Kurt W.; Fosbroke, Sandra L.C., eds. Proceedings of the 10th central hardwood forest conference. Gen. Tech. Rep. NC-197. U.S. Department of Agriculture, Forest Service: 159-164.

Miller, Gary W.; Schuler, Thomas M.; Smith, H. Clay. 1995. Method for applying group selection in central Appalachian hardwoods. Res. Pap. NE-696. Radnor, PA: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 11 p.

Murphy, Paul A.; Shelton, Michael G.; Graney, David L. 1993. Group selection—problems and possibilities for the more shade-intolerant species. In: Gillespie, Andrew R.; Parker, George R.; Pope, Phllip E.; Rink, George, eds. Proceedings of the 9th central hardwood forest conference. Gen. Tech. Rep. NC-161. U.S. Department of Agriculture, Forest Service: 229-247.

Sander, Ivan L. 1977. Managers handbook of oaks in North Central States. Gen. Tech. Rep. NC-37. U.S. Department of Agriculture, Forest Service. 35 p.

Smith, H. Clay; Lamson, Neil I. 1982. Number of residual trees: a guide for selection cutting. Gen. Tech. Rep. NE-80. U.S. Department of Agriculture, Forest Service. 33 p.

PREDICTING SAPLING GROWTH AND RECRUITMENT IN DIFFERENT SIZE CANOPY GAPS

John M. Goodburn and Craig S. Lorimer[1]

Abstract—Accurate simulation of the effects of uneven-aged management on future stand structure and species composition requires good empirical data on sapling growth, survival, and canopy recruitment, particularly as these dynamics are influenced by the number and size of canopy openings created at each harvest. In order to examine the response of various species to different levels of understory light and woody competition, we measured annual growth increments (1993-97) on tree saplings in 30 northern hardwood stands in north-central Wisconsin and adjacent western Upper Michigan. Understory conditions in the 100 "gap" plots sampled ranged from closed canopy to group selection openings (>400 m^2). Average daily radiation received by individual saplings was estimated from computer analysis of hemispherical canopy photographs taken directly above the 510 subject saplings (size 25 cm tall to 10 cm dbh). In addition, data on woody competition experienced by subject trees were collected for all sapling and overstory trees in the plot.

Shade-tolerant sugar maple reached close to its maximum height growth rate (30-40 cm/yr) at surprisingly low light intensities typical of small single tree gaps (2-15 percent full sunlight). Increasing gap size had little further effect on growth rate for maple. In contrast, height growth rates of midtolerant species continued to increase with increasing light intensity, consistent with the normal expectation that midtolerant species will have a substantial competitive advantage in larger openings. At moderately high gap light intensities (40-60 percent full sunlight, typical of gaps 100-300 m^2), growth rate increased substantially for both red oak and white ash (~60 cm/yr). For all species, height growth rate clearly increased with sapling size (fig. 1). For example, at a given light intensity of 25 percent full sunlight, the height growth rate of a large white ash sapling (>2.5 cm dbh) was nearly three times greater than that of a smaller white ash sapling (<1.5 m tall) in the same light environment. Species specific regression equations for height growth as a function of sapling height and PPFD (photosynthetic photon flux density) had R^2 values in the range of 0.47 - 0.62.

Sapling growth rates can be expected to continually change over time as a result of increasing sapling size, changes in sapling height relative to competitors, and decreasing gap size due to lateral gap closure. To forecast interspecific competition over time in response to canopy gap formation and closure, we sought to develop predictive equations for sapling growth and mortality based on simple measures that could be easily updated within the context of an iterative model. While irradiance levels increased with gap size, light reaching individual saplings was also strongly influenced by its horizontal position within the gap and its relative height. Almost half of the variation in PPFD could be explained by four variables (i.e., subject tree height, square-root of gap area, distance of the sapling north of the gap's southern border, and a competition index based on the aggregate crown projection area of all taller saplings; R^2=0.48). These four variables were also found to be among the most significant in regression equations predicting sapling height growth. Coefficients of determination (R^2) for height growth equations ranged from 0.54 for the sugar maple up to 0.85 for red oak. These equations are being incorporated into a spatially explicit, crown-based model of forest dynamics (CROWN) that keeps track of the size, growth, and mortality of individual trees within a stand.

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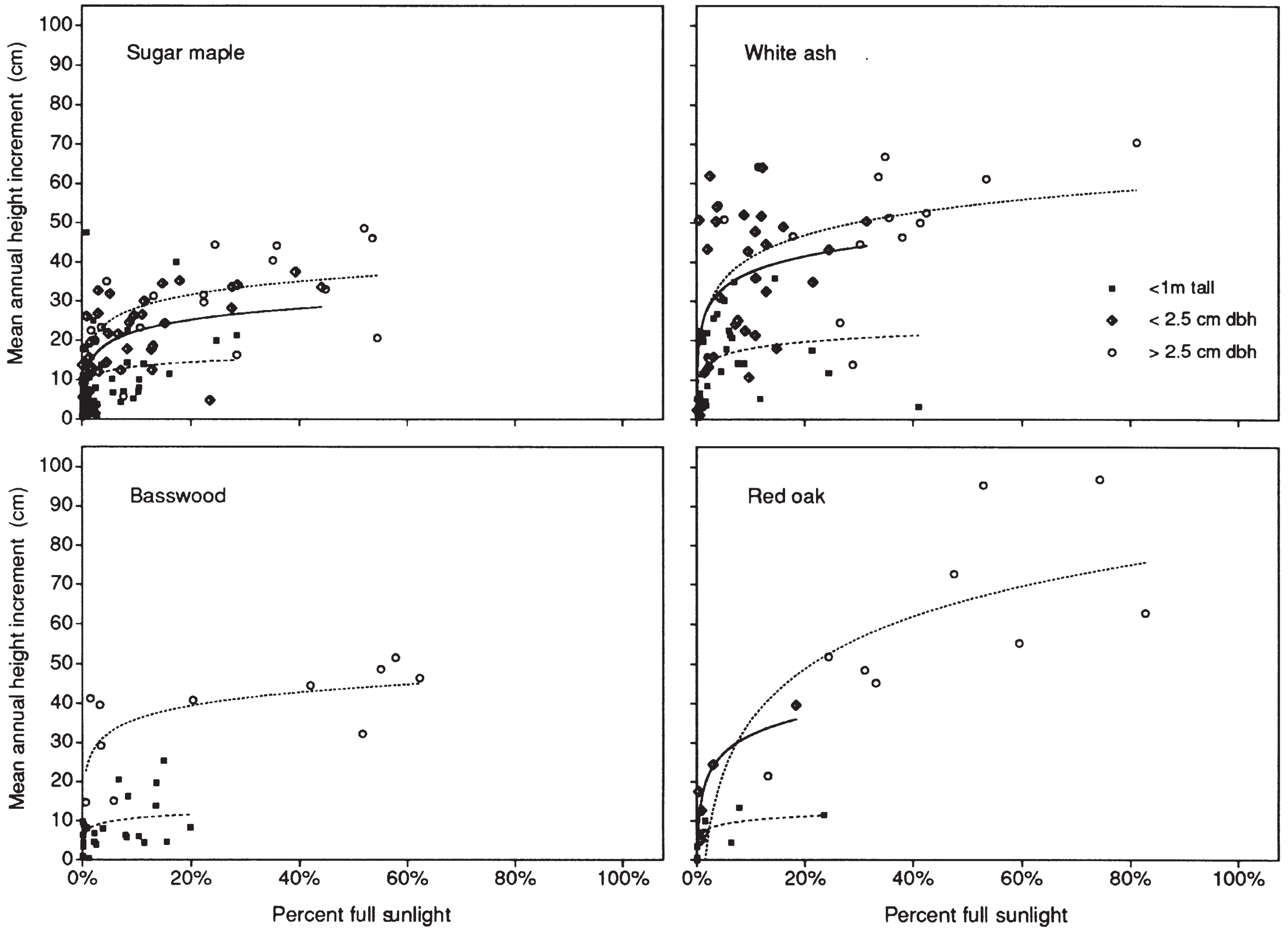

Figure 1—Mean annual height growth versus the amount of solar radiation reaching the sapling. For each species, separate response curve are displayed for three different sapling size classes (< 1.5 m tall, >1 m tall but <2.5 cm dbh, >2.5 cm dbh).

Disturbance Effects

INDIVIDUAL TREE FIVE-YEAR BASAL AREA AND CROWN DIAMETER GROWTH IN APPALACHIAN HARDWOOD STANDS AS INFLUENCED BY THINNING AND GYPSY MOTH DEFOLIATION

Kurt W. Gottschalk[1]

Abstract—I evaluated silvicultural treatments to minimize gypsy moth effects on forests in experimental plots on the West Virginia University Forest. Two treatments, presalvage thinning and sanitation thinning, were used. As part of the evaluation, I measured individual tree basal area growth and crown diameter growth over a five-year period. This period began with pretreatment measurements of stem dbh and crown diameter in 1989 before thinning and defoliation. Thinnings were installed during the winter of 1989-1990 and had paired control stands that were not thinned with four replicates of each treatment and control. Gypsy moth defoliated six of the 16 stands in 1990 and 1991. Mortality resulting from the defoliation-induced stress along with drought stress in 1991 occurred over the following three years. At the end of this period of stress and mortality, stem dbh and crown diameter were remeasured for all living trees. Stem dbh was converted to basal area. Basal area growth and crown diameter growth (or shrinkage in some cases) were calculated by taking the difference between the two measures.

Mean change in crown diameter was 3.5 feet and was normally distributed. About 35 percent of the trees had a reduction in crown diameter due to dieback. The range of crown diameter changes was -28 to +31 feet. Crown diameter change was significantly correlated with species, treatment (control, control+defoliation, thinning, thinning+defoliation), and the pretreatment crown class, but was not correlated with the pretreatment crown vigor nor the number of sides released. Despite the significance of the correlated variables, they explained very little of the variation in crown diameter change.

Mean change in basal area growth was 0.07 square feet and had a skewed distribution. Less than 1 percent of the trees had a decrease in basal area and 65 percent had only small increases in basal area (0.0 to 0.1 ft^2). The distribution dropped in an exponential pattern up to the maximum change of 1.28 square feet. Basal area change was significantly correlated with species, treatment, pretreatment crown class, pretreatment crown vigor, number of sides released, and crown diameter change. In a forward stepwise regression, the first and most important variable was pretreatment crown class. It was then followed by crown diameter change, species, and pretreatment crown vigor but all of the variables explained only a small portion of the additional variation.

Evaluation of treatment effects on crown diameter change showed that defoliation reduced crown diameter growth regardless of thinning treatment and thinning treatment increased crown diameter growth regardless of defoliation. Mean crown diameter growth (in feet) by treatment was:

Control + defoliation	2.5a
Thinned + defoliation	2.9ab
Control	3.2b
Thinned	4.7c

Evaluation of treatment effects on basal area change showed that thinning treatments increased growth regardless of defoliation and defoliation actually increased growth of surviving trees but less so in thinned stands than in control stands. Mean basal area growth (in square feet) by treatment was:

Control	0.055a
Control + defoliation	0.073b
Thinned + defoliation	0.084c
Thinned	0.084c

The positive effects of thinning on crown diameter and basal area growth trends especially when defoliated along with information on the mortality rates in the thinned versus unthinned stands support the use of thinning before gypsy moth defoliation as an useful technique to minimize gypsy effects on forests.

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INDIVIDUAL TREE MORTALITY PREDICTION FUNCTIONS FROM GYPSY MOTH DEFOLIATION AS WELL AS TREE, STAND, AND SITE VARIABLES

J.J. Colbert[1]

Abstract—Stands from central Pennsylvania were followed for fifteen years to provide records of plot slope, position, aspect, site index, land capability class; tree species, diameter, crown class, vigor or health, as well as annual gypsy moth defoliation. These data are used to create individual tree mortality prediction models for species and species groups under various degrees of defoliation. These models are tested against independent data and compared to previously published models for predicting tree mortality in similar situations. These results are then compared to log-odds-ratio analysis of variance results for these same data to provide further understanding of utility of these models and their precision for use in long term predictions of forest stand conditions. While analysis of variance provides a very useful characterization of mortality over these data, the difference in utility of the individual tree mortality models is demonstrated for predicting continuing impacts of defoliation over a wider range of defoliation histories and tree conditions. It is demonstrated how individual tree mortality models utilize defoliation history on individual trees and can account for the cumulative effects of defoliation over a number of years while other analytic procedures account for a effects associated with a prescribed pattern of defoliation over a fixed time interval.

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CHARACTERISTICS OF THE CHESTNUT BLIGHT FUNGUS ISOLATED FROM SCARLET OAK IN PENNSYLVANIA

D.D. Davis and M.L. Torsello[1]

Abstract—More than 100 isolates of the chestnut blight fungus, *Cryphonectria parasitica* (=*Endothia parasitica*), were collected during a survey across Pennsylvania from infected scarlet oaks (*Quercus coccinea*) suffering from basal Cryphonectria cankers. Comparisons were made among isolates, as well as to a standard virulent (relative to American chestnut) and standard hypovirulent isolate in terms of linear growth on agar, lesion area induced in apple fruit, and canker area induced on American chestnut saplings. A ranking test indicated that growth on agar, lesion area induced in apple, and canker size induced on American chestnut were correlated. Most isolates grew on agar, infected apple fruit, and infected chestnut stems in a manner similar to that of the standard virulent isolate. A reservoir of virulent inoculum from scarlet oaks may confound efforts at biological control of chestnut blight on American chestnut. The possibility that a small number of isolates from scarlet oak may be hypovirulent on American chestnut should be investigated because one isolate had characteristics similar to the known hypovirulent isolate.

INTRODUCTION

Cryphonectria parasitica (Murrill) Barr (=*Endothia parasitica* (Murrill) P.J. Anderson & H.W. Anderson), virtually eliminated the American chestnut (*Castanea dentata* (Marsh.) Borkh.) as a forest tree throughout its range in the USA in the early 1900s (Roane and others 1986). Recently, there has been renewed interest in control of chestnut blight on American chestnut through the use of biocontrol (Nuss 1992). However a confounding factor in this biocontrol effort is that the surviving sprouts of American chestnut often grow in the understory of oak forests, and many of companion oak species, including scarlet oak (*Quercus coccinea* Muenchh.), also are susceptible to *C. parasitica* (see review in Torsello and others 1994). Working with oaks in southeastern USA, Nash and Stambaugh (1982) suggested that infected oaks could serve as reservoirs of virulent *C. parasitica* inoculum, especially in forest stands where American chestnut has been eliminated They also indicated that the virulence of some of the southeastern oak isolates, when placed into American chestnut, was equal to or greater than those isolates originally obtained from American chestnut (Nash and Stambaugh 1987).

In addition to harboring these virulent isolates of the chestnut blight fungus, scarlet oak may also contain less virulent, or hypovirulent, strains of the fungus. In fact, some biocontrol efforts in controlling chestnut blight in American chestnut are based on the use of this hypovirulence (Nuss 1992). Although hypovirulent isolates of *C. parasitica* have not been reported from hosts other than American chestnut, infected oaks may represent a potential, unstudied reservoir of hypovirulence for use in this biocontrol effort. Determination of isolate virulence with respect to American chestnut involves inoculation and evaluation of canker development on chestnut stems in the field (Elliston 1982, 1985; Scibilia and Shain 1989). However, such field trials are time consuming, and the limited number of uninfected chestnut stems present in some forest stands may not allow studies involving large numbers of isolates. Therefore, Elliston (1982, 1985) suggested inoculation of apple fruit as a rapid and efficient means to initially screen large numbers of isolates of *C. parasitica* to estimate virulence. Fulbright (1984) reported that the rate of colonization of “Granny Smith” apple fruit infected by *C. parasitica* isolates obtained from American chestnut might be used to estimate relative virulence of the same isolates in chestnut. In addition, Bedker (1989) suggested that linear growth rate in culture might be used to estimate the virulence of *C. parasitica* isolates in chestnut.

Linear growth in culture and ability to colonize apple fruit, in conjunction with chestnut stem inoculations to evaluate virulence, have not been examined for isolates of *C. parasitica* from scarlet oak. If successful, estimation of potential virulence of isolates based on linear growth in culture and/or apple colonization would allow a quick identification of source material for further studies dealing with virulence of *C. parasitica*. Also, inoculation of American chestnut stems under field conditions may yield insights into the possible role of inoculum from oaks on biological control of chestnut blight.

The objective of this study was to compare isolates of *C. parasitica* from scarlet oak growing in Pennsylvania, in terms of linear growth in culture, lesion size produced in apple fruit, and canker size induced on American chestnut stems. Comparisons were also made to known virulent or hypovirulent isolates.

METHODS

Linear Growth in Culture

During a survey across Pennsylvania, we collected 102 isolates of *C. parasitica* from cankered scarlet oak (SO);

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detailed methods regarding isolation and culturing have been presented (Torsello and others 1994). Initial isolations were cultured on acidified Difco potato dextrose agar (aPDA, 1 ml of 85 percent lactic acid per liter of potato dextrose agar). Subsequent cultures were maintained on acidified PDA containing methionine (100 mg/l) and biotin (1 mg/l) (PDAmb) (Anagnostakis and Aylor 1984). Occasionally, more than one SO isolate was obtained from a single canker. Since several vegetative compatibility types can occur in a single canker (Nash and Stambaugh 1989), each isolate was treated as a distinct entity. The linear growth of the 102 SO isolates was compared to the linear growth of EP155 and EP713, two *C. parasitica* isolates of known virulence. EP155 is a “standard” (Anagnostakis 1992) virulent isolate, American Type Culture Collection (ATCC) #38755, and has a normal phenotype. EP713 is a slow-growing, hypovirulent isolate (ATCC #52571) (Anagnostakis 1992). EP155 was obtained from W. MacDonald, West Virginia University, and EP713 was obtained from S. Anagnostakis of The Connecticut Agricultural Experiment Station.

One 7-mm diameter plug of PDAmb with mycelium was removed from each 7-day-old SO isolate, as well as EP155 and EP713, and placed on PDAmb against the wall of a petri plate. Four sets of each isolate were grown upside down in petri plates to minimize moisture accumulation on the agar surface. Plates were maintained in darkness at 21^{o}C on four shelves in a controlled environment room utilizing a randomized block design with two replications. Shelves located perpendicular to a possible vertical temperature gradient in the room were considered as blocks. Each block consisted of a complete set of 102 SO isolates, EP155, EP713, and a control. After 7 days, the morphology of each colony was described and the growth of each isolate measured by taking two linear measurements per plate, from the edge of the plug to the distal edge of the colony, and averaged.

The 102 SO isolates, EP155 and EP713 were ranked in order of linear growth produced after 7 days (Minitab 1991). Growth data also were analyzed using ANOVA and Tukey’s Honestly Significant Difference (HSD) multiple comparison procedure (p=0.05) to determine if significant differences in growth occurred among isolates (SAS 1985).

Apple Inoculations

One isolate from the linear growth study failed to grow. Therefore, 101 SO isolates, EP155, and EP713, plus a sterile PDAmb plug as a control were used to inoculate apple fruit. Fruit of the apple cultivar “Granny Smith” were sorted for uniformity, washed with sterile distilled water and detergent, wiped with 65 percent ethanol, and placed in a laminar flow hood. Each fruit was labeled with a waterproof marker and inoculated with three isolates. Each apple was inoculated with EP155 as a standard; the remaining two inoculations per apple were either SO isolates, EP713, or the control. A 7-mm diameter plug was removed from each fruit using a sterile cork borer, and a 7-mm plug of PDAmb containing the isolate, or sterile PDAmb agar as a control, was placed into the hole. Non-absorbent cotton was placed over the plug and the apple was wrapped with a strip of parafilm to minimize evaporation. Apples were placed on five vertically arranged shelves in a randomized complete block design. Shelves were considered as blocks against a possible vertical temperature gradient. Inoculated apples were incubated in the dark in a controlled-environment room set at 22°C. Temperature was recorded daily on shelves using glass thermometers.

Isolates, except for EP155, were randomly selected for inoculation of each apple within a block. Inoculated fruits were then randomized with respect to shelf position. Lesion length (mm) and width (mm) on each fruit were measured on the 12th day after inoculation. The experiment was terminated after 12 days since isolate interactions within individual fruits became a possibility as the lesions enlarged. Lesion area was determined and standardized by dividing the lesion area induced by each test isolate by the lesion area induced by the standard EP155 on the same apple.

The 101 SO isolates, as well as EP155 and EP713, were ranked in terms of lesion area induced (Minitab 1991). Data also was subjected to General Linear Model (GLM) analysis and Tukey’s HSD mean comparison test (p=0.05) was used to test for differences in mean lesion areas induced by the various isolates (SAS 1985).

Chestnut Inoculations

Eighteen American chestnut sprouts growing in a 10-year-old clearcut in the Savage River State Forest, New Germany, Maryland were selected for inoculation. Sprouts selected did not have *C. parasitica* cankers within the proposed inoculation height on the main stem (0.5m to 3.0m), nor any other obvious large cankers on the stem. Due to the limited number of suitable stems available, all isolates could not be tested. Five of the *C. parasitica* isolates which induced the largest lesions on apple fruit(SO isolates 9, 31, 57, 77, and 86), and five of the isolates which induced the smallest lesions (SO isolates 12, 17, 39b, 69 and 99) were selected, along with the standard virulent EP155, hypovirulent EP713, and a PDAmb agar control.

On May 17, 1991, four inoculation wounds per stem were made by removing circular areas of bark with a sterile, 7-mm diameter core borer. Plugs of inoculum from 7-day-old cultures of *C. parasitica* on PDAmb, or sterile agar plugs as controls, were placed in the wound (mycelium towards the pith) and covered with parafilm. Each chestnut sprout was inoculated with the standard virulent isolate EP155. The remaining three wounds on each stem received the SO isolates, the hypovirulent EP713, or the control in a random distribution. Three repetitions of each isolate, other than EP155, were used. Inoculation wounds were not placed directly beneath one another on the same stem to minimize the possibility of conidia spread by stemflow from one inoculation point to another. Inoculation points were spaced at maximum allowable distances to minimize over-lapping canker growth between or among isolates.

Length (mm) and width (mm) of each canker was measured on August 29, 1991, 104 days after inoculation.

Field observations during canker development indicated that this time period was sufficient to assure that cankers were well established, yet not so large as to girdle the stem or overlap. The linear length and width measurements were converted to area prior to analysis. Since stems were variable in size and shape, lesion area was standardized by dividing the area induced by each isolate by the area induced by EP155 on the same stem, and ranked (Minitab 1991). The ranking was compared to the rankings for linear growth in culture, as well as the ranking for lesion size induced in apple fruit, and tested using Spearman's rank correlation (Minitab 1991). Data also were subjected to GLM analysis and Tukey's HSD mean separation test (p=0.05) (SAS 1985).

RESULTS

Linear Growth in Culture

There was no significant difference in linear growth among blocks (shelves in the incubator), but mean growth between the two replications was significantly different. However, in replication 2, growth of most isolates was poor and erratic, apparently due to laboratory techniques. Therefore, only data from replication one is presented in this paper (see Torsello (1992) for data from replication two).

Few significant differences in linear growth were observed among isolates; complete datasets for all 102 SO isolates are not presented, but have been reported elsewhere (Torsello 1992). Only isolate 68a of the 102 SO isolates grew significantly greater than that of the standard virulent strain EP155. Isolates 81 and 99, as well as the hypovirulent EP713, grew significantly less than EP155. The linear growth of SO isolate 99 and the hypovirulent EP713 were similar and were significantly less than all other isolates. There were no significant differences in growth among the remaining SO isolates.

The general culture morphology of all isolates, except for SO isolate 99, was similar to that of EP155, the standard virulent isolate. The morphology of isolate 99 was similar to that of the known hypovirulent EP713.

Lesion Induction in Apple

Monitoring with thermometers within the controlled environment room revealed no temperature gradient. However, apples on block 2 (shelf 2) had a slightly, but significantly, greater mean lesion area as compared to block 4 (shelf 4). Other comparisons among blocks were non-significant. Scarlet oak isolate 99 induced significantly smaller lesions on the fruit compared to all other isolates, including the standard hypovirulent EP713 and the standard virulent isolate EP155. Only SO isolates 22 and 47 induced significantly larger lesions than the hypovirulent EP713, but the lesions were not significantly different in size from those induced by EP155 (Torsello 1992). Other isolates induced lesions on apples not significantly different from those induced by EP155. The sterile agar plug resulted in minimal browning around each inoculation wound.

Canker Induction in Chestnut

Data from one stem were eliminated due to failure of EP155 to induce lesions, which precluded standardization. Considerable variation in canker size was noted, even among repetitions of the same isolate. Six isolates induced lesions larger than the standard EP155, and five isolates induced cankers smaller than EP155, but differences in lesion area among replications, individual stems, or isolates were not significant (data not presented). Isolate 99 induced the smallest lesion of all isolates. The sterile agar plug resulted in minimal, but measurable, browning around some inoculation wounds.

Ranking of Isolates

The relative ranking, standardized with reference to EP155, of canker size induced on American chestnut by 10 isolates and hypovirulent strain EP713 is shown in Table 1. Rankings from the apple fruit inoculations were very similar to the rankings derived from the chestnut stem inoculations, with a significant (p=.01) Spearman's rank correlation coefficient of 0.874. Likewise, the rankings from the first replication of the linear growth study correlated significantly (r=0.877, p=.01) with the rankings from the chestnut inoculations. However, rankings from the second replication of the linear growth study were not correlated (r =-.009, p=non-sig.) with rankings derived from the chestnut inoculations.

DISCUSSION

In terms of linear growth on agar, there were few differences among the 102 SO isolates of *C. parasitica* (See Torsello 1992 for complete dataset). Only scarlet oak isolate 99 (SO99) and the known hypovirulent EP713 (Anagnostakis 1992) grew significantly less than the standard virulent EP155 in both replications. The general culture morphology of all isolates except SO99 was similar to the phenotype of EP155 in both replications. SO99 exhibited submerged hyphae, reduced fruiting, and a mycelial morphology similar to that of some hypovirulent strains (Anagnostakis 1990). Based on linear growth in culture, we conclude that most of the SO isolates to be more similar to the virulent than to the hypovirulent strains, with the exception of SO99.

With regard to apple fruit, most of the 102 SO isolates induced lesions similar in size to those caused by the standard virulent isolate EP155. However, SO99 induced significantly smaller lesions on the fruit compared to all other isolates, including the standard hypovirulent EP713 and the standard virulent isolate EP155, again indicating characteristics of possible hypovirulence (Fulbright, 1984).

On the stems of American chestnut, considerable variation in canker size was noted, even among repetitions of the same isolate. Six isolates induced lesions larger than the standard EP155, and five isolates induced cankers smaller than EP155 (Table 1), but differences in lesion area among replications, individual stems, or isolates were not significant (Torsello, 1992). The lack of significance in the chestnut stem canker data is attributed in part to a severe drought that occurred during canker development, arresting growth of some inoculated saplings and causing mortality

Table 1—Relative ranking of growth of the 12 *C. parasitica* isolates which were common to all studies, in terms of canker size (ratio) produced on American chestnut, lesion size (ratio) induced in apple fruit, and linear growth (mm) in culture

Chestnut stem			Apple fruit		Linear growth	
ID	Area	Rank	Area	Rank	mm	Rank
99	0.450[a]	1	0.216[b]	1	7.00[c]	1
69	0.508	2	0.716	6	39.25	7
713[d]	0.523	3	0.608	2.5[e]	14.00	2
12	0.585	4	0.618	4	38.75	5
17	0.642	5	0.608	2.5	37.50	3
155[d]	1.000	6	1.000	7	39.00	6
39b	1.126	7	0.638	5	38.00	4
77	1.399	8	1.236	10.5	41.50	8.5
9	1.702	9	1.222	9	44.50	12
86	5.078	10	1.236	10.5	41.50	8.5
31	5.225	11	1.198	8	43.75	11
57	6.252	12	1.300	12	43.00	10
Correlation coefficient:			0.874[f]		0.877	

[a] Ratio of (canker area induced by each isolate) / (canker area induced on the same stem by EP155) 104 days after inoculation.
[b] Ratio of (lesion area induced by each isolate) / (lesion area induced on the same apple fruit by EP155) 12 days after inoculation.
[c] Linear growth (mm) after 7 days on PDAmb.
[d] 155 is the standard virulent isolate (EP155) and 713 is a known hypovirulent isolate (EP713).
[e] The same number (ending in .5) appearing within a column indicates equal ranking.
[f] Spearman's rank correlation coefficient comparing the relative ranking of respective columns to the results of the American chestnut inoculations.

of others, thus reducing the sample size and introducing variability. This variability confounded the critical comparison between the stem inoculation results with the linear growth and/or apple inoculation results, since stem canker production is the ultimate test for hypovirulence. However, it is important to note that, of all the isolates studied, SO99 induced the smallest lesion on American chestnut.

The relative rankings (Table 1) also reveal that the virulence ranking based on apple fruit inoculations were very similar to the rankings derived from the chestnut stem inoculations. Also, the rankings from the first replication of the linear growth study correlated significantly with the rankings from the chestnut inoculations. However, rankings from the second replication of the linear growth study were erratic, and not correlated with rankings derived from the chestnut inoculations, thus confounding any comparisons of our results with those of Bedker (1989). The rankings do support the use of inoculation of apple fruit as a possible means to initially screen large numbers of isolates of *C. parasitica* to estimate virulence (Elliston (1982, 1985; Fulbright, 1984). However, because of the results of replication 2 of the linear growth study, these results do not necessarily support those of Bedker (1989) who suggested that linear growth rate in culture might be used to estimate the virulence of *C. parasitica* isolates in chestnut.

Isolate SO99 should be investigated for the presence of viruses conferring hypovirulence. Hypovirulent isolates of *C. parasitica* have not been reported from hosts other than chestnut. However, results from this study indicate that only a small percentage of the isolates (perhaps 1 percent) from scarlet oak have characteristics of those known to be hypovirulent. If hypovirulence is present within *C. parasitica* on scarlet oaks, even in a small percentage, this population should be considered when formulating biocontrol efforts of chestnut blight on American chestnut involving use of hypovirulence.

However, most of the isolates collected from scarlet oaks across Pennsylvania had characteristics similar to the known virulent isolate, EP155. We concur with Nash and Stambaugh (1982), who found that cankered oaks in the southeastern USA may serve as important reservoirs of virulent *C. parasitica* inoculum, and that virulence (in

American chestnut) of some oak isolates is equal to or greater than that of isolates originally obtained from American chestnut.

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REFERENCES

Anagnostakis, S.L. 1990. Recovery of additional flat mutants of *Cryphonectria parasitica*. Mycologia. 82: 515-518.

Anagnostakis, S.L. 1992. Measuring resistance of chestnut trees to chestnut blight. Canadian Journal of Forest Research. 22: 568-571.

Anagnostakis, S.L.; Aylor, D.E. 1984. The effect of temperature on growth of *Endothia (Cryphonectria) parasitica* in vitro and in vivo. Mycologia. 76: 387-397.

Bedker, P.J. 1989. Hypovirulent strains of *Cryphonectria parasitica* in New Jersey [Abstract]. Phytopathology. 79: 1142.

Elliston, J.E. 1982. Hypovirulence in *Endothia parasitica* and its suggested procedures for its detection and analysis. In: Smith, H.C.; MacDonald, W.L., eds. Proceedings of the American chestnut symposium. Morgantown, WV: West Virginia University.

Elliston, J.E. 1985. Characteristics of dsRNA-free and dsRNA-containing strains of *Endothia parasitica* in relation to hypovirulence. Phytopathology. 75: 151-158.

Fulbright, D.W. 1984. Effect of eliminating dsRNA in hypovirulent *Endothia parasitica*. Phytopathology. 74: 722-724.

Minitab Inc. 1991. Minitab reference manual: release 7. State College, PA: Minitab Inc.

Nash, B.L.; Stambaugh. W.J. 1982. Disease incidence, symptomatology, and vegetative compatibility types of *Endothia parasitica* on chestnut and oak hosts in North Carolina. In: Smith, H.C.; MacDonald, W.L. Proceedings of the American chestnut symposium. Morgantown, WV: West Virginia University: 74-82.

Nash, B.L.; Stambaugh, W.J. 1987. Lack of host specificity in North Carolina isolates of *Endothia parasitica*. Phytopathology. 77: 414-417.

Nash, B.L.; Stambaugh, W.J. 1989. Greater diversity of vegetative compatibility groups of *Cryphonectria parasitica* on scarlet oak in North Carolina. Plant Dis. 73: 416-418.

Nuss, D.L. 1992. Biological control of chestnut blight: an example of virus-mediated attenuation of fungal pathogenesis. Microbiol. Rev. 56: 561-576.

Roane, M.K.; Griffin, G.J.; Elkins, J.R. 1986. Chestnut blight, other Endothia diseases, and the genus *Endothia*. St. Paul, MN: Amer. Phytopath. Soc. Press.

SAS. 1985. SAS user’s guide: statistics. Cary, NC: SAS Institute, Inc. 956 p.

Scibilia, K.L.; Shain, L. 1989. Protection of American chestnut with hypovirulent conidia of *Cryphonectria (Endothia) parasitica*. Plant Dis. 73: 840-843.

Torsello, M.L. 1992. Cryphonectria canker of scarlet oak in Pennsylvania: incidence, severity, and influence on growth. University Park, PA: The Pennsylvania State University, Department of Plant Pathology. 73 p. M.S. thesis.

Torsello, M.L.; Davis, D.D.; Nash, B.L. 1994. Incidence of Cryphonectria parasitica cankers on scarlet oak (*Quercus coccinea*) in Pennsylvania. Plant Dis. 78: 313-315.

THE EFFECT OF SOIL MANGANESE ON JAPANESE LARCH (*LARIX LEPTOLEPIS* SIEB. AND ZUCC.) SEEDLINGS IN THE GREENHOUSE

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Abstract—Preliminary analysis of 9 year old Japanese larch trees and soil subjected to applications of triple ambient annual nitrogen (N) and sulfur (S) deposition revealed elevated available soil and foliar manganese (Mn) levels and decreased growth compared to controls. A greenhouse study was conducted in which Japanese larch seedlings were grown in field collected soil amended with 0, 100, 500, and 1000 mg of Mn as $MnCl_2$ per 8352 cm^3 of soil to determine the role of Mn in these growth differences. Growth was measured for 73 days. Soil samples were analyzed for magnesium (Mg), Mn and pH and foliar samples collected on day 73 were analyzed for Mn. Total chlorophyll concentrations were also determined. Control Japanese larch seedlings had significantly greater mean chlorophyll concentrations than treated seedlings. Japanese larch seedlings responded to increased Mn supply with increased uptake of Mn. Height and diameter growth were not significantly different ($\alpha \geq 0.05$) among the four treatments. However, overall height growth of Japanese larch was 10 percent less in the three treatments compared to the control. These results are supportive of the hypothesis that elevated available soil Mn may have contributed to the observed growth differences between control and treated Japanese larch in the field.

INTRODUCTION

Very little information exists about the potential phytotoxic role Mn may play with regard to forest health and tree nutrition. In the United States, deposition of anthropogenically produced N and S introduces strong mineral acids to the soil which influence soil chemistry. Deleterious effects to forest trees have been attributed primarily to soil changes, such as decreased levels of exchangeable base cations (especially calcium (Ca) and Mg), elevated hydrogen ion concentrations (lower soil pH) and higher levels of toxic aluminum (Joslin and others 1992; Thornton and others 1989). Manganese availability also is increased as a consequence of these soil changes, but has been given little attention (Elamin and Wilcox 1986). Gradual base cation depletion and low pH lead to soil Mn levels that may be detrimental to plant growth (Ohki 1984; Terry and others 1975). Excessive Mn has been associated with disruption of many physiological functions, such as reduced enzyme, hormone and chlorophyll production, inhibition of ATP formation, and reduced respiration (Elamin and Wilcox 1986).

Plant tolerance variations to excessive Mn are large (Kohno and others 1984; Simon and others 1986). Plant tolerance also has been associated with decreased transport of absorbed Mn from roots to leaves (Smith and others 1983). With some species, a reduction in chlorophyll content of the leaves sometimes accompanies the accumulation of toxic concentrations of Mn in the plant (Morgan and others 1976). Toxicity has been attributed to Mn induced Fe deficiency (Smith and others 1983).

Plants may differ considerably within and among species in Mn tolerance due to genetic characteristics and environmental factors such as nutrient availability in the soil. The presence of other ions including Fe, Ca, and Mg can modify Mn uptake (Goss and Carvalho 1992). Maas and others (1969) showed that Ca ions further enhanced the inhibition of Mn uptake by Mg. The mechanism responsible for selective uptake gave rise to the concept of carriers with varying affinities for the elements selectively accumulated. Because none of the effects between Ca, Mg, and Mn can be explained by mutual competition for the same transport site (Moore and others 1961), the regulatory action must result at a site other than the actual absorption site (Maas and others 1969). Manganese appears to function like Ca in maintaining membrane integrity (Maas and others 1968). Foy and others (1969) found increasing the Ca concentration in soil reduced Mn toxicity by reducing Mn uptake by roots or by reducing its transport to stems and leaves.

Ouellette and Dessureaux (1958) reported that excess Mn becomes detrimental only when enough of it moves from the roots to the above ground biomass. Therefore, Mn determination in leaves and stems provides a good indication of toxicity. Ohki (1974) defined critical Mn levels as the concentration in tissue associated with a 10 percent reduction in maximum growth and used this critical level to evaluate response in wheat to Mn. Kohno and others (1984) used the lowest Mn concentration level in the leaves at which toxicity symptoms developed as a more sensitive measure of plant tolerance to Mn. However, critical levels of Mn for various tree species have not been ascertained. If critical level data were available, the evaluation of foliar Mn status of field grown trees could be used as a guideline for diagnosing Mn toxicity.

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Analysis of Japanese larch foliage samples in 1993 taken from watershed 9, an 11.6 ha experimental watershed located 13 Km west of Parsons, West Virginia, revealed elevated Mn concentrations and reduced height growth as a result of annual treatments with 169 kg/ha of ammonium sulfate (Pickens and others 1995). The treatments began in 1987 four years after this watershed was clearcut and root raked and three years after it was planted with 2-0 Japanese larch seedlings at 1.8 x 1.8m spacing. Vegetation existent at the time of clearcutting consisted of mixed low grade hardwoods that had colonized abandoned agricultural fields.

Increased solubility of soil Mn was expected as a result of the ammonium sulfate treatments. The observed increased foliar Mn coupled with the sparse amount of information available on the potential effects of elevated soil Mn to trees prompted this investigation. Although Japanese larch is not now an important commercial tree species in the central hardwood region, it has been used extensively for strip mine reclamation plantings. Interest has also been expressed in converting low grade Appalachian hardwood stands to Japanese larch for fiber production. Watershed 9 where the initial observations of Mn response were made was converted to Japanese larch for this purpose (Kochenderfer and Helvey 1989). The study presented here was designed to evaluate Japanese larch seedling growth responses to various amounts of added Mn, and to determine whether or not Mn toxicity could explain the reduced growth of the treated Japanese larch observed in the field. In particular, we tested the following null hypotheses: (1) Japanese larch growth would not be reduced under the highest soil Mn levels; (2) Foliar Mn levels would not increase with increasing soil Mn levels; (3) Soil Mn levels could not be used to predict foliar Mn levels; (4) Elevated foliar Mn would not interfere with chlorophyll production.

MATERIALS AND METHODS

Soil Collection

Mineral soil obtained from Watershed 9, an experimental watershed operated by the USDA Forest Service in north central West Virginia, was collected on May 11, 1993 and used as the growth medium for the seedlings in this study. The soil was a Calvin channery silt loam (loamy skeletal, mixed, mesic Typic Dystrochrept weathered from the Hampshire sandstone formation (Losche and Beverage 1967). This soil had received 19 ammonium sulfate applications over the previous 7 years prior to collection. Under heavy acidic inputs, soils high in potentially available Mn release large amounts of this element (Kazda and Zvacek 1989). Manganese shows a strong association with Fe in rocks and soil, and is found adsorbed onto the surface of fine grained soil minerals. The Calvin channery silt loam used in this study was developed in uplands and weathered from sandstone and acid red shale (Losche and Beverage 1967). The pH of this soil was 4.22. The concentration of Mn in these sedimentary rocks has been reported to be in the range of 170-600 mg/kg. The average content of United States surface soils is 560 mg/kg (Gilkes and McKenzie 1988).

Soil was collected by extracting the A horizon mineral soil from three adjacent soil pits located on the northern boundary of watershed 9. The soil was mixed thoroughly and sieved to remove stones (> 1 cm^2), and placed into 48 clear, acrylic tube planters (45.2 cm tall x 15.2 cm in diameter). Each planter contained approximately 8352 cm^3 of soil. The bottom of each plastic planter was covered with nylon mesh fabric to allow drainage and covered with a porous polyethylene cap for additional support. Each planting cylinder was wrapped with aluminum foil to reduce soil heating and prevent algal growth.

Study Design

Seedlings were selected randomly for planting from a bundle of 200 2-0 stock Japanese larch seedlings (Saratoga Tree Nursery, New York DEC, Saratoga Springs, NY, seedlot 811, seed orchard #13). At planting, seedling heights (nearest 0.1 cm, meter stick) from root collar to tip of the dominant terminal and diameters (nearest 0.1 mm, Doall Electronic Digital caliper, Maxcal, USA) at 1 cm above the root collar were measured. All seedlings were breaking dormancy at the time of planting. All planters received 2 liter of distilled water at the time of planting. The planters were arranged in blocks of 12, placed in a greenhouse and the seedlings were allowed to grow from May 18 until July 8, 1993. Each block contained three planters of each treatment including controls. There were four treatment blocks in this randomized block design for each species. Dead or dying seedlings were replaced prior to July 8.

The soil in the planters was amended by adding 100 mg (treatment 2), 500 mg (treatment 3) and 1000 mg (treatment 4) of Mn in the form of $MnCl_2$ to each of three planters in each block (24 total planters per treatment) on July 8. An additional 24 planters served as controls. The $MnCl_2$ was dissolved in 1 liter of deionized water. The bottle used to add the Mn solution was rinsed three times with 50 ml (total) of deionized water and this water also was added to each planter. Each planter received 271 ml of water two times per week (2.97 cm/week). This amount was sufficient to keep soil moisture replenished. Growth measurements commenced on July 8 and ended on September 18. Seedling height and diameter data represent net growth during this 73 day period.

Twenty-four hours after treatment a soil sample was collected from the top 7-10 cm in each planter. Soil was collected in paper bags, air dried, and analyzed for 0.01 molar $SrCl_2$ extractable (Joslin and Wolfe 1989) Mg and Mn by atomic absorption spectrophotometry. Atomic absorption spectrophotometry analysis was performed within 48 hours of extraction. Soil pH was determined in 1:1 water/soil paste (Black 1964).

The experiment was terminated on September 18, 1993. Seedling heights and diameters were measured. Foliar samples for chemical analysis of Japanese larch were obtained by removing and compositing all needle whorls on two lateral branches on each seedling. All foliar samples were rinsed in deionized water, placed in paper bags, and oven dried at 105 °C for 24 hours. Samples were then ground in a Wiley mill (Thomas Scientific, USA) fitted with a 20 um screen and submitted for ICP (Inductively Coupled Plasma Emission Spectroscopy) analysis to the Agricultural Analytical Services Laboratory (College of Agricultural Sciences, The Pennsylvania State University, University Park, PA 16802) to determine aluminum (Al), boron (B), Ca, copper (Cu), iron (Fe), potassium (K), Mg, Mn, sodium (Na), phosphorus (P), and zinc (Zn) (Dahlquist and Knoll 1978). Only Ca, Mg, Mn and Fe are reported here.

Quality assurance/control for all analysis included analytical duplicates and standard reference materials. Precision was determined by analyzing one duplicate soil and foliar sample with every 12 samples. Differences between the chemistry of the sample and its split were not significantly different from zero.

Total chlorophyll (chlorophyll a+b) concentration was measured on a randomly selected subsample of four Japanese larch trees for each of the four treatments. Preparation, extraction and determination of chlorophyll followed the method of Arnon (1948).

Data Analysis

All statistical analyses were performed using SAS statistical packages (SAS Institute 1985), following a randomized block design. Within a treatment, analysis of variance showed no significant differences for each chemical parameter among blocks, and values were then pooled by treatment. General linear regression was performed using treatment means of foliar and soil measurements.

RESULTS AND DISCUSSION

The height of Japanese larch seedlings was consistently reduced in the three treatments, but diameter growth was not (Figure 1). None of the changes was statistically significant, but a greater than 10 percent height growth decrease was observed, for all treatments which at least one author has considered important (Ohki 1985). Acceptance or rejection of null hypothesis one was thus somewhat uncertain. Statistical significance in height growth across treatments may have been achieved with a larger sample size.

Control soil had significantly greater hydrogen ion concentration and significantly lower Mn compared to all other treatments (Table 1). Mn concentrations followed treatment 4 > treatment 3 > treatment 2 > control (all significantly different) for the soil Mn values. Soil Mg did not differ significantly among treatments.

Comparisons of foliar Mn for Japanese larch are given in Figure 2. All treatments were significantly different with the relative magnitudes of foliar Mn concentrations matching the treatment Mn additions. Japanese larch seedlings responded to increased Mn supply with increased uptake and foliar Mn concentrations. The relationship between initial soil Mn supply and foliar Mn concentration for Japanese larch is given in Figure 3. The relationship is significant (p=0.0001) and the two variables have an R^2 = 0.50. Thus, null hypotheses two and three were rejected.

The presence of other ions can modify the uptake of Mn from solution. In studying Mn toxicity to melons, Simon and others (1986) found that competition exists between Mg and Mn for specific binding sites. However, Maas and others (1969) explained the effects of Ca and Mg, on Mn as Mn absorption being non-competitively inhibited by Mg and stimulated by Ca. No differences existed for soil Mg among

Table 1—Initial soil sample mean concentrations (0.01 M $SrCl_2$ extractable Mn and Mg; pH in water) and statistical comparisons among treatments

Soil parameter	Control	Trt. 2	Trt. 3	Trt. 4
Mn (meq/100g)	0.012a	0.127b	0.374c	0.633d
pH (pH units)	4.22	4.01	3.95	3.83
pH (meq H^+/100g)	0.060a	0.098b	0.110b	0.148c
Mg (meq/100g)	0.074a	0.080a	0.085a	0.089a

Soil parameters with different letters indicate significant difference among treatments at $\alpha \leq 0.05$; n=24; pH was not tested.

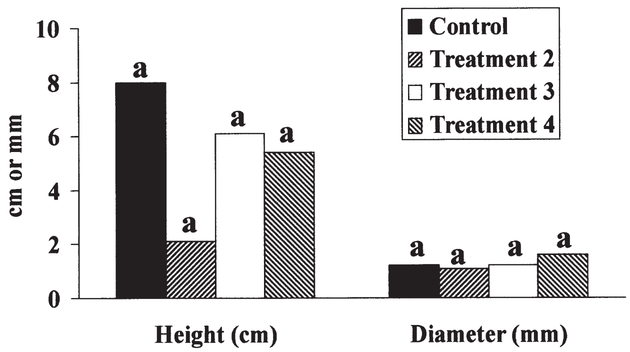

Figure 1—Mean height and diameter growth for Japanese larch from July 8, 1993 to September 18, 1993. Different letters above bars indicate significant differences at alpha = 0.05.

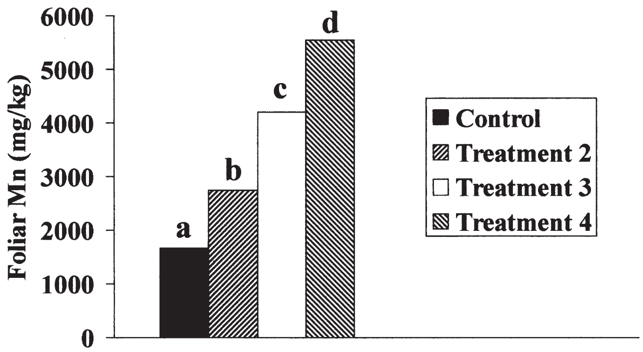

Figure 2—Japanese larch foliar Mn concentration comparisons among treatments. Different letters above bars indicate significant differences at alpha = 0.05.

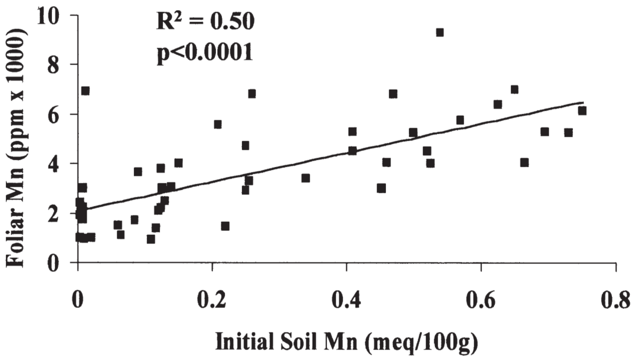

Figure 3—Regression relationship between initial soil Mn concentrations and Japanese larch foliar Mn at the end of the study for all treatments including control.

treatments in this study. For Japanese larch seedlings in this study, soil Mn concentrations and foliar Ca, Mg, and K were not consistently nor strongly correlated.

Foliar observations were recorded for all trees on June 15, July 28, August 18, and September 17, 1993. Japanese larch foliage did not exhibit any visual deficiency/toxicity symptoms regardless of treatment throughout the experiment.

Average chlorophyll concentrations (Figure 4) were greater in the control Japanese larch seedlings when compared to the other three treatments (reject null hypothesis four). Ohki (1985) found that excessive Mn in solution culture (500 mg/l) resulted in reductions of chlorophyll concentration in wheat. Others have reported chlorophyll synthesis inhibition by excessive Mn (Clairmont and others 1986; Csatorday and others 1984). The probable site of Mn inhibition is a Fe requiring step following the insertion of Mg in the tetrapyrrole ring of the chlorophyll molecule (Clairmont and others 1986). No correlations were found between soil Mn and foliar Fe, nor between foliar Mn and foliar Fe. There were no significant differences in foliar Fe among treatments.

Ohki (1985) defined the Mn critical toxicity level as the foliar concentration associated with a 10 percent growth

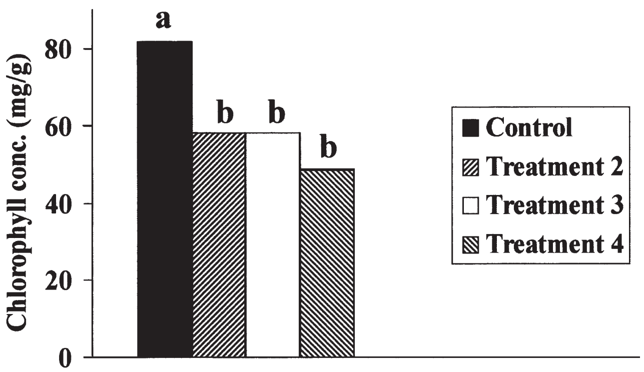

Figure 4—Japanese larch chlorophyll concentrations in mg/g. Different letters above bars indicate significant differences at alpha = 0.05.

reduction. Using the relative growth of the control as the maximum amount achievable for this experimental time period. A 10 percent reduction for Japanese larch would have resulted in a 7.2 cm height increase and a 1.1 mm diameter increase. Larch height growth in treatments 2, 3 and 4 all fell below this value, averaging 2.1 cm, 6.1 cm, and 5.4 cm, respectively. Diameter growth did not exhibit a response. These growth changes occurred in the absence of visual foliar symptoms. The critical toxicity level as defined by Ohki (1985) may have some merit in predicting growth changes in Japanese larch. Although no observed Mn symptoms in larch seedlings were recorded, the trends of decreasing growth and chlorophyll concentration along with significantly greater levels of foliar Mn with soil Mn additions suggested that some inhibitory effects to the photosynthetic process may have occurred.

SUMMARY AND CONCLUSIONS

Under acidic soil conditions, soil macronutrients such as Ca and Mg and potentially toxic micronutrients such as Mn become more mobile, enhancing their uptake by tree roots. Sensitivity of most forest trees to elevated Mn remains unknown. Under conditions of low soil pH and low soil-available Mg and Ca commonly found in extremely acidic forest soils, Mn toxicity could occur. Soil Mn levels were a good predictor of Japanese larch foliar Mn, at least for the range of Mn availability used in this study. The results of this study suggest that Mn toxicity and subsequent foliar chlorophyll reductions may play a role in the reduced height growth observed in the N and S acidified Japanese larch plantings reported by Pickens and others (1995) and Kochenderfer and others (1995) on an acidified watershed. Further investigation of the impacts of increasing Mn availability and toxicity to other tree species that commonly grow in relatively Mn rich, extremely acid edaphic environments in the central hardwoods region would seem to be prudent.

ACKNOWLEDGMENTS

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REFERENCES

Arnon, E.I. 1948. Copper enzymes in isolated chloroplasts. Polyphenoloxidase in *Beta vulgaris*. P.L. Physiol. 24(1): 1-15.

Black, C.A., ed. 1964. Methods of soil analysis. Part 1. Chemical and microbiological properties. 1st ed. Agronomy 9. Part 2. Madison, WI: American Society of Agronomy, Inc.: 111-113.

Clairmont, K.B.; Hagar, W.G.; Davis, E.A. 1986. Manganese toxicity to chlorophyll synthesis in tobacco callus. Pl. Physiol. 80: 291-293.

Csatorday, K.; Gombos, Z.; Szalontai, B. 1984. Mn^{2+} and Ca^{2+} toxicity in chlorophyll biosynthesis. Proc. Natl. Acad. Sci. USA. 81: 476-478.

Dahlquist, R.L.; Knoll, J.W. 1978. Inductively coupled plasma atomic emission spectrometer: analysis of biological materials and major, trace, and ultra-trace elements. Applied Spect. 32(1): 1-29.

Elamin, O.M.; Wilcox, G.E. 1986. Effect of magnesium and manganese nutrition on muskmelon growth and manganese toxicity. J. Am. Soc. Hort. Sci. 111: 582-587.

Foy, C.D.; Fleming, A.L.; Armiger, W.H. 1969. Differential tolerance of cotton varieties to excess manganese. Agron. J. 61: 690-694.

Gilkes, R.J.; McKenzie, R.M. 1988. Geochemistry and mineralogy of manganese in soils. In: Graham, R.D.; Hannam, R.J.; Uren, N.C., eds. Manganese in soils and plants. The Netherlands: Kluwer Academic Publishers: 23-34.

Goss, M.J.; Carvalho, M.J. 1992. Manganese toxicity: the significance of magnesium for the sensitivity of wheat plants. Plant and Soil. 139: 91-98.

Joslin, J.D.; Kelly, J.M.; VanMiegroet, H. 1992. Soil chemistry and nutrition of North American spruce-fir stands: evidence for recent change. Journal of Environmental Quality. 21: 12-30.

Joslin, J.D.; Wolfe, M.H. 1989. Aluminum effects on northern red oak seedling growth in six forest soil horizons. Soil Sci. Soc. Am. J. 53: 274-281.

Kazda, M.; Zvacek, L. 1989. Aluminum and manganese and their relation to calcium in soil solution and needles in three Norway spruce (*Picea abies* L. Karst) stands of upper Austria. Plant and Soil. 114: 257-267.

Kochenderfer, J.N.; Helvey, J.D. 1989. Hydrologic impacts of mechanized site preparation in the central Appalachians. In: Kochenderfer, J.N.; Crews, J.T.; Smith, H.C. Effects of fertilization on the growth and development of a Japanese larch plantation in West Virginia: the 7th central hardwood forest conference; 1995 March 5-8; Carbondale, IL. Res. Pap. NE-700. U.S. Department of Agriculture, Forest Service.

Kohno, Y.; Foy, C.D.; Fleming, A.L.; Krizek, D.T. 1984. Effect of Mn concentration on the growth and distribution of Mn and Fe in two bush bean cultivars grown in solution culture. J. Pl. Nutr. 7(1-5): 547-566.

Losche, C.K.; Beverage, W.W. 1967. Soil survey of Tucker County and part of northern Randolph County, WV. U.S. Soil Conservation Service, Forest Service, and WV Agricultural Experiment Station. Washington, DC: U.S. Government Printing Office. 78 p.

Maas, E.V.; Moore, D.P.; Mason, B.J. 1968. Manganese absorption by excised barley roots. Pl. Physiol. 43: 527-530.

Maas, E.V.; Moore, D.P.; Mason, B.J. 1969. Influence of calcium and magnesium on manganese absorption. Pl. Physiol. 44: 796-800.

Moore, D.P.; Jacobson, J.; Overstreet, R. 1961. Uptake of calcium by excised barley roots. Pl. Physiol. 36: 53-57.

Morgan, P.W.; Taylor, D.M.; Joham, H.E. 1976. Manipulation of IAA-oxidase activity and auxin-deficiency symptoms in intact cotton plants with manganese nutrition. Physiol. Pl. 37: 149-156.

Ohki, K. 1974. Manganese nutrition of cotton under two boron levels. II. Critical Mn levels. Agron. J. 66: 572-575.

Ohki, K. 1984. Manganese deficiency and toxicity effects on growth, development, and nutrient composition in wheat. Agron. J. 76: 213-218.

Ohki, K. 1985. Manganese deficiency and toxicity effects on photosynthesis, chlorophyll, and transpiration in wheat. Crop Sci. 25: 187-191.

Ouellette, G.J.; Dessureaux, L. 1958. Chemical composition of alfalfa as related to degree of tolerance to manganese and aluminum. Can. J. Pl. Sci. 38: 206-214.

Pickens, C.J.; Sharpe, W.E.; Edwards, P.J. 1995. The effects of doubling annual N and S deposition on foliage and soil chemistry and growth of Japanese larch (*Larix leptolepis* Sieb. And Zucc.) in north central West Virginia. In: 10th central hardwood forest conference; March 5-8; Morgantown, WV: 131-140.

SAS Institute Inc. 1985. SAS user's guide: statistics. Version 5 ed. Cary, NC: SAS Institute Inc.

Simon, J.E.; Wilcox, G.E.; Simini, M. [and others]. 1986. Identification of manganese toxicity and magnesium deficiency on melons grown in low-pH soils. Hort Sci. 21: 1383-1386.

Smith, G.S.; Edmeades, D.C.; Upsdell, M. 1983. Manganese status of New Zealand pastures 1. Toxicity in ryegrass, white clover and lucerne. New Zealand J. Agri. Res. 26(2): 215-221.

Terry, N.; Evans, P.S.; Thomas, D.E. 1975. Manganese toxic effects on leaf cell multiplication and expansion and on dry matter yield of sugar beets. Crop Sci. 15: 205-208.

Thornton, F.C.; Schaedle, M.; Raynal, D.J. 1989. Tolerance of red oak and American and European beech seedlings to aluminum. Journal of Environmental Quality. 18: 541-545.

NORTHERN RED OAK GROWTH REPONSE TO CLIMATE AND INDUSTRIAL AIR POLLUTION IN WESTERN PENNSYLVANIA

J.R. McClenahen, D.D. Davis, and R.J. Hutnik[1]

Abstract—Northern red oak (*Quercus rubra* L.) radial growth response over time and space was examined along an inferred air pollution gradient on Laurel Ridge, a northeast - southwest anticlinal ridge, in relation to local and historically varying air pollutant emissions from coal-burning power generation and iron production within the greater Johnstown area (Conemaugh Gap) in west-central Pennsylvania. The specific objectives were to determine the effects, if any, of industrial air pollution (primarily sulfur dioxide) on: (1) the relationship of tree growth and climate, (2) tree growth as measured by basal area increment and, (3) the separate growth responses that may be related to long-term air pollution from Johnstown industry and more recent coal-burning power generation. On the east side of Conemaugh Gap, the city of Johnstown has been a major iron production center from the 1920's until 1977. Two power generating stations on the west side of Conemaugh Gap began operation in 1950 and 1970, respectively. We collected pairs of tree cores from at least 20 canopy northern red oaks in 11 mostly ridge top stands ranging from 11 km downwind to 34 km upwind of Conemaugh Gap. Indexed tree-ring chronologies for each stand were derived by standard dendrochronological techniques, and these chronologies were modeled with temperature and precipitation variables by stepwise multiple regression. The resulting growth-climate models indicated that growth of northern red oaks on Laurel Ridge most consistently and positively responded to July precipitation of the growth year, and warmer than normal summer temperatures in the preceding year. In general, upwind control stands showed the strongest relationship with climate, while growth in stands nearer to and downwind of Conemaugh Gap exhibited weaker or virtually no relationship with climate. The spatial patterns of basal area growth rates generally mimicked those of the growth-climate models; an increasing growth rate was evident with distance away from Conemaugh Gap. We conclude that growth of canopy northern red oaks on the upper slopes of Laurel Ridge near to, and downwind of, Conemaugh Gap exhibited anomalous long-term low growth rates and climatic decoupling. The most likely cause of this growth anomaly is historical industrial emissions from Johnstown, with little or no indication of an additional growth impact when the power generating stations went on-line. Whether red oak growth and climatic sensitivity recover in the Conemaugh Gap area as a result of improving air quality is under investigation.

INTRODUCTION

The health of forests and natural ecosystems is a major focus of public agencies and private forest owners. This focus has emerged from an awareness that native and introduced pests, air quality, changing land use patterns, forest management practices, climatic extremes, and many other potential stressors may be increasingly affecting forest health (Smith 1990). Measuring forest health is a major challenge in part because the concept lacks specific definition (Kolb and others 1994). There is no concensus about which critical indicators to measure and there is a deficiency of data about the normal range of many proposed health indicators. A further complication is the difficulty in identifying causal factors after forest or tree declines occur (Manion and Lechance 1992).

Tree growth variation, as manifested in annual ring widths, is one approach for identifying stress or declining vigor (McClenahen 1995). Some studies have shown marked changes in tree-ring response to climate and (or) growth reductions in a variety of species in the presence of high pollutant dose (Sutherland 1990, Thompson 1981, Fox and others 1986). A few instances of growth-climate decoupling or growth decreases have been reported in the absence of unusual mortality or decline symptoms (McClenahen and Dochinger 1985, Phipps and Whiton 1988, Puckett 1982, Tryon and others 1979). Further, dead or symptomatic trees may reveal anomalous tree-ring responses compared to healthy trees (McClenahen 1995), but means for detecting predisposing stress that may portend declining health have not been specifically developed.

This study focuses on the potentially interacting effects of climate variation and regional air pollution on tree growth. The research was conducted on Laurel Ridge in the vicinity of Johnstown, a major industrial city, in westcentral Pennsylvania, USA (Fig. 1).

Laurel Ridge affords partial isolation of pollutant emissions from industrial Johnstown to the east and two local power generating stations to the west. Periods of non-overlapping emissions from these two sources also provided a temporal isolation of emissions. Thus, a unique opportunity was available to study the separate and combined effects of air pollution from these two pollutant sources on tree growth.

We compared the regional patterns of tree growth-climate responses and growth rates of northern red oaks (*Quercus rubra* L.) across an inferred gradient of industrial emissions from Johnstown and the two power generating stations

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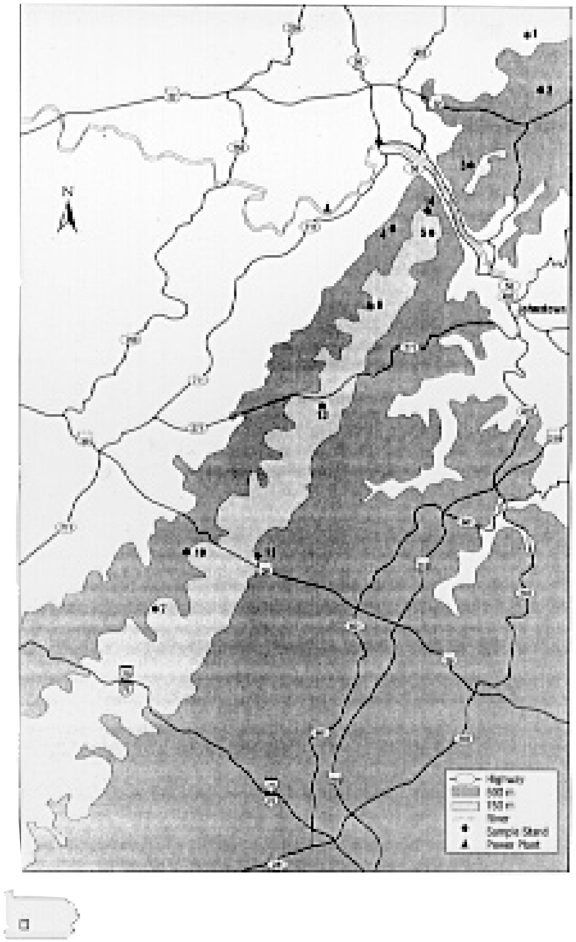

Figure 1—Study area and numbered sample stands. Scale is approximately 1 cm = 5 km.

(Fig. 1). The specific objectives of this study are to determine the effects, if any, of air pollution within the Conemaugh Gap (Greater Johnstown) industrial area on: (1) the relationship of tree growth to climate, (2) tree growth as measured by basal area increment and, (3) the separate growth impacts that may be related to long-term air pollution from Johnstown industry and more recent power generation from local coal-burning power stations.

METHODS

Study Area

Tree coring sites were located on Laurel Ridge, a NE to SW-oriented anticlinal ridge 700-800 m in elevation. The area lies within the Allegheny Plateaus Physiographic Province in portions of Cambria, Indiana, Westmoreland and Somerset Counties (Fig. 1). The climate is continental with a mean growing season temperature of 9.8°C and annual precipitation averaging 104 cm. Forests on Laurel Ridge are second and third growth mixed hardwoods. Oaks (*Quercus* spp.) dominate the ridgetops and drier slopes. Coves and moist slopes have mixtures of *Prunus serotina* Ehrh., *Acer rubrum* L., *Liriodendron tulipifera* L., *Quercus* spp., and others.

Historically, sulfur dioxide (SO_2) has been the primary air pollutant in the Conemaugh Gap area since the late 19th century (Fig. 2). On the southeast side of Laurel Ridge at the eastern side of Conemaugh Gap, Johnstown iron and steel production, and attendant coke plants, began operation in 1872, increasing exponentially from about the 1920's through the 1960's (Brown 1989). Beginning about 1977, a drastic decline in iron production and cleaner technology resulted in reduced SO_2 emissions (Brown 1989).

Two coal-fired power plants are located on the northwest side of Laurel Ridge near the western side of Conemaugh Gap (Fig. 1). The Seward plant began operation in 1950 with one 64 mw unit and a stack height of 70 m. A second unit added in 1957 increased generating capacity to 137 mw. Stack height was increased to 183 m in 1977. The Conemaugh station, with a stack height of 305 m, started one 850 mw unit in 1970 and a second unit of equal capacity in 1971. SO_2 emissions from two other large power stations to the northwest apparently do not appreciably impinge the area of this study (Hutnik and others 1989).

Sample Stands

Beginning in 1987, data were collected within 11 mostly ridge top or upper slope, mature, closed canopy, even-aged, northern red oak stands (1-10 ha). Ideally, stands lacking partial cutting were chosen, but many had received light thinnings as part of forest management practices. Stands 7, 10 and 11, to the southwest and upwind of pollutant sources, served as controls in regard to pollutant emissions from the power generating stations and Johnstown industries to the northeast (Fig. 1), although transport of sulfur and nitrogen from sources outside the region and subsequent deposition onto Laurel Ridge is well-recognized (e.g., Pierson and others 1989).

Soils on the study sites are well to moderately well drained, very stony loams and silt loams, predominantly Typic Dystrochrepts, occasionally including some Typic Hapludults and Aquic Fragiudults.

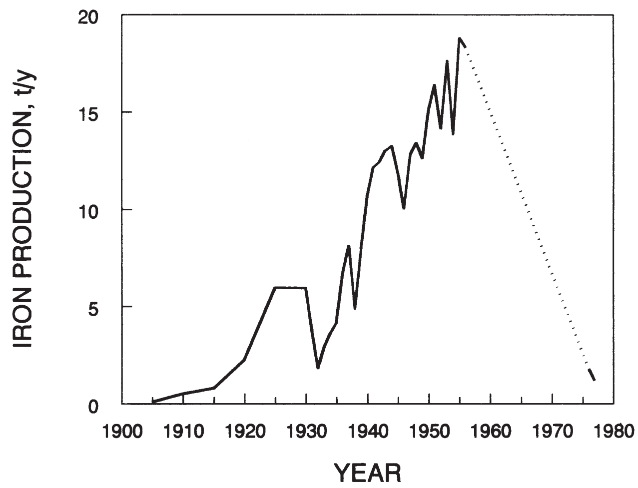

Figure 2—Historical iron production at Johnstown (Data from Brown, 1989).

Tree-Ring Sampling

Pairs of 5 mm diameter cores were extracted at 1.3 m height from boles of at least 20 dominant and codominant red oaks in each of the 11 stands. Trees without dieback or mechanical crown or bole injury were selected. Cores were air-dried, glued into wooden holders, and surfaced with successively finer sanding papers. Visual crossdating was performed, ring widths were measured to 0.01 mm precision, and crossdating was verified using computer program COFECHA (Holmes 1985).

Statistical Analyses

The ring-width series of each core within a stand was standardized to create stationary time series and to remove autocorrelated trend largely associated with age and competition. This was achieved by a double-detrending procedure followed by autoregressive modelling, using Cook's (1987) method and computer program ARSTAN. First, a linear or negative exponential trend line was fit to each series and indices were computed from the ratio of curve value to ring width. The second detrending was achieved by fitting a spline curve to the initial indices and recalculating. Lastly, an appropriate autoregressive (AR) model was fit to each detrended series, and to the mean stand chronology. This step yielded two types of mean stand chronologies: the mean indices after AR modeling of individual series (residual chronology), and the arstan chronology consisting of the residual chronology plus that portion of the original trend (persistence) common to most individual series. Thus, the residual chronology represents the mean "white noise" stand chronology. The arstan chronology additionally contains the common (pooled) persistence that may be associated with some exogenous factors affecting growth.

Stepwise multiple regression models were developed to relate red oak residual and arstan growth indices for each of the eleven stands to regional climate (objective one). Total monthly precipitation and mean monthly temperature data were each averaged from Historical Climatic Network database stations at Johnstown, New Castle and Uniontown (Boden 1987, Bradley and others 1985). A 17-month climate year (prior May through September of the growth year) comprised the 34 total variables in the stepwise regression analyses with residual and arstan indices for each stand. A common calibration period of 1920-1969 (51 years; pre-Conemaugh power station emissions) was used. Stepwise regression climate calibrations were performed using PRECON (Fritts and others 1991). A value of F = 4.00 was used for adding variables. The resulting models were evaluated in terms of lack of fit, Durbin-Watson statistic, variance inflation factors, and the residuals plots. A series of tests described by Cook and others (1987) were employed to verify the calibration models using independent climate data from 1970-1986, and to address objective three. Actual and predicted indices were examined for similarity of pattern by simple linear correlation, for significant predictive bias by t-tests of the residuals for difference from zero, and for indication of predictive skill by reduction of error (RE).

Annual basal area increment (BAI) was computed for each ring width series from core radii and the ring widths using computer program AREA (Phipps and Fields 1988). Radial growth trend differences among stands were examined by comparing the slopes of linear regressions fitted to the individual stand cumulative mean BAI data (objective two).

RESULTS

Chronology Characteristics

Chronology statistics were initially examined for indication of comparative stand growth disturbance histories (Table 1). Autoregression parameter differences, in terms of the model coefficients (Φ) and percent explained variation (R^2), between the mean of individual series models and the pooled model for all series imply a spatially varied disturbance history within the stand, while similar parameter values suggest spatially homogeneous disturbances (Cook 1988). In this regard, stands 1, 2, 10, and 11 show relatively heterogeneous disturbances by virtue of large differences in both R^2 and Φ. In fact, charred snags indicated the occurrence of fire in stand 1; stand 2 had a light partial cutting approximately 20 years earlier. However, growth disturbance histories appeared fairly uniform within most stands. In fact, the remaining seven stands were similar in common persistence (pooled R^2) and pooled AR coefficients (Φ) (Table 1). These stands had 30-45 pct common growth persistence, indicative of even-aged forests lacking significant heterogenous disturbances.

These chronologies portray the historic natural disturbances, stand development trends, and direct human interventions such as cutting and fire that partly molded present forests on Laurel Ridge, but they show no specific relationship to the Conemaugh Gap area.

Growth-Climate Models

Red oak growth relationships with climate varied considerably among stands, with the percentage of growth explained by climate variables ranging from 9.3 to 49.1. Mean monthly temperature variables entered the regressions more frequently than precipitation variables (Fig. 3). Growing season temperature, especially in the prior year, had a generally positive growth relationship, although the specific temperature variables were not very consistent among stands or between residual and arstan chronology models within stands. Some of these inconsistencies could be related to variation in site and tree ages, although sample tree ages were mostly similar among stands (Smith and Rennie 1995). The single most consistent and important climate variable appearing in models was July precipitation in the growth year, which was always positive in its relationship.

The strength of the climate relationships among stands, indicated by the percentage of explained variation (R^2(pct), adjusted for number of variables), revealed some striking spatial patterns (Fig. 4). Growth in the southernmost, upwind stands (control stands 7, 10, 11) was consistently and relatively strongly coupled with climate. This was reflected in both the residual and arstan chronology models. These three stands were originally selected as controls by virtue of their distance and upwind direction from the Conemaugh

Table 1—Red oak chronology statistics; Φ_1 and Φ_2 are the series mean or pooled autoregression (AR) coefficients for models up to AR(2); R^2 is the variation explained by autoregression averaged over individual series or from pooled autoregression

				Autoregressive model					
				Series means			Pooled		
Stand	Chron. length	No. series	AR order	Φ_1	Φ_2	R^2	Φ_1	Φ_2	R^2
	No. yrs.					*Pct.*			*Pct.*
1	103	40	1	.529	-	29.6	.409	-	16.7
2	114	40	1	.489	-	25.8	.431	-	18.6
3	73	40	1	.584	-	35.9	.604	-	36.5
4	107	40	1	.617	-	39.1	.675	-	45.6
5	91	40	2	.533	.064	36.7	.502	.184	40.0
6	141	46	1	.540	-	30.1	.549	-	30.2
7	141	42	3	.478	.047	31.5	.384	.122	33.8
10	132	31	1	.518	-	29.4	.434	-	18.8
11	129	42	1	.469	-	23.2	.394	-	15.5
13	140	40	2	.575	.034	37.5	.518	.158	39.3
14	75	40	1	.541	-	30.8	.537	-	28.9

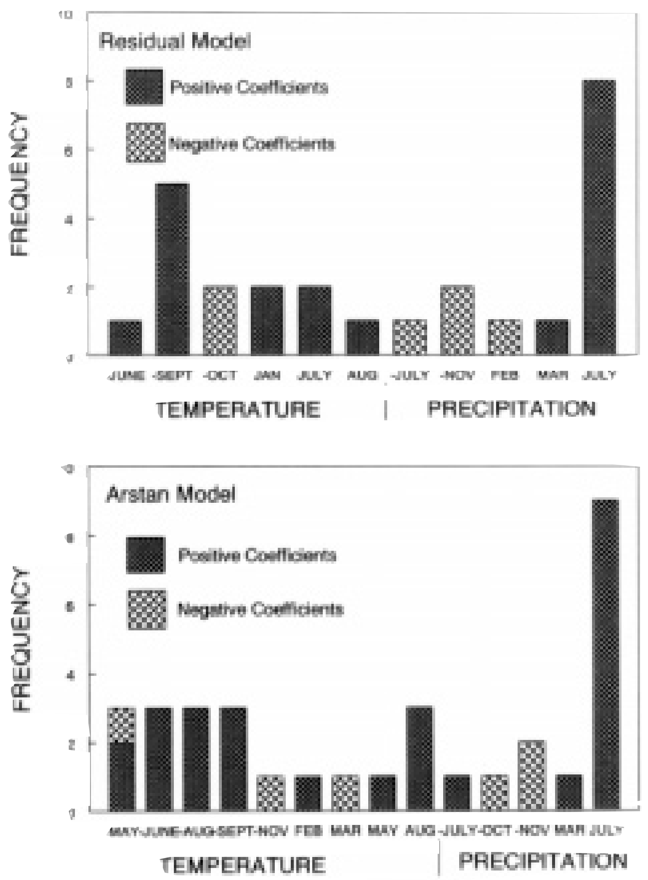

Figure 3—Frequencies of monthly mean temperature and monthly total precipitation variables appearing in calibration models of the eleven stands using residual and arstan chronologies. Numbers indicate stands. Negative signs preceding months indicate prior year.

Gap area. Compared to the control stands, residual chronology models of stands near Conemaugh Gap were notably weaker; while common (pooled) growth persistence in the arstan chronologies dramatically strengthened the growth-climate relationships in certain of these stands (viz., stands 1, 2, 4, 13). However, stands 5 and 6 exhibited virtually no growth response to climate variation.

Evaluation of the growth-climate calibration models for each stand indicated that all were significant in terms of correlated patterns of actual and predicted indices, and all exhibited some predictive skill (Table 2). However, these criteria revealed weaker models for stands 5 and 6 and, to a lesser extent, stands 1, 3 and 14.

The calibration model verifications using the period 1970-1986 showed uniformly poor predictive potential (Table 2). Although none had significant bias, there was no evidence that predicted indices correlated with observed indices, and no models exhibited predictive skill. Thus, although potentially useful calibration models were obtained for many of the stand chronologies, none were adequate for predicting growth in the post-1969 period. Further, calibrations based on the entire 1920-1986 period for each stand chronology yielded consistently lower percentages of explained variation than the 1920-1969 calibrations. Calibration of the control chronology (mean of stands 7, 10 and 11) with climate for the entire 1920-1986 period identified only two significant variables, July precipitation and prior April temperature, and it explained less than half of the growth variation (21.9 pct) explained by the 1920-1969 calibration model. These results portray a regionally

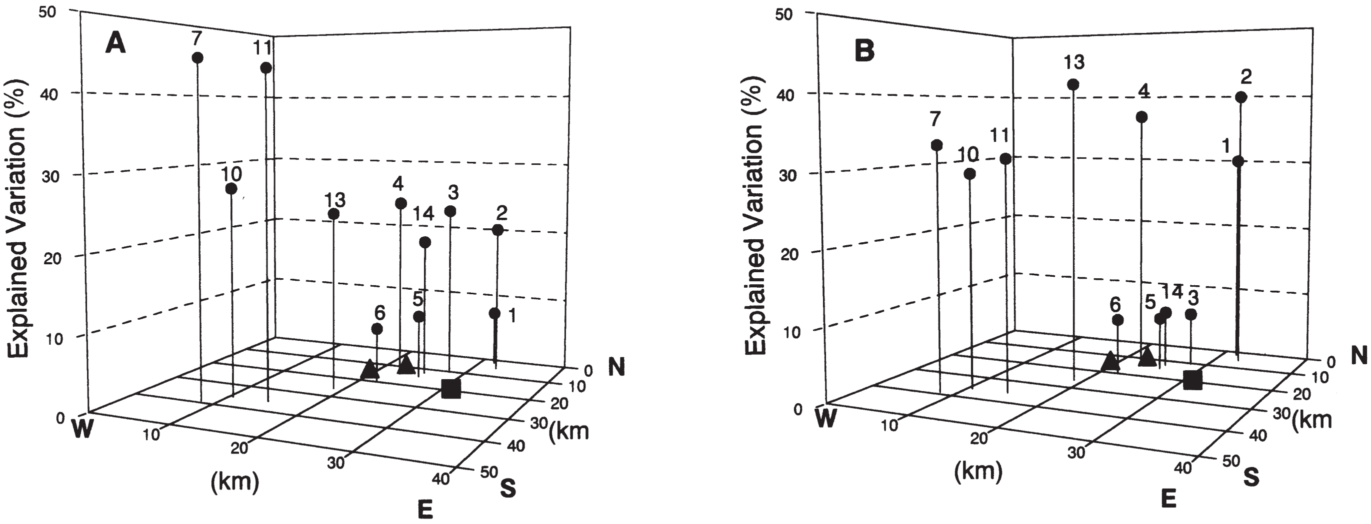

Figure 4—Geographical patterns of percentages of explained variation ($R^2_{adj.}$) for climatic calibration models based on (A) residual and (B) arstan tree-ring chronologies. Triangles indicate power station locations, the square indicates the location of Johnstown.

Table 2—Comparison of arstan and residual indices-based climate model performance for 1920-1969 (calibration period) and 1970-1986 (verification period); the simple correlation coefficient (r) measures the correspondence in pattern between actual and predicted indices; bias is the mean residual, with significant values an indication of over- or under-prediction; reduction of error (RE) >0 suggests predictive skill over using the mean index value (no significance test available)

		Residual			Arstan		
Stand	Period	r	Bias	RE	r	Bias	RE
1	Calib.	0.38*		0.09	0.60*		0.36
	Verif.	0.24	0.018	0.05	0.01	0.040	-0.84
2	Calib.	0.50*		0.25	0.69*		0.47
	Verif.	0.01	-0.001	-0.32	0.02	0.039	-0.41
3	Calib.	0.52*		0.27	0.32*		0.10
	Verif.	0.34	-0.058	-0.01	0.10	-0.056	-0.32
4	Calib.	0.53*		0.28	0.66*		0.43
	Verif.	0.16	-0.014	-0.05	0.10	-0.045	-0.63
5	Calib.	0.32*		0.11	0.31*		0.09
	Verif.	-0.20	0.032	-0.48	-0.12	-0.019	-0.19
6	Calib.	0.31*		0.09	0.31*		0.10
	Verif.	0.23	0.039	0.02	0.17	0.030	0.01
7	Calib.	0.70*		0.49	0.61*		0.37
	Verif.	0.12	0.020	-0.17	0.28	-0.057	-0.12
10	Calib.	0.63*		0.39	0.66*		0.44
	Verif.	0.08	-0.036	-0.11	0.06	-0.030	-0.63
11	Calib.	0.69*		0.48	0.60*		0.36
	Verif.	0.17	0.054	-0.07	0.21	0.038	-0.01
13	Calib.	0.54*		0.29	0.69*		0.47
	Verif.	0.13	0.022	-0.20	0.00	0.013	-1.43
14	Calib.	0.47*		0.22	0.31*	0.10	
	Verif.	0.26	-0.019	-0.01	0.07	0.008	-0.12

*Significant at P ≤ 0.05.

unstable growth response to climate in addition to the spatial differences noted above.

Basal Area Growth

Linear regressions closely and accurately described the cumulative mean basal area increments between 1920 and 1986 for 10 of the 11 stands ($R^2 \geq 0.98$). Stand 14 had a curvilinear growth pattern (linear $R^2 = 0.90$) due to a growth release around 1955 and is excluded from the following discussion.

The spatial pattern of basal area growth rates (Fig. 5) generally mimics that of the growth-climate model strengths depicted in Figure 4. As a group, stands 2, 3, 5, 6 and 13 exhibited lower growth rates than stands 1, 4, 7, 10 and 11. These growth rates depict a generally increasing growth gradient away from the Conemaugh Gap area. An exception is stand 4 on the northwest base of Laurel Ridge near the power plants, which exhibited a growth rate comparable to stands remote from Conemaugh Gap despite its proximity to the power stations. The physiographic location of stand 4 would probably shield it from pollutants originating at Johnstown (Fig. 1). Stands 5 and 6 were notable for lacking a growth relationship with climate, and the mean sustained growth rates of these stands (and stand 2) implies a history of non-climatic growth stress. There was no physical evidence within these stands to indicate a possible explanation.

DISCUSSION AND CONCLUSIONS

The 1920-1969 climate response calibration models show that above normal July rainfall in the growth year, and warmer than normal summer temperatures in the preceding year, promote red oak growth in these stands. The arstan chronologies were generally better correlated with climate than the residual chronologies, indicating a general persistence in the effect of climate on growth that may be a physiological characteristic of the species. However, this

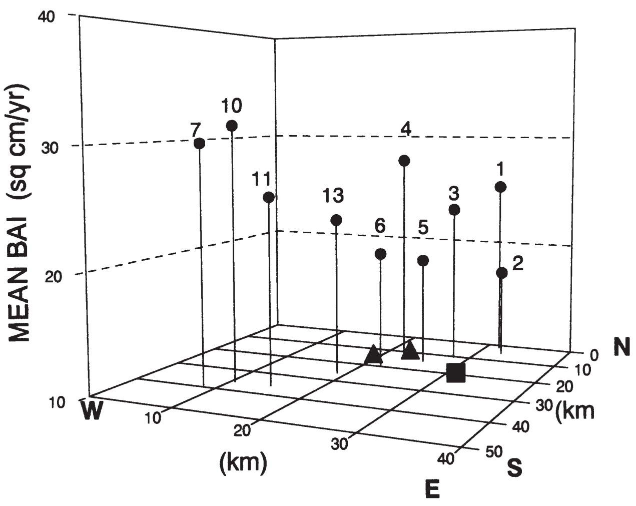

Figure 5—Geographical patterns of stand cumulative mean basal area increment (BAI) growth rates. Numbers indicate stands. Triangles show power station locations and the square indicates the location of Johnstown.

persistence was especially evident in chronologies from the vicinity of Conemaugh Gap, suggesting the presence of additional exogenous environmental stress in that area.

The climatic response is generally consistent with the ecophysiological nature of oak growth. Oaks tend to occupy relatively warm, dry habitats. Their ring-porous characteristic results in a dependency on stored energy from the previous year for earlywood formation, while latewood production relies more upon current season growing conditions. July precipitation is clearly the most critical climatic factor in growth of red oaks on Laurel Ridge.

Red oaks within the northeastern portion of the study area generally exhibited an anomalous uncoupling of the climate-growth relationship that generally predates the Conemaugh, Seward or other local power stations. Red oaks have also had notably low growth rates within this same locality during pre- and post-operational times. These findings raise the possibility that anthropogenic disturbances, perhaps including fire, destructive logging practices, and industrial emissions from Jownstown, may have strongly impacted red oak growth in the locality at least two decades prior to 1986.

Stands 5 and 6 show the most consistent lack of climatic growth response (and the lowest basal area growth rates), followed by stands 1, 2, 3 and 14. Notably, these appear to lie within the Johnstown airshed. In particular, stands 5 and 6 are situated on the ridge top directly above the city. Although geographically within the same general area, stands 1, 4, 13 and 14 were somewhat better correlated with climate. However, stands 1 and 13 are more remote from Johnstown, while stands 14 and 4 lie respectively on the mid-slope and at the base of the opposing side of Laurel Ridge from the city, and may have been relatively sheltered from Johnstown emissions. Although a similar sheltering effect might be true for stands 1, 2 and 3, Johnstown emissions could be carried to these stands through Conemaugh Gap by prevailing winds.

Evaluation of 1920-1969 growth-climate calibrations, and comparisons with models derived for 1920-1986, indicated that the relationship of red oak growth on Laurel Ridge has not remained stable. Precisely when this transition took place cannot be established from our analyses, but it is evident in the post-1969 period at all sites. Comparatively high frequencies of insect defoliations and droughts during 1970-1992 resulted in extensive red oak mortality on Laurel Ridge between 1990-1992 (McClenahen and others 1997). Comparisons of red oak and tuliptree chronologies indicate that relatively few oak defoliations occurred in the study area between 1930 and 1957, while the more recent, recorded defoliations were confirmed (unpublished data of the authors). These droughts and increasing defoliation events, a number of which fell within the chronology period 1969-1986, may have contributed greatly to the general red oak growth-climate uncoupling.

Summarizing these results: (1) the importance of common persistence in the arstan calibrations may reflect an interactive effect of climate and other environmental stress within ridgetop and upper slope stands near the Conemaugh Gap during power plant pre-operational times, (2) there was a long-term depressed growth-climate relationship for most stands within the Conemaugh Gap area, especially on the ridgetop above Johnstown, (3) the comparative structures of the AR models mitigate against differences in endogenous disturbance histories as a sufficient explanation for these anomalous spatial patterns of growth-climate response and, (4) stands most likely exposed to Johnstown industrial emissions exhibited comparatively lower basal area growth trends.

We conclude that growth of canopy northern red oaks on the upper slopes of Laurel Ridge near to, and downwind of, Conemaugh Gap exhibited an anomalous long-term low growth rate and more pronounced climatic decoupling. The most likely cause of this growth anomaly is historical industrial emissions from Johnstown. There is little indication that sulfur emissions from local power stations established mostly in the late 1960's and early 1970's had an additional growth impact. Whether comparative red oak growth and climatic sensitivity recover in the Conemaugh Gap area as a result of improved air quality in Johnstown (and reduced power plant emissions) is presently under investigation.

ACKNOWLEDGMENTS

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REFERENCES

Boden, T.A. 1987. United States historical climatology network (HCN) serial temperatures and precipitation data. Oak Ridge, TN: Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory.

Bradley, R.S.; Kelly, P.M.; Jones, P.D. [and others]. 1985. A climatic data bank for northeastern hemisphere land areas, 1851-1980. Washington, DC: Carbon Dioxide Res. Div., U.S. Dept. Energy.

Brown, S.A. 1989. Historic resources study: Cambria Iron Company, U.S. Department of Interior, National Park Service. 513 p.

Cook, E.R. 1987. The decomposition of tree-ring series for environmental studies. Tree-Ring Bulletin. 47: 37-59.

Cook, E.R. 1988. A tree-ring analysis of red spruce in the Southern Appalachian Mountains. In: Van Deusen, P.C., ed. Analysis of Great Smoky Mountain red spruce tree ring data. Gen. Tech. Rep. SO-69. U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station: 6-19.

Cook, E.R.; Johnson, A.H.; Blasing, T.J. 1987. Forest decline: modeling the effect of climate and tree rings. Tree Physiol. 3: 27-40.

Fox, C.A.; Kincaid, W.B.; Nash III, T.H. [and others]. 1986. Tree-ring variation in western larch (*Larix occidentalis*) exposed to sulfur dioxide emissions. Canadian Journal of Forest Research. 16: 283-292.

Fritts, H.C.; Vaganov, E.A.; Sviderskaya, I.V.; Shashkin, A.V. 1991. Climatic variation and tree-ring structure in conifers: empirical and mechanistic models of tree-ring width, number of cells, cell size, cell-wall thickness and wood density. Climate Res. 1: 97-116.

Holmes, R.L. 1983. Computer-assisted quality control in tree-ring dating and measurement. Tree-Ring Bull. 43: 69-75.

Hutnik, R.J.; Davis, D.D.; McClenahen, J.R. 1989. Evaluation of vegetation near coal-burning power plants in southwestern Pennsylvania. Part I. Sulfur content of foliage. Journal Air Pollut. Contr. Assoc. 39: 1440-1443.

Kolb, T.E.; Wagner, M.R.; Covington, W.W. 1994. Concepts of forest health: utilitarian and ecosystem perspectives. Journal of Forestry. 92:10-15.

Manion, P.D.; Lachance, D. 1992. Forest decline concepts: an overview. In: Manion, P.D.; Lachance, D., eds. Forest decline concepts. St. Paul, MN: The American Phytopathological Society: 181-190.

McClenahen, J.R. 1995. Potential dendroecological approaches to understanding forest decline, In: Lewis, T.E., ed. Tree rings as indicators of ecosystem health. Boca Raton, FL: CRC Press: 59-79.

McClenahen, J.R.; Dochinger, L.S. 1985. Tree-ring response of white oak to climate and air pollution near the Ohio River Valley. Journal of Environmental Quality. 14: 274-280.

McClenahen, J.R.; Hutnik, R.J.; Davis, D.D. 1997. Patterns of northern red oak growth and mortality in western Pennsylvania. In: Pallardy, S.G.; Cecich, R.A.; Garrett, H.G.; Johnson, P.S., eds. Proceedings 11th central hardwood forest conference. Tech. Rep. NC-188. North Central Forest Experiment Station: 386-399.

Phipps, R.L.; Fields, M.L. 1988. Basal area increment programs. Reston, VA: Tree Ring Laboratory, U.S. Geol. Surv.

Phipps, R.L.; Whiton, J.C. 1988. Decline in long-term growth trends of white oak. Canadian Journal of Forest Research. 18: 24-32.

Pierson, W.R.; Brachaczek, W.; Gorse, R., Jr. [and others]. 1989. Atmospheric acidity measurements on Allegheny Mountain and the origins of ambient acidity in the Northeastern United States. Atmos. Environ. 23: 431-459.

Puckett, L.J. 1982. Acid rain, air pollution, and tree growth in southeastern New York. Journal of Environmental Quality. 11: 376-381.

Smith, E.R.; Rennie, J.C. 1991. Changes in the relationship between annual tree growth and climatic variables for four hardwood species. In: McCormick, L.H.; Gottschalk, K.W., eds. Proceedings 8th central hardwoods forest conference. PA: Penn State University, State College: 224-246.

Smith, W.H. 1990. The health of North American forests: stress and risk assessment. Journal of Forestry. 88: 32-35.

Sutherland, E.K.; Martin, B. 1990. Growth response of *Pseudotsuga Menziesii* to air pollution from copper smelting. Canadian Journal of Forest Research. 20: 1020-1030.

Thompson, M.A. 1981. Tree rings and air pollution: a case study on *Pinus monophylla* growing in east-central Nevada. Environ. Pollut. 26: 251-266.

Tryon, E.H.; Gallegly, M.E.; Raphelson, M.S.; Linsky, B. 1979. Coal-fired power plant emissions affect the growth of trees. In: 72nd annual meeting of the air pollution control association. Cincinnati, OH.

Genetics / Tree Improvement

SURVIVAL AND GROWTH OF A *QUERCUS RUBRA* REGENERATION COHORT DURING FIVE YEARS FOLLOWING MASTING

Kim C. Steiner and Brian J. Joyce[1]

Post-harvest regeneration of oak in the mid-Atlantic region is often limited by an insufficiency of numbers and size of advance reproduction seedlings. In Pennsylvania, this problem is sometimes (but not always) associated with a dense ground cover of ferns that appears to inhibit tree seedling growth, both before and after harvest. An unnaturally dense population of white-tailed deer is at least partly to blame for weak levels of oak advance reproduction, and the paucity of low woody vegetation occasioned by deer browsing may contribute to the abundance of fern growth in some stands. This suggests the hypothesis that, in many mature hardwood stands, deer browsing has tipped the competitive balance between ferns and understory woody plants (including tree regeneration), with the result that the scarcity and small size of woody plant regeneration is both a cause and an effect of profuse fern growth.

If this hypothesis is correct, then what happens to seedlings that germinate under a fern cover and are protected from browsing? Can they survive and grow sufficiently to eventually dominate the fern, or do both deer browsing and fern competition need to be controlled to permit tree seedlings to establish? In this study we followed two “reproduction cohorts” for five years, from the time of masting in early autumn of 1990 until early autumn 1995. The objectives were 1) to document losses at various stages of the reproduction process and 2) to determine the ability of unbrowsed seedlings to grow out of dense fern ground cover.

The study was performed in two stands located 2.6 miles from one another on Moshannon State Forest in central Pennsylvania. Both stands were about 75 to 80 years of age at the beginning of the study. In the “Smith Road” stand northern red oak (NRO) represented 68 percent of the 96 ft^2/acre basal area, and the forest floor was 43 percent covered with herbaceous vegetation (bracken fern, *Pteridium aquilinum,* and hayscented fern, *Dennstaedtia punctilobula*) and 19 percent covered with woody vegetation (predominantly *Vaccinium* spp.). In the “Firebreak” stand NRO represented 48 percent of the 139 ft^2/acre basal area, and the forest floor was 65 percent covered with herbaceous vegetation (predominantly hayscented fern) with negligible woody ground cover. Site indices of the two stands are 63 and 73 ft, respectively. Advance reproduction of NRO for these stands in any given year has been one to a few thousand, with no seedlings observed in excess of 30 cm in height. More information on these stands may be found in Steiner and others (1993).

The study reported here was prompted by heavy mast crops in both stands in 1990. Acorn production, autumn removals by vertebrates, and viability loss caused by insect or disease injury were determined by sampling as described by Steiner (1995). Winter removals by vertebrates were determined indirectly by the difference between potential germinants in November and actual germinants in May of the following year. Immediate post-germination losses (to both insects and vertebrate predation) were determined by successive visits in May and June. In June of 1991, 400 (200 per stand) first-year NRO seedlings were located, mapped, labeled, and measured for height. Study seedlings were fairly evenly scattered over an area of about 4 acres at each site but all were located within fern ground cover. To permit an examination of growth and survival in the presence and absence of deer browsing, half (100) of the study seedlings in each stand were protected by means of a 3.25 x 36-inch tubular plastic net (sold as a “Rigid Seedling Protector Tube”) supported by two bamboo stakes. This material casts a measured 11 percent shade. Seedling heights and survival were determined 1.0, 1.5, 2.0, and 4.5 years after germination (up to 5.0 years following masting). Stand canopies remained essentially intact over the course of the study.

Survival of the reproduction cohort from fall 1990 until fall 1995 is shown in Table 1, and height growth of protected and unprotected seedlings is shown in Table 2. In general, the two stands were very similar in their overall pattern of acorn loss and seedling mortality, so only mean values will be presented. Of the average regeneration cohort of 141,075 filled acorns per acre, 90 percent were destroyed or consumed before germination in spring 1991 (as quantified in May), and 99 percent were lost by the end of the fifth growing season. Pre-germination losses to insects or disease (not tabulated separately in Table 1) accounted for an average of only 2.2 percent of the acorn crop. Some unquantified losses to insects occurred immediately after germination, but total cohort mortality attributable to insects or disease was no more than 11.2 percent and probably less.

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Table 1—Survival over 5 years of NRO reproduction cohorts in two Pennsylvania stands, beginning at masting in 1990

		Smith Road		Firebreak		Mean	
	Year	*Number*	*Percent*	*Number*	*Percent*	*Number*	*Percent*
Fall 1990[a]	0	198,510	100	83,640	100	141,075	100
May 1991[b]	0.5	23,390	11.78	5,130	6.13	14,260	10.11
June 1991[c]		2,548	1.28	667	0.80	1,608	1.14
Spring 1992	1.5	1,631	0.82	360	0.43	995	0.71
Fall 1992	2.0	1,554	0.78	340	0.41	947	0.67
Spring 1993	2.5	1,249	0.63	240	0.004	360	0.26

[a] Production of filled acorns (with naturally abscised caps), representing initial size of the reproduction cohort.
[b] Percentage germination after losses to insects, disease, and consumption by vertebrates.
[c] Seedling survival after immediate post-germination losses to insects and disease, and consumption by vertebrates. Subsequent survival figures are based on marked, unprotected seedlings selected in June 1991.

Table 2—Heights (cm) of advance reproduction of NRO in two Pennsylvania stands in June of first growing season (when protectors were installed) and after 1, 2, and 5 growing seasons, protected and unprotected against browsing by white-tailed deer

	Smith Road		Firebreak	
Seedling age	Unprotected	Protected	Unprotected	Protected
Years				
0	10.40 b	9.04 a	12.67 c	13.33 c
1	10.48 a	10.69 a	11.38 a	14.93 b
2	11.61 a	12.36 a	12.44 a	15.88 b
5	10.85 a	12.72 a	10.00 a	15.28 b

Means within a row not sharing the letter notation are significantly different at $P<0.05$.

Fall and winter consumption (or removals) of acorns by vertebrates accounted for an average of 87.7 percent of the cohort. Among the animals known to consume NRO acorns in Pennsylvania are white-tailed deer, black bear, gray squirrel, white-footed mice, wild turkey, and ruffed grouse. However, deer were probably the major consumer of acorns in these stands for reasons discussed by Steiner (1995). Of 14,260 germinated seedlings per acre, only 995 (7.0 percent) were still alive after 12 months and only 360 (2.5 percent) were surviving after five growing seasons.

From June 1991 (when seedling protectors were installed) until the termination of the study in September 1995, survival was 72.3 percent for protected seedlings and 15.8 percent for unprotected seedlings (a significant difference at $P<0.05$), indicating approximately 28 percent mortality over five growing seasons from causes other than deer browsing and approximately 56 percent additional mortality from deer browsing. Seedlings exposed to deer browsing exhibited no net height growth between 1991 and 1995 and even became significantly shorter at Firebreak. Seedlings protected from deer browsing grew a small (but statistically significant) amount before age 2 years, but the seedlings were no taller at age 5 than at age 2. Of 400 marked seedlings, the tallest after five growing seasons was only 28 cm in height, and no seedling emerged from the fern cover.

The following are our conclusions:

- Despite an unusually large crop of acorns in both stands, advance reproduction after five years was inadequate in terms of quantity and size of seedlings.
- Survival of the reproduction cohort as a function of time since masting resembles a negative power function: only 10.1 percent of the cohort still survived at approximately the time of germination (7 months after masting), 0.7 percent survived at seedling age 1 (19 months after masting), and 0.3 percent survived at the end of the study (60 months after masting). In other words, losses occurred very rapidly at first and more and more slowly as time went on.
- The failure of advance reproduction to develop in these stands is overwhelmingly attributable to mammal and bird consumption of *acorns*, since the majority (87.7 percent) of the reproduction cohort was lost in this manner. Circumstantial evidence indicates that deer are the major single factor in this loss. By contrast, only a small fraction of the entire reproduction cohort (0.6 percent in this study) was lost because of browsing of s*eedlings* after mid-summer of the first year.
- Although seedlings protected from deer had 72 percent survival after five growing seasons, they exhibited little

increase in height after June of the first year and none after age 2. In oak stands with a heavy ground cover of fern, deer fencing alone will not permit the development of strong advance reproduction. At a minimum, control of the ferns in addition to deer fencing will be necessary if the goal is to establish advance reproduction of NRO. A light thinning of the canopy may be necessary as well.

REFERENCES

Steiner, K.C. 1995. Autumn predation of northern red oak seed crops. In: Proceedings, 10th central hardwood forest conference. Gen. Tech. Rep. NE-197. U.S. Department of Agriculture, Forest Service: 489-494.

Steiner, K.C.; Abrams, M.D.; Bowersox, T.W. 1993. Advance reproduction and other stand characteristics in Pennsylvania and French stands of northern red oak. In: Proceedings, 9th central hardwood forest conference. Gen. Tech. Rep. NC-161. U.S. Department of Agriculture, Forest Service: 473-483.

CHARACTERISTICS OF NORTHERN RED OAK SEEDLINGS GROWN BY FAMILY IN A TENNESSEE NURSERY

Stacy A. Lay and Scott E. Schlarbaum[1]

Abstract—Thirty-eight northern red oak (*Quercus rubra* L.) families were grown for 1 year in a commercial tree nursery in eastern Tennessee, and analyzed for first-order lateral root, height, diameter, and flush growth. Growth was negatively impacted by irregular fertilization and irrigation practices as well as heavy rains during the growing season. Average height and diameter growth were smaller than previously published standards for minimum oak planting stock size of 50 cm in height and 8 mm in root collar diameter (RCD). Provenance effects on nursery seedling growth were nonsignificant for progreny that was one generation removed from the original seed source, except for number of flushes during the growing season. Progeny from related mother trees, i.e., half-siblings, generally did not exhibit similar growth performance. Among-family differences in seedling growth suggests that nurseries should identify superior seed sources by evaluating the performance of progeny in the nursery. Root collar diameter was the best indicator of both height and number of first-order lateral roots (FOLR). Certain families had relatively high distributions of seedlings with above average RCD growth and low within-family variation in RCD. These results indicate that family mean should not be the sole criterion for selection of superior families. Additional nursery evaluations will identify mother trees consistently producing progeny with low numbers of FOLR, small heights, and small RCDs. These trees should be deleted from future seed collections or rogued, if in a seed orchard.

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FIELD PERFORMANCE OF IN VITRO PROPAGATED WHITE ASH MICROPLANTS

J.W. Van Sambeek, John E. Preece, and James J. Zaczek[1]

Abstract—White ash (*Fraxinus americana* L.) can routinely be propagated by in vitro axillary shoot proliferation and in vitro rooting of microshoots; however, no reports exist on performance and clonal variation following field planting of microplants. To obtain preliminary estimates of among and within clonal variations, we established a small planting with twelve white ash clones in 1992. Ten non-stratified seed from fifteen individual tree collections were cut and germinated in vitro on a medium consisting of Murashige and Skoog (MS) salts and organics, thidiazuron, 6-benzyladenine, indole-3-butyric acid (IBA), 2 to 3 percent sucrose, and 0.7 percent agar. In vitro germinants exhibiting high axillary shoot proliferation rates were repeatedly subcultured to produce microshoots for in vitro rooting experiments.

Microshoots were pulsed for 1 week on quarter-strength MS medium supplemented with various combinations of 1 to 5 uM of IBA and naphthaleneacetic acid. Auxin-pulsed microshoots were then transferred to 0.25X MS medium without plant growth regulators where most microshoots produced up to seven adventitious roots within 1 to 2 weeks. When 6 to 8 weeks old, microplants were transplanted to soil in rootrainers and acclimatized under mist to greenhouse conditions. In the greenhouse, most microplants showed one flush of new shoot growth and extensive root development from the adventitious roots produced during the in vitro rooting phase. Microplants were overwintered in a cooler, then moved to a shade house for a month, and planted as in-leaf containerized stock in June 1992 at the SIU Tree Improvement Center in Jackson County, Illinois.

Overall the microplants averaged 2.7 adventitious roots per microshoot following in vitro rooting. Microplants from Family 14 averaged more than 3.2 adventitious roots compared to only 1.4 adventitious roots for Family 13 with the fewest roots. Average height for the original in-leaf planting stock ranged from 5 to 50 cm following field planting. Most microplants produced little additional height growth in the field the first growing season. Early in the third growing season, many microplants produced abnormal leaves and lateral shoots presumably in response to herbicide drift that resulted in multiple short shoots from the axillary nodes and narrow leaflet blades on short leaves. Net height growth averaged less than 20 cm for the third growing season. Severity of symptoms declined in the fourth growing season and most trees had recovered by the fifth growing season. Net height growth is averaging more than 70 cm per year and individual tree heights range from 0.3 to 5.0 m after five years. Multiple regression analysis of fifth year height indicated that the number of adventitious roots and severity of deer browse damage were not related to fifth year tree height; however, putative herbicide damage negatively affected growth.

Most clones have retained similar height rankings from the second growing season through the sixth growing season. Exceptions are Clone 14-10, one of the shortest clones at planting and now one of the tallest after six years, and Clone 15-2, the tallest clone at planting and now in the shortest one-third of the clones after six years. Clone 99 was one of the shortest clones at planting and is now above average in height. Interestingly, Clone 99 originated from organogenic callus, produced thin microshoots in vitro, and had a low in vitro rooting percentage. Initially, variation for height among clones was more than six times that of within clonal variation. After the six years, variation among clones was only twice that of the within clone variation.

Survival of the microplants of all clones after six years was between 70 and 100 percent except for Clone 6-3 where all microplants had died. Microplants of Clone 6-3 produced the fewest adventitious roots during the in vitro rooting period and had the lowest survival during greenhouse acclimatization. In 1997 the basal stem diameter averaged 6.3 cm for all the microplants. Microplants from clones of Family 2, 6, and 14 averaged 7.8 cm in diameter and were 3.8 cm larger than the microplants from Family 5 and 13, the families with the slowest growing microplants. Substantial variation existed among the clones within some families. For example, the basal diameter of Clone 6-1, one of the fastest growing clones, averaged 9.9 cm compared to only 5.7 cm for Clone 6-6 both from Family 6. Likewise, the basal diameter for Clone 4-6, another one of the fastest growing clones, averaged 7.8 cm compared to 4.3 and 4.7 cm for Clones 4-4 and 4-7 from the same single tree collection.

In conclusion, in vitro propagated microplants of white ash planted as in-leaf containerized stock can be successfully established in field plantings. Within clone variation for growth was significantly reduced when compared to among clonal variation. Additional field plantings using microplants from white ash exhibiting a wider range of in vitro axillary shoot proliferation rates and more clones per family are needed to test these preliminary results.

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Posters

SOIL SAMPLING ON SURFACE MINED SPOILS: SYSTEMATIC VS. SYSTEMATIC-COMPOSITE VS. RANDOM

William R. Thomas, Matthew Pelkki and James Ringe[1]

Abstract—When sampling soils, there is a balance between complete and adequate description of the resource and the sampling effort. Any technique that can reduce the cost of sampling without reducing its descriptive value is worthwhile. This problem is especially relevant when dealing with surface mine spoil. Mining procedures result in spoils with greater heterogeneity than agricultural soil or naturally developed forest soils. Most soil sampling techniques have been developed with intensively managed annual crops in mind, and the ones that deal with surface mine lands allow a composite sample to represent from 5 to 20 acres (Evangelou and Barnhisel 1981). However, research plots are much smaller and verification of soil attributes requires greater precision. Three different sampling techniques (systematic, systematic-composite, and random) were used on translocated surface mine spoil in eastern Kentucky and evaluated for similarities in their ability to describe soil characteristics.

METHODS

The spoil was moved from above the coal seam and placed into overburden piles which were then moved again and deposited into research plots. This translocation caused a mixing of the original topsoil and parent rock material. Based on this expected heterogeneity, we felt that systematic sampling would provide the best description of the soil attributes. From this baseline we compared systematic-composite and random sampling, which reduced the costs of sampling.

The systematic sampling consisted of five samples taken from each of nine research plots. A composite sample was taken from the five systematic samples and a separate random sample was taken from each plot. All samples were submitted for analysis of eleven parameters: organic matter, phosphorus, potassium, calcium, magnesium, pH, nitrate nitrogen, soluble salts, total nitrogen, sodium, and water holding capacity. A 95 percent confidence interval was established for the eleven parameters based on the systematic sample. If systematic-composite or random sampling can provide means for the eleven parameters that are within the 95 percent confidence interval for the systematic sample, these lower cost methods are preferable.

RESULTS AND DISCUSSION

While the outcome can be influenced by the soil parameters chosen, these parameters are typical in surface mine reclamation. Of the eleven parameters measured on the nine plots, three of the systematic-composite samples and eight of the random samples fell out of the systematic sample confidence interval (Table 1). The results indicate that a significant time and cost savings (Table 2) can be realized if systematic-composite sampling is used. Random samples offer a lower degree of statistical precision while only providing marginal cost savings over systematic-composite samples.

REFERENCES

Evangelou, V.P.; Barnhisel, R.I. 1981. Sampling surface mine lands before and after mining. Ext. Publ. AGR-41. University of Kentucky.

Table 1—Soil sampling parameters falling outside the 95 percent confidence interval established by the systematic sampling

Research plot	Systematic composite	Random
1	None	Organic matter
		Calcium
		Total nitrogen
2	None	None
3	None	None
4	None	None
5	None	Organic matter
6	Sodium	Sodium
	Water holding capacity	Water holding capacity
7	None	None
8	None	Phosphorus
9	Water holding capacity	Water holding capacity

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Table 2—Soil sampling costs associated with the various soil sampling approaches on a per sample and per plot (20 m^2) basis

Sample method	Field[a]	lab	total	Number of samples	Total cost
	----- *Cost per sample* ----				
Systematic	$1.67	$4	$5.67	5	$28.35
Systematic composite	$2.50	$4	$6.50	1	6.50
Random	$1.67	$4	$5.67	1	5.67

[a] Field costs are based on a wage of $10/hr.

QUAKING ASPEN EMERGENCE AND INITIAL SURVIVAL UNDER DIFFERENT RELATIVE HUMIDITY, MOISTURE, AND SEED PLACEMENT TREATMENTS

L.A. Ahlswede and T.W. Bowersox[1]

Abstract—Quaking aspen (*Populus tremuloides* Michx.) is the most widely distributed tree species in North America. When present in a stand, quaking aspen can be a prolific root sprouter, especially after a disturbance. This seems to be the primary mode of reproduction for the species. A better understanding of the sexual reproduction of quaking aspen is needed to ensure genetic diversity and colonization of new areas.

A study was designed to examine the effects of different relative humidity, moisture and seed placement treatments on the emergence and initial survival of quaking aspen in an environmental chamber. There were a total of six trials in the environmental chamber, each lasting for fourteen days. Either a low (60 percent) or high (90 percent) relative humidity was selected for each trial. A constant temperature of 65° F was maintained for all trials. Three moisture treatments were no additional water, 5 ml every other day, and 10 ml every other day. Seed placement in potting material was at the surface, at 5 mm depth, and at 10 mm depth. Emergent counts were taken after 72, 144, 240 and 336 hours.

Analysis of variance of the environmental chamber study showed that there were significant differences between relative humidity, water, and seed placement treatments. On average, the treatments that had the greatest number of quaking aspen emergents develop and the highest survival were high humidity, 10 ml of water, and seed placement at the surface. This appears to demonstrate the importance of moisture for the establishment of quaking aspen seed.

The moisture and seed placement treatments were applied to a clearcut that was irrigated weekly with 5 cm of treated wastewater. Quaking aspen seeds were planted one day after an irrigation cycle. Moisture treatments of no additional water, 5 ml every other day, and 10 ml every other day were administered. All quaking aspen seed plantings received a second irrigation cycle after six days. Seed placement treatments were the same as in the environmental chamber. Emergents were counted after seven and fourteen days.

Under these conditions, there were no differences in the number of emergents that developed or fourteen-day survival among moisture treatments. However, there was a significant difference among seed placement treatments. Seeds placed at the surface had an average of 83 emergents develop and survive after fourteen days. Seeds placed at the 5 mm and 10 mm depth had an average of 32 and 4 emergents develop and survive, respectively. These studies indicate the opportunity to establish quaking aspen seedlings is greatest when seeds at the soil surface have sufficient moisture. Additional studies are needed to determine precise moisture requirements for the field germination and initial survival of quaking aspen.

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STRATEGIES FOR IMPROVING ESTABLISHMENT AND PRODUCTIVITY OF HARDWOODS PLANTED ON MARGINAL AGRICULTURAL LANDS IN SOUTHERN ILLINOIS

John W. Groninger and James J. Zaczek[1]

Abstract—Low inherent productivity, frequent flooding and changing landowner objectives are rendering much acreage in southern Illinois marginal for row crop production. Owners of these lands are attracted to reforestation by tree planting incentive programs such as the Conservation Reserve Program (CRP) which covers costs associated with site preparation, planting and stand establishment. Continued participation of these lands in set aside programs is often limited by poor survival and slow growth, forcing landowners to return these lands to agricultural production. Further, well-established and productive stands will more likely encourage landowners to retain the stand for the duration of a commercial rotation upon expiration of CRP leases.

Some factors that may be hindering the success of reforestation in southern Illinois include aggressive weed communities, poor matching of species and site, high populations of deer and rodents, and sub-optimal planting stock. A further challenge to foresters in southern Illinois is the transitional nature of its forests making uncertain the applicability of research results from elsewhere in the Central Hardwoods and Southern Bottomland Hardwoods regions. To reduce these uncertainties, we are initiating a series of studies to enhance the survival and productivity of plantings with the goal of producing fully-stocked stands in a minimum amount of time following establishment. All treatments are designed to fall within the budgetary constraints in place on reforestation incentive programs. The following research strategies will be implemented during the coming years as funding and other resources become available.

DEVELOPING LOW COST PRODUCTIVITY PREDICTIONS AND SPECIES SELECTION GUIDELINES FOR FORMER AGRICULTURAL LANDS

Many tree species are sensitive to subtle changes in topography and soil properties in bottomland settings. Failure to recognize these differences may lead to slow growth or regeneration failures. Further, site changes brought about by long-term row crop cultivation may limit the utility of site-species recommendations based on pre-agricultural conditions. Using readily available soil information, existing plantings and the experience of foresters in the region, we are developing recommendations to minimize species-site incompatibilities.

ASSESSING THE NEED FOR HERBICIDAL WEED CONTROL IN FALLOW BOTTOMLAND SITES FORMERLY IN SOYBEAN PRODUCTION

Observations by southern Illinois foresters indicate a divergence of opinions regarding the need for competition control on recently abandoned soybean lands. Some maintain that competing vegetation slows the growth of seedlings while others believe that vegetation control is unnecessary and increases seedling vulnerability to deer damage. We plan to establish vegetation control treatments throughout the region to determine the magnitude of a growth response due to vegetation management. Effects of the timing and duration of vegetation control treatments will also be evaluated. Further analyses will be conducted to determine the cost-effectiveness of these treatments.

OPTIMIZING PLANTING STOCK SELECTION FOR SITE CONDITIONS

Critical for maximizing hardwood planting success is utilizing nursery stock that is properly cultured and conditioned to begin rapid growth soon after planting. Planting stock must be able to not only tolerate but thrive when faced with competition for limited resources such as water, nutrients, or light. Stock must also be resistant to damage from herbivores. Our intent is to investigate the economic and ecological feasibility of using highly cultured and conditioned non-traditional bareroot planting stock for use in the rapid reforestation of marginal agricultural lands. We believe that the extra costs associated with utilizing higher quality planting stock will be offset by savings in post-planting care and maintenance.

STRATEGIES FOR IMPROVING ESTABLISHMENT SUCCESS ON TALL FESCUE-DOMINATED FIELDS

While the antagonistic relationship of tall fescue to hardwood tree species is well recognized, practical control measures are not fully developed. Herbicides newly labeled for forestry applications show potential to aid hardwood establishment but prescriptions to ensure establishment are still lacking. An alternative strategy involves the use of loblolly pine (*Pinus taeda* L.) as a nurse crop to suppress tall fescue and accelerate the growth of interplanted hardwoods. This strategy is especially attractive because profits from the removal of pine may be realized during the lifetime of the landowner.

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MODELING LANDSCAPE CHANGE IN THE MISSOURI OZARKS IN RESPONSE TO ALTERNATIVE MANAGEMENT PRACTICES

Stephen R. Shifley, Frank R. Thompson III, William D. Dijak and David R. Larsen[1]

Abstract—Management of Central Hardwood forest ecosystems requires an understanding of how forest landscapes will change under alternative management practices. We calibrated a landscape simulation model, LANDIS, for the Missouri Ozarks and used it to predict changes in forest size structure and species composition that will result from even-aged harvesting, uneven-aged harvesting, and no-harvest management. We simulated forest vegetation response to harvest, fire, and wind disturbance for mapped landscapes ranging from 800 to 25,000 ha in extent in the heavily forested Missouri Ozarks. The most extensive simulations were for a 842-ha mature forest landscape that was inventoried as part of the Missouri Ozark Forest Ecosystem Project.

We simulated three disturbance regimes that differed in the type and intensity of harvest. The first regime simulated even-aged management by clearcutting. Ten percent of the area was harvested each decade with oldest stands harvested first. The second disturbance regime simulated uneven-aged management by group selection. Group openings were created on 5 percent of the area of each stand in each decade. Opening size ranged from 0.1 to 0.3 ha. The third disturbance regime had no harvesting. All three regimes included simulated fire disturbance with a 300-year mean return interval (similar to fire disturbance under current levels of active wildfire suppression) and simulated wind disturbance with an 800-year mean return interval.

Required input maps for the simulation included the initial species and age class of the forest vegetation (derived from an current inventory information), ecological land types, and stand boundaries. Output maps by decade included forest age structure, species composition, type and location of harvest, intensity and location of fire, and intensity and location of wind disturbance. These maps graphically illustrate anticipated changes in forest age structure and species composition through time across the landscape. This information can be used to derive additional maps of forest type and size class (seedling, sapling, pole, and sawlog). Maps of simulated landscape change under alternative management scenarios provide opportunities to view and discuss the spatial implications of management decisions. The digital landscape maps can be further analyzed with a geographical information system to summarize landscape features such as change in forest size distribution through time, patch size, amount and type of forest edge, or other features associated with wildlife habitat quality (figures 1 and 2).

More complicated harvest patterns can be simulated by subdividing any landscape into management areas that each receive a different harvest regime. Although this simulation system will not predict the exact location of future harvest, wind, and fire disturbance events, it predicts expected large-scale vegetation patterns that result from alternative management and disturbance regimes.

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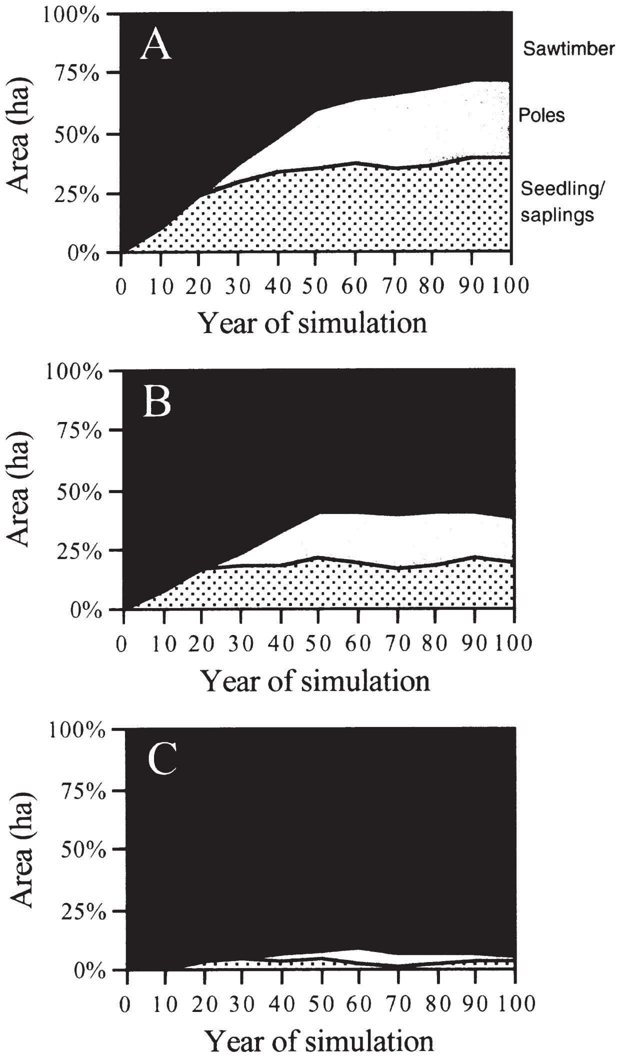

Figure 1—Area by size class over time on an 842-ha upland oak-hickory forest in the Missouri Ozarks under three simulated disturbance regimes: (A) even-aged management; (B) uneven-aged management; (C) no harvest.

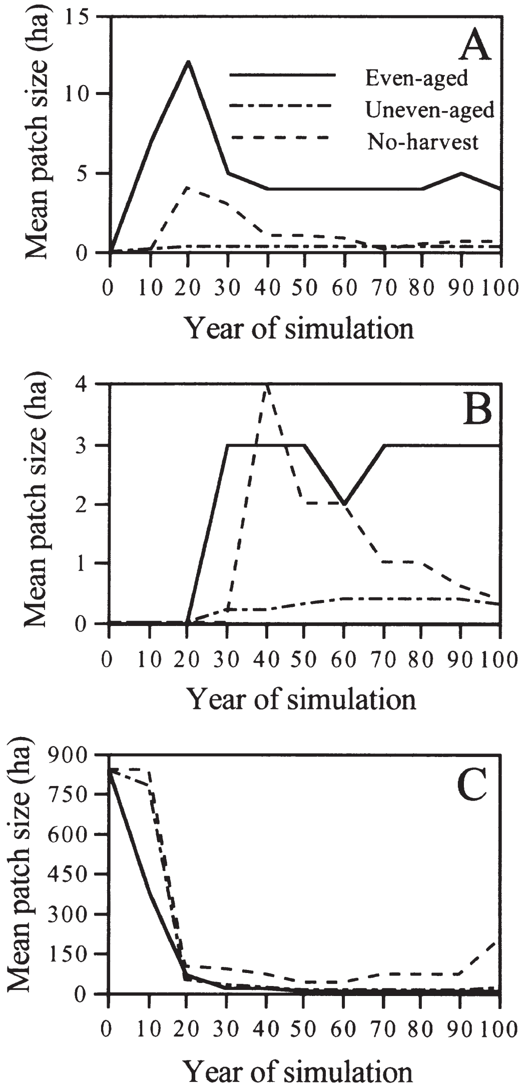

Figure 2—Mean patch size over time for three simulated disturbance regimes applied to an 842-ha upland oak-hickory forest in the Missouri Ozarks: (A) seedling/saplings; (B); pole timber; (C) sawtimber.

A FORESTLAND ALLOCATION MODEL FOR URBANIZING LANDSCAPES

Andrew D. Carver[1]

Abstract—With rapid increases in rural population and continuing expectations of economic growth, pressures on land resources within the central hardwoods region have increasingly become a topic of public debate. Controversy over the allocation of rural and urban fringe forestland often results from the competition between forest management and low-density residential development. Land allocated to forest management provides a flow of both market and non-market benefits to society. These same forests, on the other hand, are sought by developers for profitable building sites.

Though forests provide many economic and environmental benefits to communities, local land use plans and zoning ordinances rarely consider forest management as the highest and best use of rural land. This study employs a multi-criteria/multi-objective decision making model to allocate land within a 32,000 acre study area in north central Indiana to competing uses. Results form an appraisal of each 30x30 meter cell in a raster GIS database of the study area in terms of suitability for forestry and residential use. Results of the land allocation model also identify lands with increased potential for urban/forest conflict.

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MANAGING FORESTS FOR GYPSY MOTH (*LYMANTRIA DISPAR* L.) USING SILVICULTURE: TESTING THE EFFECTIVENESS OF SILVICULTURAL TREATMENTS IN REDUCING DEFOLIATION AND MORTALITY

Kurt W. Gottschalk, Rose-Marie Muzika and Mark J. Twery[1]

Abstract—Invasion of eastern forests by the exotic insect, gypsy moth (*Lymantria dispar* L.), has resulted in widespread defoliation and subsequent tree mortality. Disturbance from these factors varies widely across the landscape; some stands have little or no mortality while other stands have almost complete mortality. With average mortality rates of 25 to 35 percent, silvicultural treatments have been proposed as an alternative to insect suppression treatments to minimize gypsy moth effects. Study objectives were: 1) to evaluate the effectiveness of two silvicultural treatments (presalvage and sanitation thinnings) in minimizing gypsy moth effects on forests; and 2) to determine the mechanisms involved in silviculture-gypsy moth interactions. Only the first objective will be addressed in this presentation.

METHODS

Sanitation thinnings have as their primary objective to reduce the susceptibility of the stand to gypsy moth defoliation. The thinning treatment achieves this objective through manipulation of the species composition; reducing the preferred host composition of mixed stands to 20 percent or less of the basal area. Presalvage thinnings have as their primary objective to reduce the vulnerability of the stand to gypsy moth-related mortality. The thinning treatment achieves this objective by removing trees with higher probabilities of mortality if defoliated (low crown vigor trees) and retaining trees with lower probabilities of mortality if defoliated (high crown vigor trees).

Four replicates of each thinning and adjacent unthinned treatment stands were installed prior to gypsy moth defoliation. Each stand was 20 to 30 acres in size and contained 20 0.1-acre permanent plots arranged in a grid. All trees larger than 2.5 inches were numbered and marked at dbh. The thinning treatments were completed in April 1990. Gypsy moth defoliation occurred in May and June of 1990 and 1991 in six of 16 stands. Three years after defoliation ended, mortality was evaluated using basal area.

RESULTS AND CONCLUSIONS

Host preference class had a significant effect on defoliation patterns but thinning did not. Susceptible species (oaks, *Quercus* spp.) had higher defoliation levels than resistant and immune species but thinned and unthinned stands of the same oak composition had similar defoliation levels.

Mortality was strongly influenced by defoliation patterns and by thinning. Stands with little or no defoliation had mortality levels similar to pretreatment conditions. Stands that were defoliated had increased mortality. Thinning and defoliation had a significant interaction: in undefoliated stands, thinning had no effect on mortality, but in defoliated stands, it reduced mortality. Defoliated sanitation thinnings did not have a significant effect on either defoliation or mortality, but thinned stands did have lower mortality rates. Defoliated presalvage thinnings had significantly lower mortality rates than unthinned stands.

This study was a worst case scenario for evaluating treatments due to the short time lapse between completion of the thinning treatments and defoliation. Had there been several years for the residual trees to adjust to the thinning treatment and increase in vigor, we might have seen even larger differences. Despite this worst case situation, the significant results for presalvage thinning support the use of silvicultural treatments prior to gypsy moth defoliation to minimize gypsy moth effects on tree mortality.

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A STAND DENSITY MANAGEMENT DIAGRAM FOR NORWAY SPRUCE PLANTATIONS IN CENTRAL NEW YORK

Lianjun Zhang, Fasheng Li, Ralph D. Nyland and James P. Halligan[1]

Abstract—Norway spruce (*Picea abies* (L.) Karst.) is an economically important species in the State of New York. It has been widely planted, especially in the 1930's, and currently occupies about 100,000 acres of forest land in New York (46 percent sawtimber, 36 percent poletimber, and 18 percent sapling). Many of these plantations have already received thinning treatments over years. However, there is little information or few management tools available to guide forest managers in deciding when to start thinning and in selecting appropriate thinning intensities. The objective of this study was to construct a stand density management diagram for the Norway spruce stands in central New York.

A total of sixty-five plots (size 0.01 - 0.25 acre) were sampled from the Norway spruce plantations in six counties in central New York. Tree diameters at breast height were recorded and summarized to stand attributes such as the number of trees per acre (TPA), stand total basal area (BA) in square feet per acre and quadratic mean tree diameter (QMD) in inches. Only fully-stocked or most crowded stands undergoing self-thinning were selected and utilized to formulate the self-thinning line. Major axis regression was applied to estimate the slope and intercept coefficients of the self-thinning equation, resulting in:

$$\mathrm{Ln(QMD)} = 6.18 - 0.63 * \mathrm{Ln(TPA)}$$

A stand density management diagram was then constructed. To illustrate the applications of the diagram, a forest growth and yield model, NE-TWIGS, was used to simulate long-term stand developments of Norway spruce for a range of initial stand densities and different thinning regimes. Figure 1 shows the time trajectories of three hypothesized Norway spruce plantation stands: (1) a stand with a density of 2722 TPA (4x4 ft spacing) at age 10 and no thinning; (2) a stand with a density of 1210 TPA (6x6 ft spacing) at age 10 and three commercial thinnings to 700 TPA at age 40, to 400 TPA at age 70 and to 200 TPA at age 100; and (3) a stand with a density of 1210 TPA at age 10 and one pre-commercial thinning to 200 TPA at age 10.

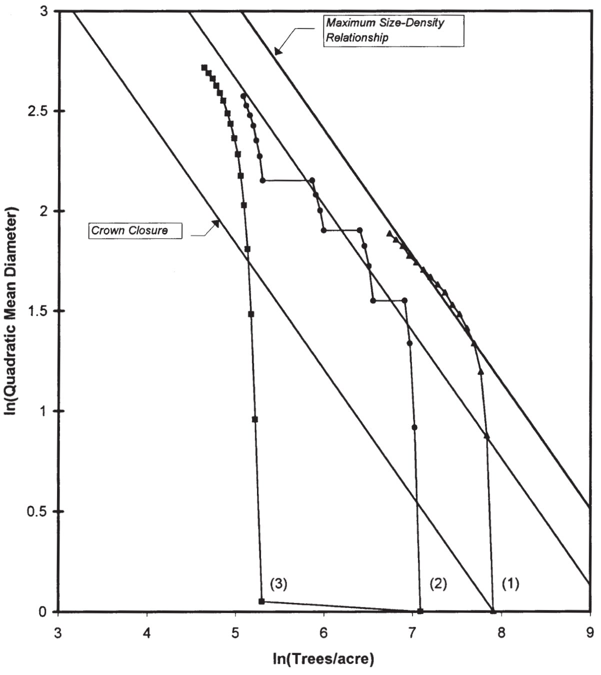

Figure 1—A stand density management diagram for the Norway spruce plantations in central New York. Simulations using NE-TWIGS for three hypothesized stands (from the right to the left): (1) a stand with a density of 2722 trees/acre (4x4 ft spacing) at age 10 and no thinning; (2) a stand with a density of 1210 trees/acre (6x6 ft spacing) at age 10 and three commercial thinnings; and (3) a stand with a density of 1210 trees/acre at age 10 and one pre-commercial thinning.

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FORCING ENVIRONMENT AFFECTS EPICORMIC SPROUT PRODUCTION FROM BRANCH SEGMENTS FOR VEGETATIVE PROPAGATION OF ADULT HARDWOODS

J.W. Van Sambeek and John E. Preece[1]

Abstract—Successful rooting of cuttings of adult hardwoods often requires that propagules be removed from the more juvenile parts of trees. Latent or dormant axillary buds found in the bark of a tree usually possess some juvenile characteristics because these buds developed when the stem or branches were first formed. In this study we evaluated the effect of different forcing environments on production of epicormic sprouts from latent buds on branch segments taken from adult trees of four hard-to-root hardwoods. In addition, we evaluated whether these sprouts were suitable as softwood or semi-woody cuttings for vegetative propagation.

In the spring of 1997 and 1998, one to four lower branches were removed from each of three phenotypically superior trees of black walnut (*Juglans nigra* L), white ash (*Fraxinus americana* L.), white oak (*Quercus alba* L), and northern red oak (*Q. rubra* L.). Branches were cut into 24 cm long segments ranging from 2.0 to 8.0 cm in diameter. Branch segments were placed horizontally in plastic 1040 trays filled with moist perlite and set in one of seven greenhouse forcing environments. Forcing environments include 1) water daily with 5 cm of water and allow to drain, 2) water daily with 5 cm of water and keep flooded 1 cm deep, 3) mist daily with 10 cm of water in 45 minutes and allow to drain, 4) mist daily with 10 cm of water in 45 minutes and keep flooded 1 cm deep, 5) place inside a humidity tent and water every other day with 5 cm of water, 6) cover trays with humidity domes and water every other day with 5 cm of water, and 7) place on shaded mist bench and mist for 6 seconds every 8 minutes during daylight hours. Due to limited greenhouse space, the forcing experiment had to be replicated over time.

Large differences were found in the number of epicormic sprouts produced per segment among trees within each species even when branch segments were taken from trees of the same age. Overall, white ash, black walnut, white oak, and northern red oak produced 5, 7, 12, and 15 sprouts per m of branch segment, respectively. The most frequently discussed forcing environment in the scientific literature, the water daily treatment, was one of the better treatments for forcing epicormic sprouts on all four hardwood species. Previous studies showed that if the epicormic sprouts were kept dry while watering the perlite, these sprouts could be surface disinfested and used as explants for in vitro culture. Branch segments under the intermittent mist treatment started producing epicormic sprouts later and produced more sprouts over a longer period of time than branch segments within any of the other six forcing environments. Shoots from the intermittent mist treatment made excellent leafy softwood and semi-woody cuttings; however, they may be unsuitable for use as explants for in vitro culture.

Branch segments in the humidity dome treatment also produced more sprouts than the branch segments in the water daily treatment. Because epicormic sprouts of white and northern red oak showed episodic growth, the sprouts inside the humidity dome had to be harvested as softwood cuttings during rapid stem elongation with immature leaves. Branch segments in the humidity tent treatment produced only half as many epicormic sprouts as branch segments in the humidity dome treatment. Presumably, the condensation inside the humidity domes reduced light penetration and kept air temperatures lower than in the humidity tent. Branch segments in the water daily with flooding and mist daily with flooding treatments produced the lowest number of epicormic sprouts. The perlite layer in these two flooded treatments retained high levels of bacteria which may have depleted the amount of oxygen, nitrogen, and carbohydrates available for epicormic sprout growth. An exploratory study with walnut and white oak showed that segments cut from the basal portion of the lower branches produced as many sprouts as segments from along the central stem.

Softwood cuttings from epicormic sprouts 4.0 cm or longer of black walnut and white oak treated with 0.1 to 4.5 percent IBA in talc and placed under intermittent mist failed to root. Subsequently softwood cuttings of all four species were dipped for 10 to 60 minutes in various dilutions of Dip'n Grow (1 percent indole-3-butyric acid and 0.5 percent napthaleneacetic acid). Over 80 percent of semi-woody epicormic sprouts from white ash dipped in a 1:24 or 1:99 dilution of Dip'n Grow rooted and could be transplanted to rootrainers for subsequent field planting. None of the softwood cuttings of black walnut or white oak rooted. Of the few northern red oak cuttings that rooted, all rooting occurred on semi-woody sprouts with full leaf expansion that had not been killed by fungi growing in the vermiculite-perlite rooting medium.

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In conclusion, epicormic sprouts can be successfully forced on branch segments cut from adult trees either by periodically watering the perlite medium or by using intermittent mist to maintain a high humidity micro-environment. Epicormic sprouts tended to root best if taken as semi-woody leafy cuttings.

LONG-TERM CHANGES IN TREE COMPOSITION IN A MESIC OLD-GROWTH UPLAND FOREST IN SOUTHERN ILLINOIS

James J. Zaczek, John W. Groninger and J.W. Van Sambeek[1]

Abstract—The Kaskaskia Woods (Lat. 37.5 N, Long. 88.3 W), an old-growth hardwood forest in southern Illinois, has one of the oldest and best documented set of permanent plots with individual tree measurements in the Central Hardwood Region. In 1935, eight 0.101-ha plots were installed in a 7.4 ha upland area consisting of xeric oak-hickory and mesic mixed hardwoods communities. The soils are cherty silt loams of the Alford and Baxter soil series in which productivity depends largely on moisture availability. The Kaskaskia Woods has never been cleared; however, increment coring in 1965 revealed a majority of the trees were either more than 160 years old or 80 to 100 years old. The area was apparently heavily cut for railroad ties in the 1880s which left most of the yellow-poplar (*Liriodendron tulipifera* L.), hickory, and oaks less than 30 cm in DBH. An abrupt change in diameter growth rates suggests a partial cut took place in the 1910's when it was likely that white oak and hickory were cut for stave wood and handle stock. The area has not been subjected to fire, grazing, cutting, or silvicultural treatments since 1933 following purchase by the USDA Forest Service.

In 1935, all trees 4 cm DBH (1.3 m above ground) or larger were tagged and identified as to species, DBH, and total height. Subsequently, tagged trees in the plots have been remeasured for survival, and DBH in 1940, 1958, 1965, 1973, 1978, 1983, and 1997 as well as height in 1958 and 1978. Ingrowth, new trees 4 cm or larger DBH, were also tagged. Six camera points were established in 1935 and rephotographed in 1958 and 1998. Individual tree locations were mapped in 1973. In the late 1990's the plots were remonumented and all live trees retagged. We report on changes in density (trees per ha) and basal area (m^2 ha^{-1}) by species composition from 1935 to the present. Importance values (IV-200), were computed by summing the percentage number of trees and percentage basal area for each species or species group.

Over the last 65 years there has been a relatively consistent and gradual increase in basal area from an initial 22.7 m^2 ha^{-1} to 34.3 m^2 ha^{-1} in 1997. During this period there have been relatively dramatic changes in species composition, and to a lesser degree, changes in density. By 1965, 479 of the 892 trees present in 1935 had died with ingrowth adding 448 new trees per ha. By 1997, 675 trees per ha had died leaving less than 25 percent of the original trees still alive. Tree density gradually declined to 788 trees per ha over the first 43 years and dropped more rapidly to 588 trees per ha during the last 20 years. The remaining trees tend to be large overstory trees with few saplings and poles in the understory.

The once dominant oaks (black, *Quercus velutina* Lam.; northern red, *Q. rubra* L.; and scarlet, *Q. coccinea* Muenchh.) and hickories (shagbark, *Carya ovata* (Mill.) K. Koch; mockernut, *C. tomentosa* (Poir.) Nutt.; pignut, *C. glabra* (Mill.) Sweet) have declined from 337 trees per ha in 1935 to less than 82 trees per ha at the present. Percentage basal area (of the total stand) for these species has dropped only slightly from 56.2 to 50.2 during the same period. The importance value (IV 200) for oak and hickory was 94.0 initially and showed a steady decline to 55.5 by 1997. The percentage of yellow-poplar basal area increased from 16.1 to 23.9 concurrently with a reduction in the number of trees (from 23.5 to 19.8 trees per ha) also reflecting the presence of massive but declining trees in the overstory.

Initially, the understory was dominated by shade-tolerant flowering dogwood (*Cornus florida* L.) and shade-intolerant sassafras (*Sassafras albidum* {Nutt.} Nees.) (247 trees per ha). By 1997, sassafras was extirpated and only 7 dogwood trees per ha remain. Black walnut, eastern redcedar, black cherry, red mulberry, and persimmon (*Juglans nigra* L., *Juniperus virginiana* L., *Prunus serotina* Ehrh., *Morus rubra* L., *Diospyros virginiana* L., respectively), present in 1935, were no longer found in the plots. In contrast, sugar maple (*Acer saccharum* Marsh.) has increased from 156 trees per ha and 1.1 m^2 ha^{-1} of basal area in 1935 to 346 trees per ha and 6.9 m^2 ha^{-1} in 1997. Over the same time period the IV 200 for sugar maple increased from 21.9 to 78.9 far exceeding any other species.

Over the course of the study, ingrowth has been primarily from shade tolerant species. Of the 637 ingrowth stems, 58.2 percent were sugar maple and 20.7 percent white ash (*Fraxinus americana* L.). Most (68.0 percent) of the sugar maple ingrowth remains alive whereas only 29.0 percent of white ash ingrowth survives. There are no surviving ingrowth trees of oak or hickory.

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Without a major disturbance for nearly 100 years, sugar maple has become the dominant species in the Kaskaskia Woods. It is likely that the coming decades will bring a continuing decline or perhaps total loss of oak and hickory as well as other associated species in this southern Illinois forest.

ASSESSMENT OF RESIDUAL STAND DAMAGE AND TREE DECAY IN PARTIAL HARVESTS

Matthew D. Seese and Mary Ann Fajvan[1]

Abstract—Partial harvesting subjects residual trees to potential stem and crown damage from felling and machinery. Stem wounds increase the potential for decay and future value loss. This study will examine the correlation between different types of logging wounds and their effects upon tree decay. Both conventional and cable logging systems will be sampled on National Forest partial harvests. Harvest types will be stratified by season of harvest, time since harvest and pre- and post-harvest stocking. Data such as tree species, dbh, merchantable height, dimensions/type of logging wounds, and individual wood samples will be gathered using 0.04 ha circular plots. All residual plot trees > 11.4 cm dbh will be assessed for damage. The dimensions of all logging wounds in the root collar and butt log (4.8m) will be measured. Each wound will be classified into three categories of increasing severity: scuff, scrape, and gouge, caused by either felling or machinery. Individual wood samples from damaged and healthy trees will be collected and specific gravity determined by the water immersion method. Volume loss due to decay will then be estimated. Relationships will be examined between 1) damage severity and harvest intensity and 2) between size and type of logging wound and potential volume loss due to decay.

Another objective of this study is to test the accuracy of three electronic decay detection devices on damaged trees. The Shigometer, Pilodyn, and an ultrasound device will be used on each logging wound to monitor the presence of decay and/or discoloration. Undamaged trees of similar size and species, will be paired with each damaged tree as a control. For the Shigometer, five 3/32 inch diameter holes will be drilled for testing. One hole will be drilled in the center of the wound, and four other holes will be drilled at right angles, 7.6 cm away from the wound's edge. The Pilodyn will be tested 2.5 cm above or below each drilled hole to evaluate its effectiveness. The ultrasound device will be tested as close to the center of the wound as possible. For the control trees, one hole will be drilled at the same height as the center hole on the damaged tree. Each device will then be tested in a similar manner as on the damaged tree. Increment cores will also be taken from each test tree to verify the presence of decay, discoloration, or sound wood.

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THE EFFECTS OF THINNING INTENSITY ON SNAG AND CAVITY TREE ABUNDANCE IN AN APPALACHIAN HARDWOOD STAND

Aaron Graves and Mary Ann Fajvan[1]

Abstract—Historically, silvicultural practices have focused on manipulating forest vegetation structure for commodity production. Impacts of silvicultural treatments on characteristics important to nongame species, such as feeding substrate, potential nesting sites, or the rate at which these habitat features form are often not considered. In this study, features such as snags, cavity trees, and defective/decayed trees, which are important to five local woodpecker species, will be compared among even-aged stands. In 1983, a 50-year-old Appalachian mixed hardwood stand was divided into twenty, 1.2 ha blocks, each of which contained a central 0.2 ha plot. All live trees ≥2.54 cm dbh in the central 0.2 ha plot were permanently tagged. Five plots each were thinned to either 75 percent, 60 percent, or 45 percent relative density, and five plots were uncut controls. Plots have been remeasured for fifteen years at five year intervals and data from these remeasurements will be used to quantify tree mortality and snag dynamics over time. Trees were classified as snags if they were ≥10 cm dbh and 1.5 m tall, and were self supporting. A total of 319 snags were sampled in plots encompassing 4 ha.

Mean snag densities show an inverse relationship to thinning intensity (Figure 1), with individual plots ranging from 9.9 snags/ha (45 percent) to 192.7 snags/ha (Control). Large diameter snags (≥30 cm) were uncommon in all treatments. Sassafras (*Sassafras albidum*) and black cherry (*Prunus serotina*) were the most common snag species, comprising 59 percent of snag density.

Similar to other studies, the percentage of snags standing decreased dramatically as time since death increased. After 0-5 years, 85 percent of mortality trees remained standing, which decreased to 59 percent at 5-10 yrs and 25 percent at 10-15 yrs. The longevity of snags in this study is greater than reported for other eastern hardwood species, which is likely due to the large component of decay resistant sassafras and black cherry.

Future efforts will focus on quantifying defects and decay in live overstory trees, and quantifying characteristics of trees containing bird excavated cavities.

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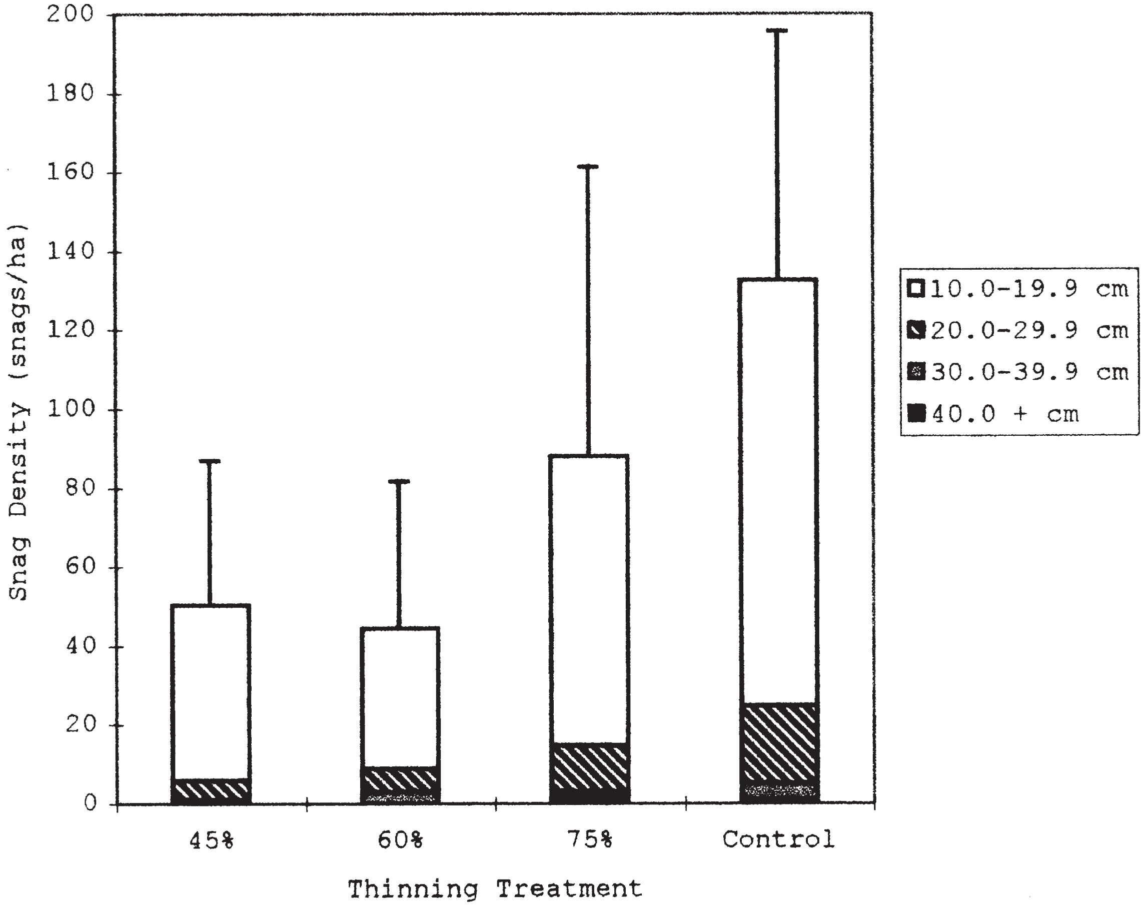

Figure 1—Snag density (snags/ha) by thinning treatment and size class. Bars = 1 SD.

A COMPARISON OF FVS/SUPPOSE COMPUTED VOLUME WITH USDA FOREST SERVICE CRUISE VOLUME ON THE MONONGAHELA NATIONAL FOREST

Melissa Thomas-Van Gundy[1]

Abstract—Summarized plot data from Forest Service stand inventory for four separate stands was entered in FVS/Suppose. The tree data, collected in 1984 and 1989, was grown to the common year 1997 in FVS using NE-Twigs. Either a clearcut or two-age harvest was applied to each stand depending on the prescription given in the environmental assessment. The resulting removal volume in board feet by species was compared to volumes calculated by the National Cruise program from a standard timber sale cruise conducted by district personnel.

Tables 1a and 1b show the total volume by species for each stand and the absolute differences between the two. Differences in removal volumes by species ranged from 1 to 76 MBF. Comparisons of volumes by species within each stand were not made because the range of differences was so great and variable. Only raw data is reported here.

The FVS/Suppose program used here has not been calibrated with either local data or Forest Inventory and Analysis plot data. This is needed before volumes created from the growth model in FVS/Suppose can be applied to individual stands. Schuler and others (1993) found NE-Twigs to be suitable for predicting basal area per acre, medial stand diameter, number of trees, and percentage of basal area in the primary species group for transition hardwood and oak-hickory forest types for West Virginia. However, the authors caution that these reasonable estimates were based on an average of numerous stands.

REFERENCES

Schuler, T.M.; Marquis, D.A.; Ernst, R.L.; Simpson, B.T. 1993. Test of four stand growth simulators for the Northeastern United States. Res. Pap. NE-676. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station.

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Table 1a—Total volumes from cruise data and FVS

Payment unit (*acres*)	Comp Stand	Chestnut oak MBF cruise MBF FVS	Hickory MBF cruise MBF FVS	Pine MBF cruise MBF FVS	Red Maple MBF cruise MBF FVS
2	58	20	0	2	2
(14)	5	19	0	18	0
Diff.		1	0	16	2
3	58	23	2	4	1
(17)	30	94	6	0	0
Diff.		71	4	4	1
4	58	5	1	0	1
(9)	30	50	3	0	0
Diff.		45	2	0	1
5	58	18	3	2	5
(17)	30	94	6	0	0
Diff.		76	3	2	5
6	58	13	2	8	1
(18)	46	33	3	35	0
Diff.		20	1	27	1
8	60	15	1	0	3
(12)	21	56	21	0	1
Diff.		41	20	0	2

Table 1b—Total volumes from cruise data and FVS

Payment unit (*acres*)	Comp Stand	N. red oak MBF cruise MBF FVS	White oak MBF cruise MBF FVS	Yellow-poplar MBF cruise MBF FVS	Other hdwds. MBF cruise MBF FVS
2	58	29	2	1	0
(14)	5	3	0	0	20
Diff.		26	2	1	20
3	58	17	2	0	0
(17)	30	56	7	0	17
Diff.		39	5	0	17
4	58	26	9	0	0
(9)	30	30	4	0	9
Diff.		4	5	0	9
5	58	60	0	3	0
(17)	30	56	7	0	17
Diff.		4	7	3	17
6	58	20	10	1	0
(18)	46	21	13	0	19
Diff.		1	3	1	19
8	60	54	3	1	0
(12)	21	11	15	2	2
Diff.		43	12	1	2

THE EFFECT OF USING CONTROL BAGS ON LITTERBAG MEASUREMENTS OF LEAF LITTER DECOMPOSITION AND NUTRIENT DYNAMICS

K.A. Holzbaur, P.E. Pope, T.W. Idol and F. Ponder, Jr.[1]

Abstract—In most litterbag decomposition studies, mass loss is calculated as simply the change in litter mass between successive time periods. However, studies have found that the influx of soil, foreign litter, and microbial biomass can add weight to litter in litterbags. This can lead to underestimates of true litter mass loss and incorrect calculations of nutrient mineralization and immobilization patterns. The purpose of this study was to assess the effect of using control bags on the mass loss and nutrient dynamics calculated from a traditional litterbag study of decomposing leaf litter. Litter and fermentation layers from four oak-hickory stands ranging in age from 5 to approximately 100 years since harvest were collected, dried, and placed in nylon mesh litterbags. Control bags consisted of inert, undecomposable material approximately the shape and size of the litter. All bags were placed in the forest floor of the 90, or 120 days. Mass loss and nutrient content of the litter was determined and calculations of the decomposition and nutrient mineralization and immobilization patterns were assessed both with and without correcting for the control bags. Results suggest that control bag corrections had a dramatic effect on calculations of mass loss and nutrient dynamics. Thus, short-term decomposition and nutrient dynamics cannot adequately be assessed in these traditional studies without the use of control bags.

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UNDERSTORY FIRE EFFECTS ON PIN CHERRY (*PRUNUS PENSYLVANICA* L. F.) SEED GERMINATION

David W. McGill, Edmund T. Bridge, and Jason B. Hudson[1]

Abstract—Pin cherry (*Prunus pensylvanica* L. f.) is an undesirable species in silvicultural systems that aim to produce high quality sawtimber and pulpwood. During stand initiation, pin cherry can rapidly capture canopy growing space and relegate other species to subcanopy positions as it is among the most rapid growers in height. It is estimated that pin cherry seed can remain viable for 50-150 years (Wendel 1990) and its germination is stimulated by regeneration cutting or other intense canopy disturbance. The intense competitive positioning of pin cherry during stand initiation can result in reduced growth or mortality of more desirable species. Although relatively short-lived, pin cherry can comprise a significant amount of stocking up to age 20. This prolonged dominance decreases efficiency and flexibility of early precommerical thinnings: thinning crews will likely focus on eliminating pin cherry rather than selecting crop trees from among competing desirable species. The intent of this poster is to document the effects of an understory fire on dormant pin cherry seed germination and to discuss implications for potential control of this species.

On April 3, 1997, a 3.8-acre wildfire was started by a downed powerline near Anjean, Greenbrier Co., WV. Merchantable stand basal area was 116 sq. ft. per acre in the 46-year-old northern hardwood stand, comprised of yellow-poplar, black cherry, red oak, sugar maple, and birch. The fire moved up the 30- to 40-percent south-facing slope burning only the uppermost portion of the leaf litter and charring the lower trunk bark of overstory trees.

On April 11, 1997, ten 0.001-acre regeneration plots were established to assess fire effects on germination of the pin cherry seedbank. Numbers of seedlings (less than 4.5 feet) were recorded. Five plots were located in the burned area and five plots in the adjacent unburned area, all were beneath pin cherry stems that ranged from 6 to 12 inches dbh. In the burned area, three of the pin cherry stems were living and two were dead; in the unburned area two were living and three were dead. In July 1997, seedlings were again counted on all plots.

Three months following the fire, pin cherry was the most abundant species on the burned regeneration plots, averaging 26,400 seedlings per acre. Root suckers of living pin cherry trees were observed but not counted in the regeneration assessment. No pin cherry seedlings were observed on the unburned plots. Species richness was higher in the unburned area (14 species) than in the burned area (7 species).

The germination of pin cherry seed observed following this understory burn was surprising because of the intact and complete canopy cover present at the time of and following this relatively low intensity burn. Most references describing the germination of pin cherry seed suggest that some degree of canopy disturbance is required to stimulate the dormant seed bank (e.g., Marks 1974); germination of dormant seed has been shown to result from high fluctuations of soil temperature (Laidlaw 1987) and from increased soil nitrate levels (Auchmoody 1979), both characteristics found in forest soils following intense canopy disturbances.

Fire also stimulates pin cherry seed germination, although no mention has been made in the literature that low intensity understory burning leads to this result. Pin cherry establishment is usually linked with some type of intense canopy disturbance. Low intensity understory burning may provide a means to reduce potential competition by pin cherry. If dormant seeds in the seedbank of untreated stands can be stimulated to germinate, then these shade intolerant seedlings will likely die out in the low light environment of the forest understory before the stands are regenerated.

REFERENCES

Auchmoody, L.R. 1979. Nitrogen fertilization stimulates germination of dormant pin cherry seed. Canadian Journal of Forest Research. 9: 514-516.

Laidlaw, T.F. 1987. Drastic temperature fluctuation—the key to efficient germination of pin cherry. Tree Planter's Notes. 38(3): 30-32.

Marks, P.L. 1974. The role of pin cherry (*Prunus pensylvanica* L. f.) in the maintenance of stability in northern hardwood ecosystems. Ecological Monographs. 44: 73-88.

Wendel, G.W. 1990. *Prunus pensylvanica* L. f.—Pin Cherry. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America: hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service. Vol. 2

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USING PRESCRIBED BURNING TO RELEASE OAK SEEDLINGS FROM SHRUB COMPETITION IN SOUTHERN CONNECTICUT

Jeffrey S. Ward and Emery Gluck[1]

Abstract—In a managed forest, control of species composition is important for both the economic value and biological health of the forest. Regenerating oaks on better quality sites is often hampered by an extensive shrub layer that suppresses the shorter oak seedlings, especially following shelterwood harvests. The abundance of mature oaks in the present forest is due, in part, to a history of periodic burning and clearcutting prior to 1920 in Connecticut. Fire is a possible method of killing (or stunting) shrubs and allowing oak sprouts to grow above the shrub layer.

A shelterwood cut in 1987 in a mature oak stand had broken up a layer of mountain laurel that dominated the understory and increased the number of oak seedlings to nearly 4000/acre by 1989. However, most oak seedlings were less than 1 foot tall and were stagnating under a dense shrub layer. A prescribed burning study was established to determine whether fire can be used to release oak seedlings from shrub competition. Thirty-six plots were located on a nominal 150 foot by 150 foot grid. Seedlings and saplings were sampled by species and height class within 1/300 acre circular plots. Trees (> 4.5 inches dbh) were sampled with a 10-factor prism. Residual basal area was 60 ft^2/ac with oak species accounting for 51 percent of the total.

Half of the 29 acre stand was burned by a surface fire on April 5, 1991, in cooperation with the Division of Forestry, Connecticut Department of Environmental Protection. The fire was implemented under mild burning conditions: dead fuel moisture 10-12 percent, winds 2-7 mph, and relative humidity 38-56 percent. The flame length of the head fire rarely exceeded 1 ft while burning in the hardwood litter. The fire extinguished itself in the wetter sections. The fire killed to the ground most stems less 1 inch in diameter. In June, it appeared that the oaks were competitive with the resprouting shrubs. Regeneration was sampled immediately before the burn and in the fall following the burn. Regeneration was again sampled three and six years after the burn. Overstory removal was completed by the fall of 1994 (4 growing seasons post-burn).

Burning has increased relative and absolute oak seedling density. Two years after final harvest, oak density was 17,475/acre on the burned section compared with 5,125/acre on the unburned sections. More significantly, oak species accounted for fully 25 percent of tree stems ≥ 3 feet tall on the burned section and only 8 percent on the unburned section (fig. 1). Prescribed burning also reduced the density of red maple, sassafras, and shrubs. Prescribed burning may be a useful for controlling vegetation that competes with oak following shelterwood cuts.

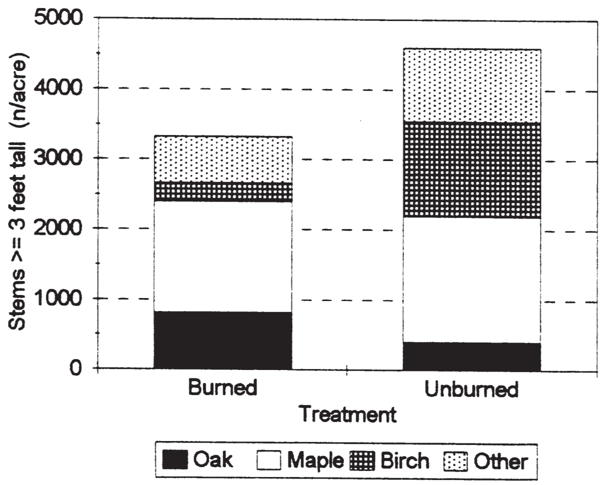

Figure 1—Distribution of stems ≥ 3 feet tall by species group 2 years after final harvest and 6 years after prescribed spring burn.

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CONSTRUCTION METHODS FOR A COUNTY-WIDE LAND USE/COVER MAP

C.J. Liu[1]

Abstract—Laboratory of Geometronics, through a cooperative agreement with the Big Sandy Area Development District, has constructed a GIS-based, county-wide digital land use/cover map to facilitate county-level economic development and resources management planning efforts. The construction procedures described can readily be used for projects of similar scope and objective.

The procedures of map construction are: (1) photointerpretation, (2) photogrammetric transformation, (3) scanning, (4) georeferencing, (5) vectorization, (6) data editing, and (7) cartographic manipulations. Detailed processing is presented below.

Sixty seven vertical aerial photos taken in March of 1997 by the National High-altitude Photography Program were acquired for photointerpration. Phototerpretation was peformed by viewing stereo pairs under a stereoscope while land cover types identified and marked. Land use polygons marked on the photos were than transferred to fourteen USGS 7.5' quadrangle topographic maps using a sketchmaster. These topographic maps were treated as the project's base maps in a GIS. In 1-bit monochrome mode, based maps with land covers identified were scanned at 100 dpi. This process converted polygonal features on base maps to their corresponding raster images. Using a set of ground control points collected on additional reference materials, raster images were georeferenced in Arc/Info. Georeferenced raster files were subsequently vectorized which converted raster images to grid files in Arc/Info. Next, boundary lines for all land use polygons were traced to produce vectors, using grid files in Arctools. These vectorized polygons were ten saved as a GIS coverage. Fourteen GIS coverages of land uses were merged, in Arcedit, to form a single coverage. The resulting coverage were attributed in Arcview by assigning land use types to polygons. The next step was to dissolve, in Arc, boundaries of polygons having the same attribute values (i.e., land-use types). After a georeferenced land-use coverage was built, several cartographic treatments were applied to the coverage in Arcview. These treatments included: (a) a 1000-meter UTM grid coverage and markers, (b) a 5-foot longitude/latitude grid coverage and markers, (c) other legends common to all paper maps.

Following the above outlined procedures, a color-coded county-wide land use maps was successfully constructed. The Magoffin County Land-Use Map is in U.S. State Plane Coordinate (Kentucky South) and North American Datum of 1983. The table below summarizes Magoffin County Land Uses.

Table 1—Magoffin County land uses

	Area	Percentage
	Acres	
Agriculture land	19,986.88	10.09
Barren land	2,277.99	1.15
Deciduous forest	163,262.51	82.42
Mixed forest	415.98	0.21
Stripped mine	9,389.28	4.74
Urban land	2,674.16	1.35
Water body (exclude streams)	99.04	0.05
Total	198,086.04	100.00

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COMPARISON OF NE-TWIGS AND ZELIG ON ACTUAL GROWTH OF TWO SITES IN KENTUCKY

Daniel A. Yaussy[1]

Abstract—Two individual-tree growth simulators are used to predict the growth and mortality on a 30-year-old forest site and an 80-year-old forest site in eastern Kentucky. An empirical growth and yield model (NE-TWIGS, Hilt and Teck 1989) was developed to simulate short-term (< 50 year) forest growth from an industrial perspective. The gap model (ZELIG, Urban 1990) is based on the theory of growth processes and was designed to simulate long-term (100 years and greater) forest succession. Based on comparisons of species specific diameter distributions, biomass, and board-foot and cubic-foot volumes, NE-TWIGS performed better for the 80-year-old site than did ZELIG. Neither simulator provided acceptable predictions for the 30-year-old site.

The ZELIG model over predicted the number of stems for the 30-year-old site, but under predicted the biomass and volumes. This implies that the model does not grow the white oaks fast enough. The red maples, on the other hand, grew too fast and the white oaks incurred too much mortality on the ZELIG simulations of the 80-year-old site.

The NE-TWIGS model cannot be easily altered to obtain better predictions for specific areas. One could annually adjust the site-index value based on seasonal temperatures and precipitation rates being above or below local averages. NE-TWIGS was designed to work in the time frame and region used in this comparison; however, it is a generalized model, not developed, specifically for an even-aged, upland oak stand.

The species parameters in the gap models can be modified, which might produce comparable growth predictions. An adjustment of site specific and species parameters was investigated for white oak on these data sets. White oak made up more than 75 percent of the stems at each site, and any improvement in prediction of the growth for this species would improve the overall predictions immensely. Soil fertility was the site parameter in which I had the least confidence. Whittaker (1975) reports net primary productivity for temperate deciduous forests ranges from 6 to 25 Mg/ha/yr averaging 12 Mg/ha/yr, much higher than was calculated for these sites. Another parameter which might be altered is the species growth constant which changes the age at which a tree of that species puts on most of its diameter growth. A larger constant implies that trees of this species grow quickly early in their life and growth tapers as the tree ages. A lower constant allows trees of the species to grow in diameter evenly as they age. The maximum age and diameter that a tree species can attain are parameters which determine many aspects of growth and mortality within the ZELIG model.

Systematic combinations of values within the ranges of the soil fertility parameter (2-24 Mg/ha/yr) and the white oak growth constant parameter (50-300) were tested within the ZELIG framework seeking a combination that would improve the simulations of both stands. Graphs of the relationships between these parameters and average dbh, number of trees per hectare, and the volume measures showed that the volume measures and average dbh were not sensitive to soil fertility values above 10 Mg/ha/yr. Number of stems on each site was quite sensitive to both parameters. Parameters were found for the 30-year-old site which satisfactorily predicted number of stems, average dbh, biomass, and cubic foot volumes; however the diameter distribution was such that the board foot estimates were still quite low (not enough large trees). These parameters simulated reasonable volume measures for the 80-year-old site, but the estimated average dbh and number of trees were not close to those of the actual site. No combination of parameters tested could produce enough trees for the eighty-year-old stand. Reasonable average dbh estimates for the 80-year-old site could be attained, but never with the proper number of trees to produce the right combination of biomass, cubic-foot volume or board-foot volume.

REFERENCES

Hilt, D.E.; Teck, R.M. 1989. NE-TWIGS: An individual-tree growth and yield projection system for the Northeastern United States. The Compiler. 7(2): 10-16.

Urban, D.L. 1990. A versatile model to simulate forest pattern: a user's guide to ZELIG. Version 1.0. Charlottesville, VA: The University of Virginia, Environmental Sciences Department. 108 p.

Whittaker, R.H. 1975. Communities and ecosystems. 2d ed. New York: MacMillan. 385 p.

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THE IMPACT OF PRESCRIBED FIRE ON HERBIVORY LEVELS OF UNDERSTORY WHITE OAK

Aaron S. Adams and Lynne K. Rieske-Kinney[1]

Abstract—Historical data and the presence of fire-resistant characteristics support the role of fire in the establishment and maintenance of the mixed-oak (*Quercus* spp.) forests of eastern North America. Following the clear cutting of eastern forests in the 1800's, fire suppression became a dominant forest management technique. Age and species mosaics declined and forest stand composition shifted, allowing more vigorous and shade tolerant species to dominate. As a result, seedlings in oak-dominated forests became suppressed by vegetative competition, and today oaks are virtually non-existent in the midstory. This shift in species composition has resulted in the economic loss of an extremely valuable hardwood group, and may also impact forest succession rate, wildlife composition and distribution, and watershed characteristics.

The use of prescribed fire as a management tool to enhance oak regeneration has recently been met with renewed interest. Fire enhances oak growth by reducing vegetative competition, increasing sunlight penetration, and increasing nutrient composition in the soil.

Traditionally research has focused on environmental factors and vegetative competition influencing oak regeneration. However, acorns, seedlings, and mature oaks are constantly exposed to insect herbivory. Prescribed fire may serve to manipulate insect composition and abundance, which could effect seedling establishment and sprout success. Fire may impact herbivore populations directly by habitat alteration and disruption of the life cycle, or it may have indirect impacts caused by alterations in food quality and availability.

OBJECTIVES

This study is identifying biotic factors attributing to the lack of oak regeneration in eastern Kentucky forests, and will assess the effects of prescribed fire on the herbivore complex associated with oak regeneration. The specific objectives of this research are to characterize the effects of single year and multiple year burns on: 1) oak seedling growth, 2) phytochemistry, and 3) herbivory levels.

MATERIALS AND METHODS

Study sites were established in the Daniel Boone National Forest (DBNF) on oak-dominated ridgetops under the following burn regimes: 1) single year burn - areas burned in late winter of 1998, 2) multiple year burn - areas burned in late winter of 1998 and 1996, following the prescribed fire management program established by the DBNF, and 3) unburned controls - areas with no recent history of fire activity.

Within each site, subplots consisting of a series of herbivore exclusion treatments were established to assess herbivory levels in each burn regime. Each subplot consists of seedling fencing/insecticide treatments designed to: 1) exclude arthropod herbivory(- fence/ + insecticide), 2) exclude mammalian herbivory(+ fence/ - insecticide), 3) exclude both mammalian and arthropod herbivory (+ fence/ + insecticide), and 4) no herbivore exclusion (- fence/ - insecticide). The split plot design involves seedling treatments (+/- fence, +/- insecticide) replicated 3 times within site treatments (1x-burn, 2x-burn, non-burned), which are also replicated three times.

To assess the effects of prescribed fire on oak growth, measurements of seedling performance (shoot elongation, height, diameter) were taken at 4 equal intervals throughout the growing season. Twenty five adjacent seedlings were flagged for periodical destructive sampling for phytochemical analysis. Foliar samples were collected to analyze for total non-structural carbohydrates, nutrients, protein, and phenolics.

Herbivory was visually assessed by measuring the area affected by mammalian and arthropod feeding on each tree at 14 d intervals throughout the season. Levels of herbivory in each of the 4 seedling (subplot) treatments were compared under the different burn regimes. Herbivory levels will be correlated with seedling growth (Objective 1) and phytochemistry (Objective 2).

To augment the relatively low (0-30 percent) natural herbivory levels, fall webworm (FWW) larvae were caged on additional seedlings. Paired seedlings were caged with and without larvae in each burn regime and allowed to feed for 6 days. After caging, leaf tissue was flash frozen for future phytochemical analysis.

RESULTS

Results of the augmented herbivory experiment are presented here. Final herbivory levels across burn treatments averaged 50 percent. Since there was no significant difference between the once and twice burned sites pre- or post-FWW challenge, results for the burn treatments were pooled. Prior to the FWW caging

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experiment, mean defoliation levels of seedlings on non-burned sites exceeded that of burned sites (Table 1). This trend was reversed for the post-challenge sample date, when defoliation on seedlings from burned sites exceeded that of non-burned sites. FWW defoliation increased by greater than 90 percent on burned sites, but only 75 percent on non-burned sites.

The overall herbivory rate of FWW challenged oak seedlings on burned sites significantly exceeded that of non-burned control sites (Fig. 1, $P = 0.0442$).

DISCUSSION

By FWW Herbivory

The lower herbivory levels in pre-challenged, burned plots may reflect enhanced production of inducible defenses

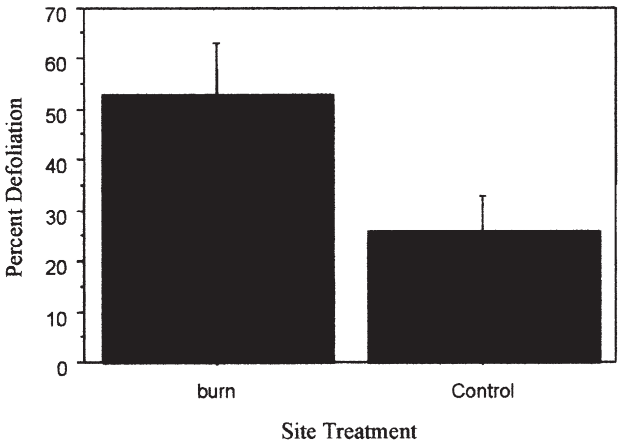

Fig. 1—Defoliation of white oak seedlings challenged.

Table 1—Mean percent defoliation (s.e.) of white oak seedlings challenged by fall webworm herbivory on burned and non-burned sites

White oak seedlings	Burned	Non-burned control
	------- *Percent defoliation* ------	
Pre-FWW challenged	6.46 (1.89)	9.66 (2.10)
Post-FWW challenged	59.29 (10.27)	35.68 (5.27)

such as phenolics due to site (burn) treatment. The increase in herbivory on burned sites for the post-FWW challenge may be attributable to an induced defensive response which has been compromised by the stresses of site treatment combined with herbivory levels. Future research will concentrate on analysis of phytochemical compounds within each treatment.

This study will elucidate some of the complex interactions between fire, oak seedling growth, phytochemistry and herbivory. Knowledge of these interactions may alleviate some of the difficulties associated with the oak regeneration issue. As natural oak regeneration and human management techniques fail to produce the desired results, prescribed burning offers some hope.

NURSERY TREATMENTS ALTER ROOT MORPHOLOGY OF 1+0 NORTHERN RED OAK SEEDLINGS

Patricia T. Tomlinson[1]

Abstract—Poor root growth is frequently associated with poor planted northern red oak seedling establishment and performance. Therefore, we determined if several nursery treatments resulted in differences in northern red oak seedling characteristics, particularly in root systems, which might increase rapidity of growth after outplanting and thus improve regeneration success. The treatments we chose were undercutting after two flushes of growth and dipping germinated seed in either low concentration (2,000 ppm) or high concentration (8,000 ppm) K-IBA prior to planting. Each of these has been reported to alter number of first-order lateral roots and/or their distribution along the taproot. However, reports of these treatments compared were not found. Half-sib seed from each of two mother trees were planted either in the fall 1992 (control and undercut treatments) or in the spring 1993 (auxin dipped seed treatments) at Wilson State Forest Tree Seedling Nursery in Boscobel Wisconsin (43N lat., 90W long.). The mother trees were located in Nicolet National Forest (SS1) and in Chequamegon National Forest (SS2) in northern Wisconsin. Seedlings in the undercut treatment were undercut at 15 cm depth when seedlings were approaching the end of the second flush of growth. Seedlings were lifted at 20 cm depth after one season of growth in April 1994. They were characterized by height; numbers of permanent first-order lateral roots on the upper and lower 10 cm of the taproot; diameters at-, 10 cm below-, and 2 cm above- root collar.

Taproot length and distribution of lateral roots along the taproot were significantly impacted by treatment. Auxin, especially high concentration, shortened taproot length and increased the proportion of lateral roots in the upper 10 cm of the taproot. However, total number of first-order lateral roots was not significantly different among cultural practices for either seed source. Root collar diameter as well as stem diameter (2 cm above root collar) was impacted by both seed source and treatment. For SS1, high concentration auxin treatment resulted in smaller diameters than control or undercut treatments. In contrast, control seedlings displayed smaller diameters compared to other treatments for SS2, although the difference was only significant for undercut versus control comparison. Taproot diameters (10 cm below the root collar) were decreased by auxin treatment and increased by undercutting. Shoot height was not significantly affected by any treatment for either seed source. We also quantified the number of growth flushes; however, this was uniform among seed sources and treatments as observed in the past over a broader range of conditions.

Thus, root morphology can be manipulated by undercutting and auxin treatments. These treatments appear to alter the root system with less impact on the shoot system of the seedling. Seedling height was not impacted although both root collar diameter and stem diameter were altered by the treatments for at least one seed source. Further, such influences on the shoot were modified by seed source whereas root diameter was dependent only on treatment. Undercutting and auxin treatments alter the distribution of first-order lateral roots along the taproot without impacting the total number of lateral roots. The impacts of these differences in root morphologies on outplanting performance and seedling establishment is being tested.

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EFFECTS OF LEAF LITTER DEPTH ON ACORN GERMINATION

Jeffrey W. Stringer and Laurie Taylor[1]

INTRODUCTION

Lack of advanced regeneration is generally considered to be the primary factor associated with the failure to regenerate oak particularly on medium to high quality sites. This is generally considered to be a function of the limited light environment often associated with undisturbed canopies on mesic sites. However, other factors such as a relatively thick litter associated with these types of sites has been shown to increase the desiccation of acorns during the initial stages of radical immersion and thus potentially reduce the number of seedlings established. This study was designed to test a litter reduction treatment (prescribed fire) on the successful germination and establishment of selected upland oak species.

METHODS

The study was located at Robinson Forest, the University of Kentucky research and demonstration forest located in the Cumberland Plateau Physiographic Province in southeastern Kentucky. Two upland oak stands, approximately 3 acres in size, on slopes of opposing aspects were used in this study. One stand was located on a predominately southwest facing slope (black oak site index=65ft) and the other stand was located on a northeast facing slope (black oak site index=90ft). Ten study plots (9 m^2) were randomly located in each stand. The study incorporated a split plot design where each plot was split into a untreated subplot and a treated (prescribed burned) subplot. Each subplot was divided into 4 species plots. One of four oak species (*Quercus rubra*, *Q. alba*, *Q. prinus*, and *Q. coccinea*) were randomly assigned to each of the 4 species plots and 20 acorns of that species were planted in October of 1997. In the untreated subplots acorns were placed on the existing litter directly prior to leaf fall. Acorns were protected from vertebrate predation and the current years leaf deposition covered the acorns. To avoid altering the environment around the acorns predation protection was removed after leaf deposition. Treated subplots were established by burning directly prior to planting in October. The burn was sufficient to remove the majority of the litter. Acorns were then dropped on the exposed soil, or in some instances charred organic matter. They were protected from predation and leaf deposition allowed to cover the acorns at which time protection was removed. A 565 cm^2 sampling frame was used to collect litter samples were collected from five areas at the edge of each subplot. Samples were dried and pre-treatment litter mass determined. Within the burned subplots litter samples were also collected after leaf fall to determine litter cover over the acorns. The litter samples were used to determine pre-treatment litter mass (kg/ha) and post-treatment average litter mass for each subplot. Plots were evaluated in June. The percentage of acorns which developed seedlings in each species block was determined and subjected to normalization using an arc sin transformation for analysis using the Wilcoxon two sample t-test to determine treatment effects for the data pooled over sites and species as well as species within a site.

RESULTS

Burning resulted in significantly higher seedling establishment percent for some species. Overall seedling establishment averaged 1.32 percent on burned plots and 1.04 percent on unburned plots. These values are similar to other reported establishment percent for years where acorn production is average or below average. Establishment rates have been shown to be as high as 10 percent for years when bumper crops occur. A more pronounced and significant difference ($p=0.0297$) was found on the northeast site compared to the southwest site. Mean seedling establishment percent pooled over all species on the northeast site was 1.38 on burned plots and 0.62 on unburned plots. Mean seedling establishment percent pooled over all species was not significantly different ($p>0.05$) between treatments on the southwest site (unburned = 1.41 percent and burned = 1.25 percent).

Northern red oak exhibited significantly higher ($p=0.0006$) establishment percent in burned plots (mean=2.0) on the northeast site compared to unburned plots (mean=0.30) (Table 1). Chestnut oak also exhibited significantly higher ($p=0.02$) establishment percent on burned plots (mean=2.0) compared with unburned plots (mean=0.50). No treatment differences were exhibited by white oak (which totally failed on the northeast site) or scarlet oak on either site.

This study indicates that prescribed burning may represent a viable treatment for increasing the development of advanced regeneration of chestnut oak and northern red oak on intermediate and high quality upland sites. While the primary effect of the burning was believed to be on litter thickness it may have also changed other factors which could have effected germination and establishment. The

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burning may have influenced population densities of predatory insects which might feed on acorns and emerging radicles or altered the potential of infection by pathogens. Regardless, this study indicates that the use of properly timed prescribed burning should be investigated as an aid in establishing oak advanced regeneration on mesic sites.

Table 1—*Quercus* seedling establishment as influenced by prescribed fire

	Q. coccinea	*Q. prinus*	*Q. alba*	*Q. rubra*
	Percent			
Southwest slope				
Burned	1.25 (0.55)[1]	1.88 (0.63)	0.63 (0.06)	1.26 (0.56)
Unburned	1.88 (0.63)	2.50 (0.65)	0.61 (0.06)	1.89 (0.90)
Northeast slope				
Burned	1.50 (0.53)	2.00[a] (0.56)	0	2.00[a] (0.76)
Unburned	1.50 (0.53)	0.50[b] (0.34)	0	0.30[b] (0.03)

[1] Values represent mean seedling establishment percent (standard errors). Values with different letters represent a significant different ($p<0.01$) between treatments using Wilcoxon two sample t-test.

DEVELOPMENT OF ADVANCED OAK REGENERATION FROM TWO-AGE RESERVE TREES

Jeffrey W. Stringer[1]

INTRODUCTION

The two age system uses methods or treatments which retain a limited number of older canopy trees (reserve trees) along with a cohort of younger regenerating stems. Methods such as “irregular shelterwood” or “shelterwood with reserves” developed from treatments such as deferment cuts can be used successfully in many hardwood stands. This method has been used as an aesthetic alternative to clearcutting and as a means of potentially developing a limited number of large diameter high value sawtimber trees in a stand. A shelterwood with reserves differs from a traditional shelterwood in that reserve trees will be maintained for a second rotation length rather than removed after regeneration goals have been met. To allow for continued rapid development of the younger age class the reserve trees must be scattered and few in number (10-30 ft^2 of basal area a^{-1}). In this case, the term shelterwood with reserves is misleading as the goal is not to shelter or modify regeneration but to allow it to develop relatively unhindered. The reserve trees must be of proper vigor, landscape position, species, age, and potential tree grade so that they will survive and provide a viable product after two rotations. Not all stands and species can be managed using the two age system. Species which are relatively long lived and are commercially important make good candidates for reserve trees.

The two age system also can be used to “life boat” species which do not have viable reproductive life forms at the time of cutting. A traditional clearcut essentially stops sexual reproduction in the stand for a substantial portion of the rotation and can limit the potential for the development of viable advanced regeneration. The reserve trees in the two age system provides for the potential for continued sexual reproduction in the stand and the ability to develop advanced regeneration which can be manipulated prior to the second regeneration cut. The maintenance of sexual reproduction throughout a rotation or a significant portion of it may be important for sporadic producers such as oak species. This study was designed to determine if small sawtimber reserve white oak (*Quercus alba*) trees left after a two-age treatment could effectively initiate the development of advanced regeneration.

METHODS

The study was located at Robinson Forest, the University of Kentucky research and demonstration forest located in the Cumberland Plateau Physiographic Province in southeastern Kentucky. Twelve 2 acre 60- to 90-year-old white oak dominated stands were used in this study. Each stand was randomly assigned one of 3 treatments including an uncut treatment, a treatment leaving only 20 canopy trees per acre, and one leaving only 34 canopy trees per acre. The treatments were imposed by full crown touching release of selected canopy trees (reserve trees). These trees were of average dbh for co-dominant and dominant trees in these stands. No site preparation treatments were used. One-half acre growth and yield plots were established in the middle of each treated stand. Trees > 2.54 cm dbh were tagged and survival and growth monitored and ten 1/100th acre regeneration plots were also randomly established in each growth and yield plot.

This study reports the development of new seedlings and advanced regeneration from white oak reserve trees 15 years after treatment. Regeneration measurements were taken during July and included the number and height of each white oak stem established after treatment. To provide a relative gauge of canopy light interception and the light environment at each regeneration plot a concave spherical crown densitometer™ (Forestry Suppliers, Inc. 24 quarter inch cross hairs) reading was taken at plot center. Data was recorded and is expressed in this paper as the number of cross-hairs where open sky was observed. At the same time a series of five photosynthetic photon flux density (PPFD) measures ($\mu mol\ m^{-2}s^{-1}$ PAR) were taken at a height above ground equal to the average height of the advanced regeneration (30 cm) at every other plot center using a quantum sensor (LI-COR, Inc.) and the values averaged by plot. All PPFD and densitometer readings were taken under clear sky conditions. Advanced regeneration data were pooled by treatment and subjected to statistical analysis using ANOVA and LSD(t) to determine treatment effects. Simple linear regression was used to establish the relationship between PPFD (dependent variable) and densitometer reading (independent variable) and advanced regeneration height (dependent variable) and densitometer reading (independent variable) pooled over all treatments. The Levenberg-Marquardt algorithm was used to establish best-fit coefficients of nonlinear functions for regeneration density (dependent) and densitometer reading (independent) pooled over all treatments.

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RESULTS

Highly significant differences ($p<0.001$) were found among treatments for white oak advanced regeneration density, advanced regeneration height and densitometer readings (Table 1). The 20 reserve tree per acre treatment developed twice the number of regenerating white oak trees as the other treatments. The height of the white oak regeneration established after the treatment was greater for both cut treatments compared to the uncut treatment. The average height of the regeneration is relatively small at this point in time and would not be expected to be competitive if the stands were regenerated at this time. It is probable that some form of manipulation will be necessary to develop high vigor advanced regeneration prior to a future regeneration harvest. However, the advanced regeneration that developed after treatment indicates that the reserve trees are providing viable propagules which are developing advanced regeneration for future manipulation and stand regeneration. Densitometer readings were also higher for the cut stands compared to the uncut stands.

A positive linear relationship (y = 0.016 + 0.00724 (densitometer reading), R^2=0.733) was found between densitometer reading and PPFD indicating a relationship between measurable canopy density and light levels at advanced regeneration height (Figure 1). A positive relationship was also found between densitometer reading and advanced regeneration height (Figure 2) and densitometer reading and advanced regeneration density (Figure 3). An exponential relationship was found between densitometer reading and regeneration density (y = 43.615 + 22.373*exp(-densitometer reading/-2.489), R^2=0.974) while a linear relationship existed between densitometer reading and regeneration height (y=0.549+0.0705 (densitometer reading), R^2=0.743).

The results of this study indicate that small sawtimber sized co-dominant reserve white oak trees are capable of successfully producing advanced regeneration which will potentially aid in the long-term maintenance of this species after future regenerative treatments. A positive correlation between canopy density and regeneration height along with the positive correlation between canopy density and light level indicates that light levels developed from the treatments encouraged regeneration development. This data indicates dramatic increases in advanced regeneration density can be obtained when the combined understory, midstory, and overstory exhibit a densitometer reading greater than 6.

Table 1—Density and height of *Quercus alba* advanced regeneration developed from reserve trees in two age stands

	Density	Height	Densitometer reading
	No./ha	*Cm*	
Uncut	227[b1]	19.8[b]	5.72[c]
20 per acre	930[a]	35.0[a]	7.16[a]
35 per acre	450[b]	29.9[a]	6.47[b]

[1]Values with different letters are significantly different ($p<0.01$) using ANOVA and LSD(t).

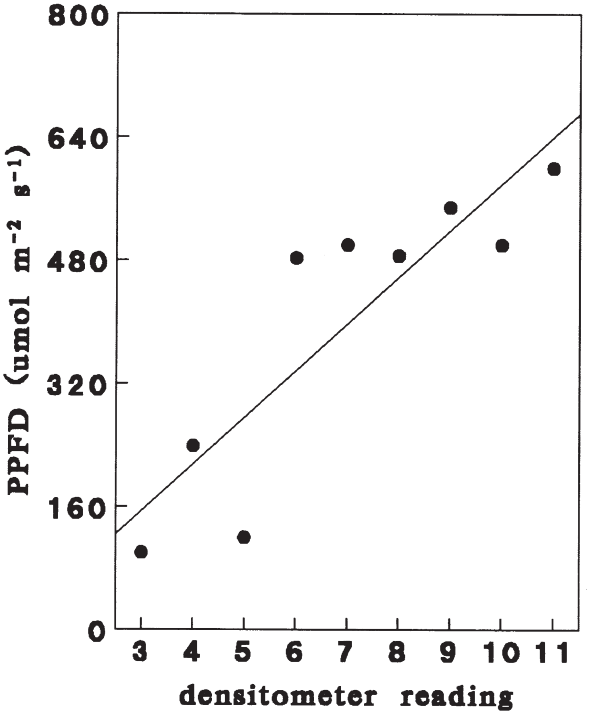

Figure 1—Data points represent average PPFD for each densitometer reading. The line represents a positive linear relationship (y = 0.016 + 0.00724(x), R^2=0.733) between densitometer reading and PPFD.

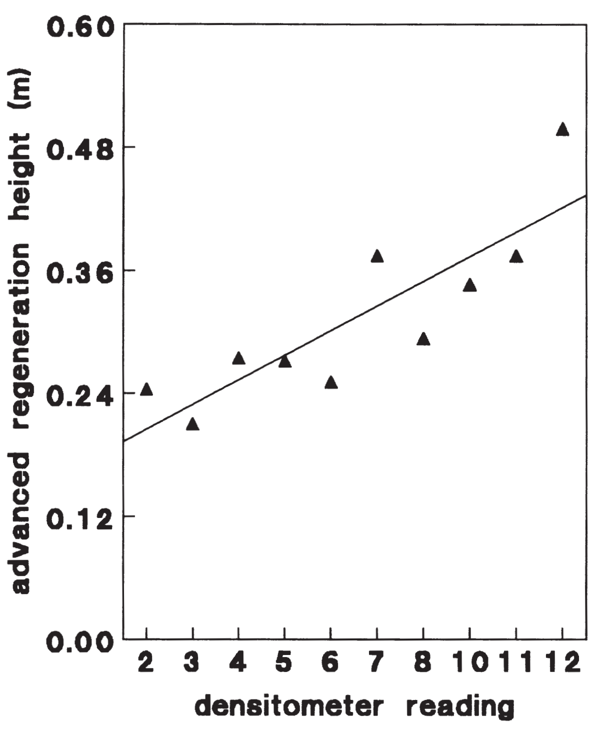

Figure 2—Data points represent average *Quercus alba* advanced regeneration stem density for each densitometer reading. The line represents an exponential relationship between densitometer reading and regeneration density (y = 43.615 + 22.373*exp(-x/-2.489), R^2=0.974).

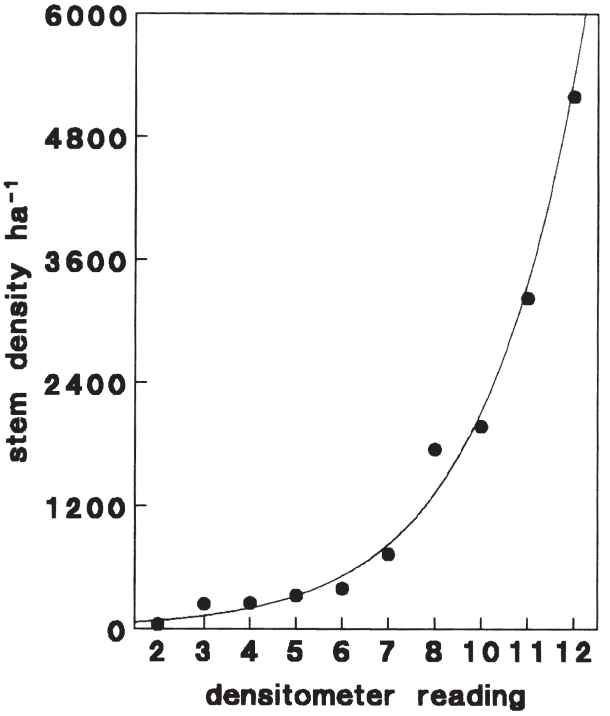

Figure 3—Data points represent average *Quercus alba* advanced regeneration height for each densitometer reading. The line represents a linear relationship between densitometer reading and regeneration height (y=0.549+0.0705(x), R^2=0.743).

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There were 32 oral presentations, 11 abstracts, and 22 poster presentations presented at the 12th Central Hardwood Forest Conference. Presentation topics included wildlife management, nutrient dynamics, stand structure, reforestation/reclamation, timber harvesting, modeling and inventory, silviculture, disturbance effects, and genetics/tree improvement.

Keywords: Air pollution, forest ecology, forest economics, forest health, forest management, harvesting, oak-hickory, reclamation, reforestation, silviculture, stand dynamics, timber harvesting, tree physiology.