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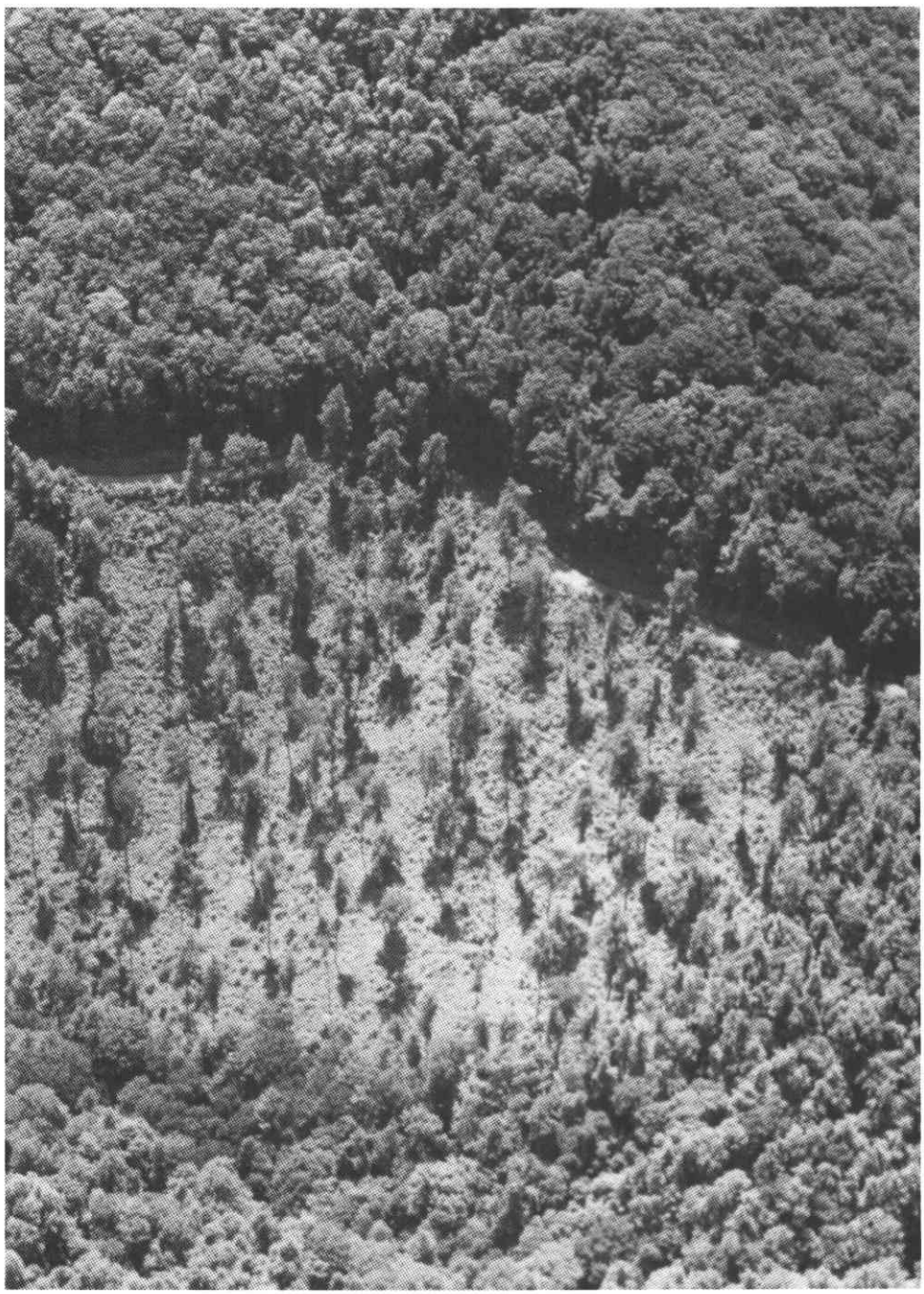

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For attending this conference, each registrant was eligible for 12 hours of Continuing Forestry Education (CFE) credit offered by the Society of American Foresters.

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ACKNOWLEDGMENTS

The editors would like to acknowledge and thank all of the people who served as manuscript reviewers for the Proceedings. The quality of the Proceedings has been improved due to your efforts. We would like to thank Becky Rosenberger for her assistance in typing the program, preparation of the table of contents, typing abstracts, proofreading, and in managing the manuscripts, reviews, and letters associated with the Proceedings. We thank Gary Miller for providing the cover photograph.

Cover Photo: This 1984 aerial photograph was taken from a helicopter three years after deferment cutting in 80-year-old central Appalachian hardwoods on the Fernow Experimental Forest near Parsons, West Virginia. (Photo by James N. Kochenderfer, USDA Forest Service.)

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Proceedings of a Meeting

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Edited by

Kurt W. Gottschalk and Sandra L. C. Fosbroke

SPONSORED BY:

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FOREWORD

This conference is the tenth in a series of biennial meetings that began in 1976 at Southern Illinois University. Other conferences have been hosted by Purdue University, University of Missouri, University of Kentucky, University of Illinois, University of Tennessee, Southern Illinois University with the North Central Forest Experiment Station (NCFES), Pennsylvania State University with the Northeastern Forest Experiment Station, and Purdue University with NCFES. The purpose of these conferences has remained the same: 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 Proceedings, a wide range of topics that represent the broad array of research programs in this area is represented.

The social and biological characteristics of the Central Hardwood Region make it unique in comparison with other forest regions of the United States. For example, one-fourth of the United States human population resides in this region. Approximately 90% of the land is in private ownership and public lands tend to be small and fragmented with private inholdings. These and related conditions play critical roles in the practice of forestry in this region. The information presented in this Proceedings is important to the long-term management of the forest resources of this unique region.

REVIEW PROCEDURES

Each manuscript published in these proceedings was critically reviewed by at least two (usually three) scientists with expertise in disciplines closely aligned to the subject of the manuscript. Reviews were returned to the senior author, who revised the manuscript appropriately and resubmitted it in a diskette format suitable for printing by the Northeastern Forest Experiment Station, USDA Forest Service where they were edited to a uniform format and type style. Manuscript authors are responsible for the accuracy and content of their papers.

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WALKING THE TALK

R. C. Kellison[1]

The preception exists that the forest industry in the United States is the antithesis of the green revolution. For instance, I recently saw the results of a poll that listed three forest products companies among the 10 worst polluters in the United States. That's all there was; their names splashed across the TV screen on the early morning news. That very appearance undoubted resulted in lost sales because some viewer would have immediately determined that those organizations, and thus the industry, were irresponsible members of society.

One wonders on what basis the poll was taken and how the results were interpreted. Did someone ride by a pulp mill on a cool morning and see plumes of water vapor emitting from pipes and stacks and mistake the innocuous condensate for "greenhouse gasses"? Or did they find the smell of hydrogen peroxide, an odiferous gas used in the chemical process of pulping, so disagreeable that the "polling team" gave a red mark to the company? Or perhaps it was the harvesting of a stand of timber that the pollster had long held in reverance as his own veiwshed even though the timber was on someone else's property.

I contend that forest industry is the leader in natural resource management and, for that matter, in industrial cleanup of the environment. What other industry has gone from about 25% to 67% in self sufficiency in energy production since the oil embargo of 1974, and what other industry has spent over a billion dollars annually during the last decade in meeting regulatory requirements? But even with that enviable record, the Environmental Protection Agency has determined that more is needed. It is proposing the "Cluster Rule" which purports to integrate EPA's various authorities over air, water and land pollution sources. The philosophy is to combine regulations, as opposed to the piece-meal approach used heretofore. The proposed rules are to be completed by 1996, and be implemented by 1999.

The American Forest & Paper Association estimates the cost of compliance of the "Cluster Rule", over three years, at $12 billion, with an additional $600 million annual operating cost. The conclusion by AF & PA and EPA is that some mills will be closed because they can't meet the standards. Even though the forest industry supports the effort to protect the environment and enhace other forest resource values, they are working to identify other, less costly ways to achieve the same results.

As a natural resource manager, forest industry sets the pace. They have surplanted the USDA Forest Service during the last decade because that agency has been beseiged by greviances, court orders and appeals that have greatly hampered their operation. Morale among its employees has suffered, and so has the management, research and extension for which the agency had worldwide recognition. Only now are they charting their path to the future.

Forest industry is also leading the way for the nonindustrial private land owner. Landowner assistance programs which have been continuously supported by organizations such as Westvaco for the last 40 years, are finding a new lease on life. Organizations without a land base, such as Stone Container and Alabama River, have become very aggressive in contracting with farmers and other nonindustrial owners in managing their forest land. This aggressiveness has caused others of the wood products industry to renew their landowner assistance programs. The reason is simple: a large part of the wood for future manufacturing needs will come from non-industry lands.

In the midst of increased production on their own lands and expanded landowner assistance programs, the forest products industry is implementing a set of principles to assure that the land they manage and, to the extend possible, the land from which they buy timber, is managed in a responsible way. The guidelines, developed by AF&PA, are

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entitled "Sustainable Forestry Principles and Implementation Guidelines". They contain principles on reforestation practices, water quality, wildlife, aesthetics, and, among other items, protecting special places. A checks-and-balance system in which the public is involved, will be used to guage the effectiveness of the program.

In keeping with the theme of this conference, forest industry is beginning to manage the native hardwood resource much more intensively than in the past. Because of the degraded condition of many of the hardwood stands in the Eastern Deciduous Forest, the best regeneration alternative is nearly always to clearcut when timber production is the objective. However, clearcutting can scar the landscape. Being sensitive to the concerns of the public, the industry has decreed that clearcutting will not be the option of choice in environmentally sensitive areas. Such areas include vistas associated with the Southern Appalachians and buffer zones of major bodies of water and recreational areas. The alternatives, even though more costly to apply and being less efficient for timber production, are to use partial-harvest systems such as shelterwood, group selection, patch clearcuts, and leave-tree cuts. Within our Hardwood Research Cooperative at North Carolina State University, the 13 industrial members and 2 state forest services have committed to the installation of 10 such studies, stretching from Virginia to Mississippi, inclusive of West Virginia. The purpose is to compare the regeneration success of a two-stage shelterwood and a leave-tree cut to a clearcut. Some of those same organizations are already applying these alternative regeneration systems to their timber stands that lie in environmental sensitive areas. This commitment to timber production in combination with environmental ethics is here to stay.

The hoped-for result from all of these actions is a list in which none of the forest products companies are among the 10 or even the 100 worst polluters. AF & PA members propose to do this by "walking the talk".

FOREST HEALTH ASSESSMENT FOR EASTERN HARDWOOD FORESTS

Daniel B. Twardus[1]

Information presented here, was obtained generally from 3 sources: the Cooperative Forest Health Protection Program, the Forest Inventory Analysis Program, and the National Forest Health Monitoring Program. The Cooperative Forest Health Protection Program is a joint State-Federal effort responsible for forest-wide surveys of forest damage. From these surveys, we gather information about the nature and extent of insect, disease, and weather caused damage to trees and forests. These surveys are conducted by State and Federal pest specialists. Forest Inventory and Analysis Surveys are conducted by the U.S. Forest Service within each State on generally a ten-year basis and provide periodic updates on such forest statistics as growth rates, volume changes, and rates of mortality and cutting. The National Forest Health Monitoring Program uses a network of permanent plots, visited annually, to monitor tree conditions such as crown vigor, signs of insect and disease damage, ozone damage, and foliar and soil chemistry. To date, plots have been established throughout New England, New Jersey, Delaware, Maryland, and the Lake States. Crown vigor is an aspect of tree condition that has become of particular interest in monitoring forest health. The Forest Health Monitoring Program has divided crown vigor into three components: foliage transparency, crown density, and crown dieback. Foliage transparency is a measure of the "fullness" of a tree's crown. That is, the greater the transparency the more light passes through, indicating that foliage may be missing. Changes in foliage transparency occur as a result of insect feeding, deformed foliage, or reduced amounts of foliage. Crown density, estimates a tree's crown in relation to a normal or expected shape and size. It is the amount of crown branches and foliage that block light coming through the canopy. Crown dieback, is defined as branchlet or twig mortality in the upper portions of the crown. Crown dieback is often thought of as the first sign of tree stress. Taken together, these three crown rating factors can be used to assess crown vigor.

FOREST CONDITIONS

Beech (*Fagus grandifolia*)

The forest type known as maple/beech/birch encompasses nearly 5 million acres in the east. The largest concentrations of beech are found in New York, Pennsylvania, and West Virginia. In New England and the mid-Atlantic States, the volume of beech has generally increased with the exception of Maine where a 6% decrease in growing stock volumes has occurred (Tables 1 and 2). New York and Vermont have higher than average annual beech mortality rates, as a percent of growing stock volume.

A very important factor in the health of beech, is the beech bark disease. This disease, first introduced into Maine in the 1930's from Canada has spread throughout New England into Pennsylvania and West Virginia. By the mid-1970's, it was estimated that over 30 percent of the beech in New Hampshire and Vermont and 50 percent of the beech in Maine over 8 inches in diameter were killed by this disease. It is worth noting that in New England, the percent of growing stock volumes in trees less than 15 inches in diameter is quite high. In Maine, where beech is decreasing, 86% of the growing stock volume is in smaller trees. Mortality has also occurred on over 500,000 acres of forest containing beech in West Virginia, and on over 250,000 acres in New York. What we are seeing in the New England and mid-Atlantic States is that older, larger beech are being killed by the disease and then replaced by new, smaller beech. In many instances, these young trees are sprouts and are just as susceptible to the disease. Most become infected by beech bark disease, but because they are able to outgrow infection, they are not always killed. However, this so-called "aftermath" forest consists of many highly deformed, defective trees.

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Results of Forest Health Monitoring, illustrate that for American beech, nearly 47% of the trees have poor crown density and slightly more than 27% have severe crown dieback. Though only small percentages of the trees are being monitored, the results are significant in that they are higher percentages than for other hardwood species.

Table 1. Change in volumes and annual mortality for American Beech - Mid-Atlantic States (data from FIA survey reports).

State	% change in growing stock volume between inventory periods ('72-'92)	% of growing stock volume in trees less than 15 inches	annual mortality as % of growing stock volume
DE	-47	47	
NJ	85	56	.1
MD	30	49	0
PA	24	63	.5
WV	19	52	.2
OH	64	38	.6
Average	34	52	.3

Table 2. Change in volumes and annual mortality for American Beech - North Central States (data from FIA survey reports).

State	% change in growing stock volume between inventory periods	% of growing stock volume in trees less than 15 inches	annual mortality as % of growing stock volume
MI	-18	63	.7
MO		32	.6
IL	-16	20	.4
WI	13	51	2
IA			
Average	-7	42	.9

Maples (*Acer* spp.)

Of the hardwood species growing in the East, oaks are the most common accounting for one-third of the hardwood volume (Powell et al. 1993). The maples (*Acer* spp.) are next in abundance, and are one of the fastest growing components of our hardwood forest. Sugar (A. saccharinum) and red maples (A. rubrum) together account for 17% of all hardwood volumes in the East (Powell et al. 1993). Table 3 illustrates changes in growing stock volumes for both sugar and red maples. The largest gains in sugar maple volumes occurred in Ohio with a whopping 98% increase in growing stock volumes between 1979 and 1991, and a 138% increase in red maple volumes for the same time period.

Why the increases? Both sugar and red maple are very tolerant of shade and both are prolific seeders. Maples will increasingly occupy sites as openings are created or other species are cut and removed. Large increases in one species mean others loose ground. In Ohio, red maple has replaced hickory as the most abundant tree species in the forest.

In recent years there has been concern expressed over the health of sugar maple. Sugar maples, used for the production of maple syrup and as a roadside and yard tree, are a valuable component of our forests in the east. In many cases, various stressing agents have been associated with dieback and decline of individual sugar maples, especially insect defoliators and physical damage. In northern New York the forest tent caterpillar defoliated over 120,000 acres of sugar maple in 1993. Pear thrips defoliation continues in Vermont, Pennsylvania, and New York, however, the area affected is down from the 1980's when the most extensive damage occurred. In northern Pennsylvania, an area of 260,000 acres has been defoliated for several years by the elm spanworm. Most heavily impacted within this area has been red maple and American beech.

The North American Maple project has been collecting information at permanent sample sites throughout eastern Canada and the northeastern US. Results of this effort have shown that more than 90 percent of all sugar maples within the survey were considered to have healthy crowns. Of those with more than 50% dieback, nearly all had major root or trunk damage. Within the survey area, sugar maple mortality is estimated to be only 1 percent of trees per year. This isn't to say the local problems won't develop. In Pennsylvania, for example, red and sugar maple volumes are up, statewide. However, several northwestern counties in an area known as the Allegheny Plateau are experiencing decreases in sugar maple growing stocking relative to other species. Some of this may be due to the affects of Pear thrips defoliation, or just to an overabundance of sugar maple resulting in increased competition for soil and water resources. Experts have been gathering information related to this localized problem, but answers are not yet available.

Oaks (*Quercus* spp.)

Our present oak-hickory forests rose out of the natural catastrophe known as the chestnut blight. Forests once referred to as oak-chestnut covered huge areas of the eastern US from southern New England to northern Georgia. Chestnut blight, caused by a fungus, was introduced to this country in 1904, and by the 1930's virtually all chestnut trees were infected and dying. Natural replacement of the chestnut followed resulting in forests in which oaks, yellow poplar, and hickory became dominant. Today, we refer to these replacement forests as oak-hickory forests, but as the noted ecologist Lucy Braun wrote in 1950, "it is impossible as yet to predict the final outcome of the partial secondary successions everywhere in progress."

Within the present day oak-hickory forests, a new forest health factor looms--the gypsy moth. This defoliating insect was introduced into the US in 1890 in Massachusetts. It has gradually spread throughout the East, now occupying a place in the forests from Maine to Virginia and west to Wisconsin. The larvae of the gypsy moth feed on a variety of hardwood species but favor oaks (*Quercus* spp), sweetgum (*Liquidambar styraciflura*), basswood(*Tilia americana*), and willow (*Salix* spp). Repeated gypsy moth defoliation can result in growth loss, tree mortality, changes in forest composition, and loss of habitat for various animal species. In some areas, oak mortality due to gypsy moth defoliation has been quite significant. Areas of south-central Pennsylvania have experienced extensive oak mortality during outbreaks in the late 1980's. Gypsy moth will not disappear from these forests, and a new balance must be struck between forest and insect. Overall, in Pennsylvania a state with a 20 year history of gypsy moth defoliation, oaks still account for 43 percent of the total forest volume (Gansner et al. 1993).

Table 3. Change in growing stock volumes for hard and soft maples (data from Forest Inventory Analysis Reports (1972-1992)).

	sugar	red	years
	----------percent change----------		
Mid-Atlantic States			
NJ	-	-	
PA	20	37	1978-1989
WV	56	74	1975-1989
OH	98	138	1979-1991
MD	47	56	1976-1986
DE	-	-	
Average mid-Atlantic	55	76	
Northcentral States			
WI	48	79	1967-1982
MI	21	78	1965-1979
IL	63	32	1962-1985
IN	65	102	1967-1986
MN	40	94	1977-1989
IA	4	44	1974-1990
Average Northcentral	40	55	

Hickories (*Carya* spp.)

Of the total amount of hardwood volume in the East, oaks comprise nearly a third while hickories account for only about 4% . Recently, some concern has been raised over declining hickories in various parts of its range-notably Wisconsin, West Virginia, and Ohio. Table 4 shows that though hickory growing stock has increased in most states during the past 20 years, much of this increase has occurred in trees of smaller size. Larger trees (over 15 inches in diameter) appear to be succumbing to mortality or cutting in Minnesota, Delaware, Rhode Island, New York, Massachusetts, and Connecticut. States with higher than average rates of hickory mortality are Massachusetts, Rhode Island, Missouri, and Illinois. At present, the causes of hickory decline in these areas is unknown though several wood-boring insects are suspected of playing a role.

White Pine (*Pinus strobus*)

White pine extends from Maine to the Lake States and south throughout the Appalachian Mountains. It grows in a variety of conditions from pure stands to stands in which it is only a minor component. I mention it here, because in the East, it is found associated with many different hardwood species including aspen, birch, oaks, and hickory, and because it is coming back. White pines' moderate shade tolerance, its long life, and its resistence to fire, make it a very adaptable species. The lumber industry of the east was based upon white pine for much of the 1800's and early 1900's. Today, however, (and due, in part, to exploitive logging) white pine sawtimber is only regionally important. In total, white and red pines account for only about 3 percent of the volume of softwood growing stock in the US. In most areas of the East, white pine is coming back. Growing stock volumes are up 105% in West Virginia, 52% in Connecticut, and 43% in Pennsylvania, all in the last 20 years. Within the Northeastern area, today, Michigan has the largest concentration of white and red pines totaling an estimated 2,017 million cubic feet of growing stock, followed by Maine with 1,916, and New York with 1,809.

Table 4. Change in volumes and mortality rates for hickory (data from Forest Inventory Analysis Reports, 1972-1993).

State	% change in growing stock volume between inventory periods	% of growing stock in trees less than 15 inches in diameter	annual mortality as a % of growing stock volume
CT	19	85	0.5
NY	23	82	.4
MA	-22	92	2.1
RI	73	95	.7
NJ	---	78	.3
MD	12	67	.6
PA	12	79	.8
WV	32	77	.4
OH	35	68	.6
DE	- 70	71	NA
IN	19	67	.5
MO	45	85	.8
IL	52	70	1.0
MN	54	87	.3
MI	3.0	79	.1
WI	82	90	.2
IA	74	82	.4

Ash (*Fraxinus* spp)

White ash and black ash are components of the elm-ash-cottonwood forest type which encompasses over 10 million acres in the northeastern and northcentral states from Maine to Minnesota. White ash is the most common native ash species, while black ash is found more often along streams and rivers.

In the central Appalachians, ash growing stock increased 36 percent since the last inventories. However, some problems have been occurring. A disease, ash yellows, caused by a mycoplasma-like organism, has been confirmed occurring in almost every state within the Northeastern Area. Not all ash problems can be attributed to this disease, since damage from drought, insects, and rust diseases are also found in association with declining ash trees.

Decline of black ash was reported in 1993 in Maine on 120,000 acres. Numerous trees with severe crown dieback have been observed along streams in Maine. In New York, ash anthracnose, a foliage disease was reported in 1993 on 105,000 acres.

Results of FHM assessments of white ash crown conditions in 1993 show that white ash is second only to American beech in percent of trees with severe dieback and poor crown density.

Butternut (*Juglans cineraea*)

Butternut, or white walnut, is associated with many other hardwood species and occurs sporadically throughout the east. The tree is most valued for its nut production for wildlife and for furniture wood. In the past the majority of butternut timber production has been in West Virginia, Wisconsin, and Indiana. However, it is hard to find healthy butternut in forest stands today, as the species is being threatened by a devastating disease known as butternut canker caused by *Sirococcus clavigignenti-juglandacearum*, a fungus of unknown origin.

The disease was first reported occurring in Wisconsin in the mid-1960s, but has probably been in the United States prior to that time. It is now present throughout the range of butternut. The cankers girdle the branches and tree stems, causing mortality. There is no known control for butternut canker disease. Efforts now center on finding resistant individual trees within infected stands.

In 1993, a ground survey confirmed the presence of the disease in central coastal Maine. Surveys for the disease were also conducted in New York, where most of the counties surveyed were found infected, and Vermont, where the presence of the disease was confirmed throughout the state. In Wisconsin, where over 1000 trees were samples in 32 counties, 91 percent of the live trees were found to be infected and 27 percent of the surveyed trees were dead. In the last decade the number of live butternut decreased 58 percent in Wisconsin and 84 percent in Michigan. Several evaluation surveys indicate that the disease has spread rapidly within infected stands.

Two other factors affecting the eastern deciduous forest that are worth mentioning are **weather** and **air quality.**

Floods

In the summer of 1993, in the mid-west, thousands of acres of bottomland hardwood areas were flooded for 4 to 12 weeks. The flood waters also backed up into upland areas, affecting urban areas forested with trees not well adapted to flooding. The states along the Missouri and Mississippi Rivers and their tributaries were severely affected. Flood stressed trees exhibited a range of symptoms including leaf yellowing, early leaf drop, and dieback. It is possible that these affected trees will show symptoms for several years. They may also be affected by various insects and diseases.

Air quality

There are 3 aspects of air quality that are presently of concern with respect to the health of eastern deciduous forest; ozone, and emissions of sulfates, and nitrates.

Ozone. Ozone is an air pollutant in the lower atmosphere, formed from the reaction of compounds produced by automobile emissions and industrial processes, in combination with sunlight. There is increasing concern as to how ozone may be influencing the health of the forest resource. Atmospheric concentrations are monitored through a network of ozone monitoring stations throughout the area. And, damage to sensitive plant species, called bioindicators, has been assessed on National Forest Lands and on State and Private Lands as part of the National Forest Health Monitoring program. The results for 1993 indicate that there was little ozone injury encountered. This is mainly because of the low incidence of damaging ozone in the atmosphere during the summer of 1993. The incidence of damage was higher in 1990, which was a year when ozone concentration were high due to dry and hot summer conditions.

Using bioindicator plants allows for a general assessment of the air quality. Research is currently being conducted to determine the link between visible symptoms and foliar injury. At this time, the bioindicator information cannot be used to make a statement about the effects on the productivity of or damage to the forest. We know that across a broad range of ozone concentrations dose-responses of net photosynthesis and growth of trees ar dependent upon ozone concentrations. We know that significant differences exist amount species in sensitivity to ozone. And, we know that ambient ozone concentrations are probably sufficient to a cause reductions in photosynthesis and growth in most vegetation of the east. However, we do need long-term ecological studies of the interactions between ozone and natural factors before we can quantify the response of forest ecosystems to ambient ozone pollution (Reich 1987).

Sulfates and nitrates. We know that the eastern US receives the highest depositions of sulfates and nitrates in the country, and that, at least for sulfates, 90% of this deposition is due to man-made sources (Galloway and Whelpdale 1980). The effects of these air borne pollutants upon forest ecosystems was described by Smith (1974) as ranging anywhere from subtle changes in species composition and forest structure to obvious structural simplification and major changes in ecosystem function. Presently, however, adverse effects of the deposition of sulfates and nitrates (acid rain) are limited to high-elevation spruce forests. The National Acid Precipitation Program (NAPAP) was unable to provide evidence of widespread forest damage in the US. (Barnard et al. 1989). Although no direct adverse effects on terrestrial vegetation has been documented, indirect effects may result from long-term changes in soil chemistry. In fact, NAPAP, concluded that changes in soil chemistry may progress for 50 to 60 years before effects on forests are observed. These concerns must be taken seriously and links between possible changes in soil chemistry and the incidence of disease and insect attacks investigated. For example, it is plausible that changes in soil chemistry may alter tree physiology to the extent that resistance to disease or insect attack is lessened.

CONCLUSIONS

Overall, if we look at all hardwoods in the East using data compiled by the USDA Forest Service for the Resource Planning Act Assessments, we find increasing volumes and mortality generally stable, except for the last few years (Fig. 1).

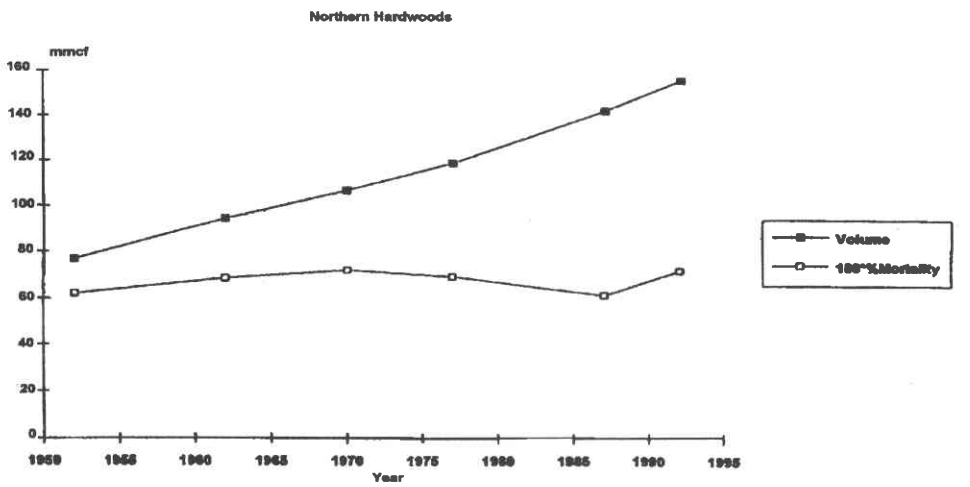

Figure 1. Data from Resource Planning Act Assessments. Personal communciaton, Eric Smith, USDA Forest Service, Ft. Collins, Co.

Evaluating forest health requires analysis of several conditions. In this paper, I have examined incidence of insects and disease, growth, rates of mortality, and species composition changes. Growth or measures of productivity, by themselves, do not reflect forest health. For the eastern hardwood forest, it is important to evaluate not only growth, but factors such as mortaltiy rates, diverstiy, changes in species composition, the effects of mangagement and fragmentation, and the exogenous effects of changes in air qualtiy. Health, as a concept in forestry is only beginning to be understood. On the surface, our tendency is to view the health of the forest from an organismic viewpoint. But forests exist as complex communities, constantly changing, and constantly being affectted by what happens within the environment within which they too exist.

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CHARACTERISTICS AND DYNAMICS OF AN UPLAND MISSOURI OLD-GROWTH FOREST

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Abstract: We describe the structure, composition, and dynamics of Schnabel Woods, an 80-acre mixed mesophytic old-growth stand located in the loess-covered River Hills adjoining the Missouri River floodplain in central Missouri. In 1982 all trees on a 1.5-ac northwest-facing plot and a 1.5-ac southeast-facing plot were mapped and measured. In 1992 we remeasured those plots to observe current conditions and changes in species composition and stand structure over time. In 1992 density was 381 trees/acre, basal area was 124 ft²/ac, and the mean dbh was 5.2 in (for trees ≥ 0.8 in dbh). The stand was fully stocked at 101%. Between 1982 and 1992 tree density decreased by 11% while total basal area remained constant. However, basal area did shift among species groups; basal area for northern red oak and white oak increased by 10% while basal area for other species groups remained the same or decreased. The volume of down wood with a minimum diameter ≥ 4 in averaged 338 ft³/ac over both plots. Litter dry weight averaged 4.9 tons/ac and was significantly greater on the southeast than the northwest aspect. Northern red oak dominated the overstory, but maple dominated the understory and comprised 67% of the ingrowth. Current dynamics are driven by fine-scale treefall gaps, where maples are replacing oaks.

INTRODUCTION

Schnabel Woods is one of the few remaining remnant old-growth tracts in the loess-covered River Hills of Boone County, within Missouri's Ozark Border Natural Division (Thom and Wilson 1980). Less than 0.07% of the original old-growth forest in the central hardwoods region remains (Parker 1989), and in Missouri only 0.05% of the existing forest is classified as old-growth (Shifley 1994). Remnant old-growth tracts such as Schnabel Woods are of interest because they are rare, but more importantly they provide the best available information about probable long-term patterns of development and succession in the surrounding second-growth forests. To the extent that silvicultural practices resemble gap dynamics, old-growth can also reveal successional tendencies in managed stands (Abrams and Scott 1989). Moreover, old-growth forests are at one end of the spectrum of forest development. As such they provide an ecological frame of reference from which to quantitatively evaluate the current structure, composition, diversity, and dynamics of second-growth forests.

Schnabel Woods was first studied by Kucera and McDermott (1955) who reported that dominant white oaks[5] averaged 165 years old with dbhs (diameter at breast height) between 27 and 31 in. The species with the highest importance values were sugar maple, northern red oak, and basswood. At that time they noted that the tract was "representative of the original forest" and Kucera (personal communication, Nov. 94) found no evidence of significant anthropogenic disturbance at the time of his study.

Since Boone County was first surveyed in 1816-17, the species composition of Schnabel Woods appears to have gradually shifted from predominantly oak to predominantly maple. Wuenscher and Valiunas's (1967) study of

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[5]Scientific names of all species are given in Table 4.

surveyor records for the River Hills of Boone County indicates that in the first quarter of the 19th century, the importance value of white oak exceeded that of sugar maple. Similarly, Howell and Kucera's (1956) study of surveyor records for Boone County, Missouri, showed that 58% of the witness trees in the original land surveys were *Quercus* spp., primarily white oak and bur oak (39% and 16%, respectively). By 1955, Kucera and McDermott (1955) reported that the importance value of sugar maple was almost twice that of red oak. Species with lower shade tolerance followed in importance ranking: *Tilia americana, Ulmus* spp., *Fraxinus* spp., and *Quercus* spp. The transition from an oak-dominated forest to one with more shade tolerant species, particularly sugar maple, is consistent with findings for numerous other mesic mature and old-growth remnants in the central hardwood region (e.g., Nigh and others 1985, Schlesinger 1989, Shotola and others 1992, Wuenscher 1967, Lindsey 1962, Parker and Leopold 1983, McGee 1984, Barton and Schmelz 1987).

In this paper we describe the current structure, composition, growth, and mortality at Schnabel Woods. We compare the structure and composition of a northwest-facing and southeast-facing aspect in Schnabel Woods and we examine the dynamics over a ten-year interval. In addition, we compare our findings with Kucera and McDermott's (1955) compositional study.

METHODS

Study Area

Schnabel Woods is located approximately 2.5 miles southeast of McBaine in Boone County, Missouri (Figure 1). The 80-acre tract sits in the River Hills immediately east of and above the Missouri River floodplain. The deep Menfro loess soils are capable of supporting rich mesic upland forest and diverse ephemeral ground flora (Kucera and McDermott 1955). With the exception of this tract, the surrounding area has been extensively cut over, grazed, and/or farmed. During the 1940s light selective logging of some high quality timber took place in or around Schnabel Woods (Tom Douglas[6], personal communication, 1994). In 1954 the tract was deeded to the University of Missouri as an arboretum

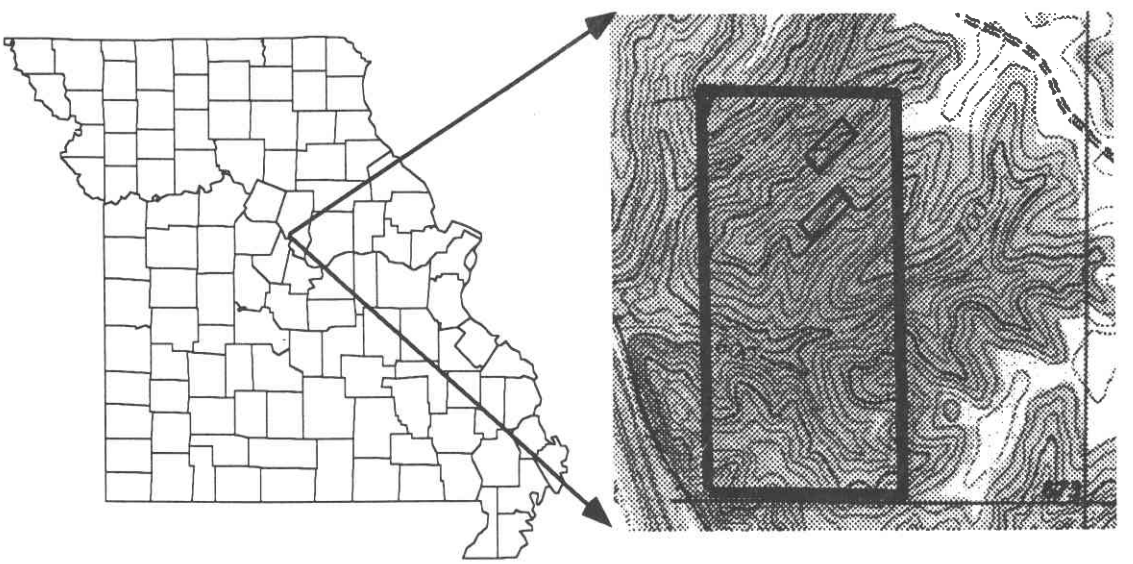

Figure 1. Boundary of Schnabel Woods showing approximate plot locations, Boone County, Missouri. Schnabel Woods legal description is W ½, SE ¼, Sect. 7, T48N, R13W.

[6]Tom Douglas owns the farm adjacent to Schnabel Woods and is a long-term resident of the area.

and demonstration woods. Since acquisition by the University of Missouri, no active vegetative management has occurred. In 1978 the tract was designated a state natural area. Three small fires have been documented as well as a few brief incidents of grazing by stray cattle from the surrounding pastures.

Field Procedure

In 1982 Chaplin and students mapped all live trees ≥ 0.8-in (2.0-cm)[7] dbh in two 1.5-ac (0.6-ha) plots. To compare the current composition and structure of two separate aspects, one plot was located on the major northwest-facing slope in the tract; the other plot was placed on the opposite southeast-facing slope on the same ridge (Figure 1). The plots have a vertical relief of approximately 100 ft and a slope of approximately 35%. Although placement of plots was not random, we believe these plots are representative of Schnabel Woods.

Species and dbh were recorded for each live tree ≥ 0.8-in dbh. Stem maps for each plot were created by subdividing each plot into sixty 0.025-ac (100-m²) square subplots and recording the Cartesian coordinates of each tree within its respective subplot. In 1992 we relocated the plots and measured dbh, crown class (dominant, codominant, intermediate, overtopped), crown ratio percent, and status (live, dead, ingrowth) for each tree ≥ 0.8 in dbh. On every second subplot (half the total sample area) the number of cavities in each individual tree was recorded by minimum opening size (1-2, 2-4, or 4+ in; 2-5, 5-10, or 10+-cm). Ingrowth during the ten-year period was defined as any live tree (≥ 0.8 in) present in 1992 that was not recorded in 1982. In 1992 down woody material ≥ 4-in (10 cm) in diameter was recorded by length and midpoint diameter on the main plots. Volume of each piece of down wood was computed as the volume of a cylinder, and each piece was classified into one of five decay stages following the classification system of Maser and others (1979). Within each main plot, forest floor litter was collected from four systematically established locations. Each litter sample totaled 10.8 ft² (1.0-m²) in area, and was collected as the composite of eight 1.4-ft² (0.125-m²) subplots. At four of the litter subplots, soil was collected to a depth of 20 cm and sent to an independent lab for organic matter determination. In 1982 soil samples were also collected and analyzed for percent organic matter by the weight loss on combustion method.

Analysis

Density (i.e. number of trees per unit area), basal area, and stocking were calculated for each tree and summed to obtain plot means or class summaries. Stocking was computed using the tree area ratio equations of Gingrich (1967) and Stout and others (1987). Trees were grouped into 2-in (5-cm) dbh classes to observe and compare dbh frequency distributions. Species groups (Table 1) were created to clarify trends for those species which were relatively rare. Contingency table analysis (chi-square test) was used to test for differences in the diameter distributions and species groups distributions between aspects. Student's t-test was used to test for differences in dry litter weight between aspects. Spatial patterns were tested using Ripley's K(t)or K12(t) (Diggle 1981, Ripley 1977). The K(t) function uses distances between mapped objects (e.g. trees) to measure the degree of clumping (attraction), regularity (repulsion), or randomness (independence) among individuals.

RESULTS

Structure

In 1992 the average density of live tree and shrub stems ≥ 0.8 in dbh across both plots was 381 trees/ac (942/ha). At 452 trees/ac (1116 /ha), density on the southeast aspect was 50% greater than that observed on the northwest aspect. However, the basal area on the southeast aspect was 10 ft²/ac (2 m²/ha) less than on the northwest aspect (Table 2).

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[7]A full set of metric equivalents is given following Literature Cited.

Table 1. Species groups used to classify trees at Schnabel Woods.

Species Group	Included species
Ash	*Fraxinus americana, F. quadrangulata, F. pennsylvanica*
Maple	*Acer saccharum, A. nigrum*
Red oak	*Quercus rubra, Q. shumardii*
White oak	*Q. alba, Q. muehlenbergii*
Hickory	*Carya laciniosa, C. texana, C. cordiformis*
Elm	*Ulmus rubra, U. americana*
Basswood	*Tilia americana*
Understory	*Cornus florida, Ostrya virginiana, Asimina triloba, Cercis canadensis*
Mesic	*Juglans nigra, Celtis* spp., *Gymnocladus dioicus*

Table 2. Summary of mean stand characteristics observed at Schnabel Woods, 1982-1992.

Characteristic	Live trees (trees/ac)	Live basal area (ft^2/ac)	Mean dbh (in)	Stocking %	Standing dead basal area (ft^2/ac)
1982 Total					
Combined	452	124	4.6	102	--
NW	367	124	5.3	103	--
SE	539	123	4.2	101	--
1992 Total					
Combined	381	124	5.2	101	9.37
NW	310	129	6.0	104	6.32
SE	452	119	4.6	97	12.4
Net change 1982-1992					
Combined	-71	0	0.6	-1	
NW	-57	5	0.7	1	--
SE	-87	-4	0.4	-4	--
Relative change 1982-1992					
Combined	-16%	0%	13%	-1%	--
NW	-15%	4%	13%	1%	--
SE	-16%	-3%	10%	-4%	--

The combined diameter distribution for both aspects has a reverse J-shape (Figure 2) and 86% of the trees are < 10 in (25 cm) dbh. Interestingly, the shapes of the diameter distributions for live and dead trees are very similar, but there is a 10-fold difference in densities. This is consistent with observations for other old-growth sites in Missouri (Shifley and others 1995).

Diameter distributions were significantly different on the two aspects (chi-square, $p = 0.004$, 15 d.f.). (Figure 3). The southeast aspect had almost twice as many stems < 10-in dbh than the northwest aspect.

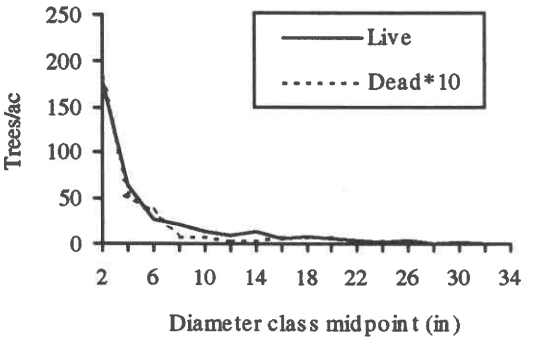

Figure 2. Number of live and dead (scaled by a factor of 10) trees/ac in 2-in dbh classes at Schnabel Woods, 1992.

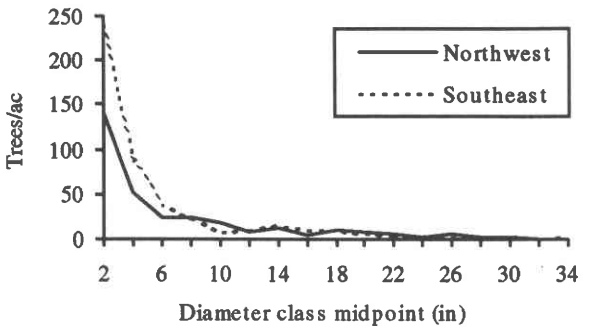

Figure 3. Number of live trees/ac by aspect in 2-in dbh classes at Schnabel Woods, 1992.

Five percent of the standing trees (live or dead) had natural or excavated cavities. The probability of a tree of a given dbh having one or more cavities increased linearly from 8% for a 4-in (10-cm) dbh tree to 30% for a 30-in dbh tree.

Down wood ≥ 4 in dbh on the two plots averaged 563 ft³/ac. Most of this volume was in the intermediate decomposition class (Table 3). The number of pieces decreased exponentially with increasing volume per piece (Figure 4). The largest piece of down wood we observed had a volume of 88 ft³. There were relatively few pieces shorter than 3 ft.

Table 3. Volume of down wood ≥ 4 in diameter at Schnabel Woods, 1992. Decomposition classes follow Maser and others (1979).

Decomposition class	Defining characteristics	Volume (ft³/ac)
Class 1	Round log, bark intact, small twigs present, log elevated on support points	0
Class 2	Round log, bark intact, twigs absent, log elevated on support points	130
Class 3	Round log, traces of bark, twigs absent, log sagging near ground	286
Class 4	Round to oval log, bark absent, twigs absent, log entirely on ground	101
Class 5	Oval log, bark absent, twigs absent, log entirely on ground	46
Total		563

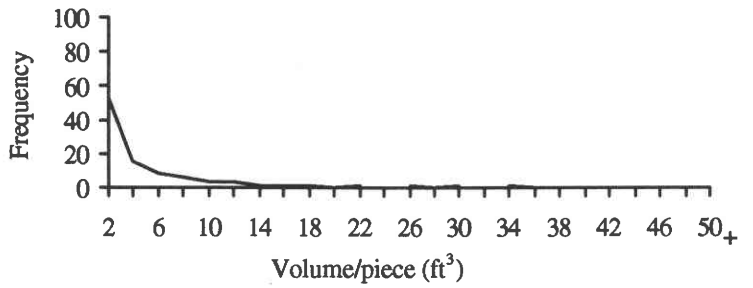

Figure 4. Frequency of down dead wood by volume (ft^3) class midpoint for Schnabel Woods, 1992. Note that graph stops at 50 ft^3 but the largest piece was 89 ft^3.

Soil organic matter averaged 8.6% in 1982 and 8.5% in 1992. No difference was observed between measurement intervals (t-test, p=.47, 20 d.f.), between aspects in 1982 (t-test, p=.47 , 4 d.f.), or between aspects in 1992 (t-test, p=.25 14 d.f.).

Dry weight of litter on the forest floor averaged 4.9 tons/ac (1.1 metric tons/ha). The mean mass of litter was significantly greater on the southeast (5.6 tons/ac) than the northwest plot (4.2 tons/ac) (t-test, p = 0.002, 6 d.f.).

Species Composition

Twenty-two tree species were observed at Schnabel Woods in 1992: 19 on the northwest aspect and 17 on the southeast aspect (Table 4). Eight species on the northwest aspect and ten on the southeast aspect had fewer than five stems. In addition, three vine species were observed on both aspects.

Sugar maple was by far the most common species on both aspects at Schnabel Woods, comprising 74% of the trees and 35% of the basal area (Table 4). In terms of importance value, northern red oak was the second most dominant species. The magnitude of the difference between a species' relative basal area and relative density (Table 4) gives some indication of that species' relative size structure. For example, the relative basal area for northern red oak is approximately 10 times its relative density, while sugar maple has a relative basal area that is half of its relative density. This indicates that northern red oaks are, on average, substantially larger in diameter than sugar maples. Dominance of northern red oak in the overstory and sugar maple in the understory is clearly illustrated by Figure 5, which shows the relative basal area by crown class. Maples comprise over 80% of the basal area at lower crown positions but < 20% in the dominant crown class. Northern red oak accounts for 64% of the basal area in the dominant crown class but is absent in the overtopped crown class. Differences among species by aspect were most notable in the smaller size classes. Maple comprised 93% of trees $\leq$ 10 in dbh on the southeast aspect, but only 67% of the trees on the northwest aspect (Table 5). The understory species group (*Cornus florida, Ostrya virginiana, Asimina triloba*, and *Cercis canadensis*) comprised an additional 19% of the stems $\leq$ 10 in dbh on the northwest aspect.

Compared to importance values reported for Schnabel Woods nearly 30 years earlier by Kucera and McDermott (1955), the importance of sugar maple increased by 20%, primarily at the expense of basswood (Table 4). However, given that the field sample by Kucera and McDermott did not necessarily coincide with our sample site, there is some ambiguity associated with interpreting such differences.

Table 4. Importance percentages for Schnabel Woods, northwest and southeast aspects combined.

Species	1992				1955
	Aspect where spp. occurs	Relative density	Relative basal area	Importance value	Importance value[a]
Sugar maple (*Acer saccharum*)	NW, SE	74.2	35.1	54.7	35
Northern red oak (*Quercus rubra*)	NW, SE	2.9	27.3	15.1	19
Chinquapin oak (*Quercus muehlenbergii*)	NW, SE	3.3	11.4	7.3	6
American basswood (*Tilia americana*)	NW, SE	1.9	5.9	3.9	18
White oak (*Quercus alba*)	NW, SE	0.9	5.2	3.0	0
Slippery or red elm (*Ulmus rubra*)	NW, SE	2.8	2.1	2.5	0
White ash (*Fraxinus americana*)	NW, SE	1.8	3.1	2.4	0
Black maple (*Acer nigrum*)	NW	1.5	2.3	1.9	0
Flowering dogwood (*Cornus florida*)	NW	3.5	0.3	1.9	4
Shagbark hickory (*Carya texana*)	NW, SE	1.1	1.7	1.4	0
Eastern hophornbeam (Ostrya virginiana)	NW, SE	2.6	0.2	1.4	4
Green ash (*Fraxinus pennsylvanica*)	NW, SE	0.5	1.9	1.2	4
American elm (*Ulmus americana*)	NW, SE	0.8	1.0	0.9	6
Kentucky coffeetree (*Gymnocladus dioicus*)	NW, SE	0.4	0.7	0.6	0
Shellbark hickory (*Carya laciniosa*)	SE	0.2	0.6	0.4	0
Pawpaw (*Asimina triloba*)	NW	0.7	0.0	0.4	0
Shumard oak (*Quercus shumardii*)	SE	0.1	0.6	0.3	0
Hackberry (*Celtis* spp.)	NW, SE	0.4	0.3	0.3	0
Black walnut (*Juglans nigra*)	NW	0.2	0.3	0.3	0
Blue ash (*Fraxinus quadrangulata*)	NW, SE	0.2	<0.1	0.1	0
Eastern redbud (*Cercis canadensis*)	SE	0.1	<0.1	0.1	0
Bitternut hickory (*Carya cordiformis*)	NW	0.1	<0.1	<0.1	4

Importance value are computed as {(relative density + relative basal area)/2}.
[a]Adapted from Kucera and McDermott (1955)

Table 5. Relative density of live trees < 10 in dbh in each species group by aspect.

Group	Northwest	Southeast
Maple	67.4	93.2
Understory	19.0	0.7
Elm	7.9	1.0
Ash	2.6	1.0
Mesic	1.3	0.5
White oak	0.8	1.7
Hickory	0.5	1.2
Basswood	0.5	0.7

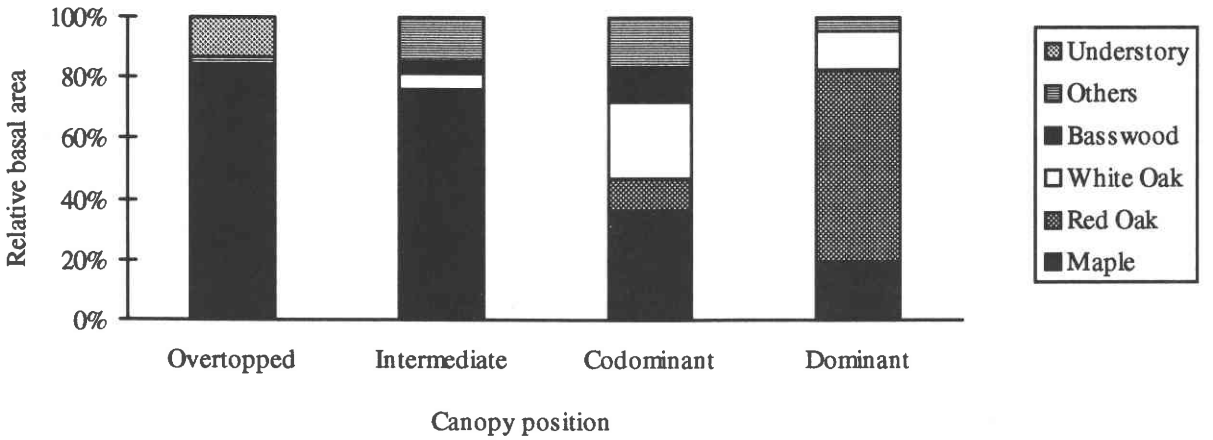

Figure 5. Relative basal area by canopy position at Schnabel in 1992. Note that "others" includes the hickory, ash, elm, and mesic species groups.

The number of trees by species group was significantly different between the two aspects (chi-square test, $p \leq 0.001$, 8 d.f.). Overall, more trees in the ash, elm, red oak, and understory species groups were found on the northwest aspect while maple and white oak species groups were more prevalent on the southeast aspect. Basswood had approximately the same number of stems on both aspects. Interestingly, only 4 stems (0.7 trees/ac) consisting of two species (eastern hophornbeam (3) and redbud (1)) were observed in the understory group on the southeast aspect while 73 stems (49 trees/ac) consisting of three species (flowering dogwood (39), eastern hophornbeam (26) and pawpaw (8)) were on the northwest side.

Forest Dynamics (1982-1992)

Basal area and stocking remained virtually unchanged over the ten-year period at 124 ft^2/ac and 101%, respectively (Table 2). However, the density of live trees decreased by 16%, from 452 to 381 trees/ac. There were shifts in the relative importance of species. Although basal area remained constant, the basal area of the oak species groups increased by 10% while the remaining species groups either decreased or did not change (Figure 6).

Individual-tree diameter growth rates varied considerably among species and by crown class for a given species (Table 6). Dominant and codominant trees grew at least twice as fast as intermediate and overtopped trees for all species. The majority of northern red oaks were dominants or codominants; whereas, the majority of sugar maples were overtopped or intermediate.

Mortality rates were lowest for the oaks and maples and highest for the understory species (Table 7). The low mortality rate for maple is consistent with its high shade tolerance and ability to persist for long periods in a suppressed state. The relatively low mortality rate for oaks can be partially attributed to their exclusive position in dominant or codominant canopy classes. However, we have observed numerous northern red oak treefalls outside the plots, suggesting that overall mortality rates for northern red oaks across the entire tract may be higher than observed in our study area. We performed a brief "pilot study" and censussed 56 gaps in a 20-ac 8-ha block encompassing the plots, and maples dominated the midstory in 82% (46) of these gaps. Due to few observations, meaningful mortality rates could not be calculated for individual crown classes. The high annual mortality rate for the understory species group reflects its characteristic high turnover rate as these trees struggle to maintain a foothold in the lower canopy strata. Spatial analysis showed that dead trees were randomly distributed throughout all distance classes (Ripley's K(t), $p > 0.05$) on both aspects.

Table 6. Tree diameter growth (inches per decade) by species and crown class at Schnabel Woods, 1982 to 1992.

	Crown class				Average of all
Species group	Overtopped	Intermediate	Codominant	Dominant	crown classes
Northern red oak	--	--	1.3	1.9	1.7
Basswood	0.0	0.5	1.2	--	1.0
Mesic	--	0.6	1.5	0.7	1.0
White oak	0.1	0.4	1.1	1.3	1.0
Elm	0.8	0.6	1.5	--	0.9
Ash	0.2	0.6	1.1	1.7	0.9
Maple	0.2	0.4	0.8	0.9	0.4
Hickory	0.0	0.2	0.8	0.0	0.4
Understory	0.2	0.7	0.0	0.0	0.2
Average	0.2	0.4	1.1	1.5	0.5

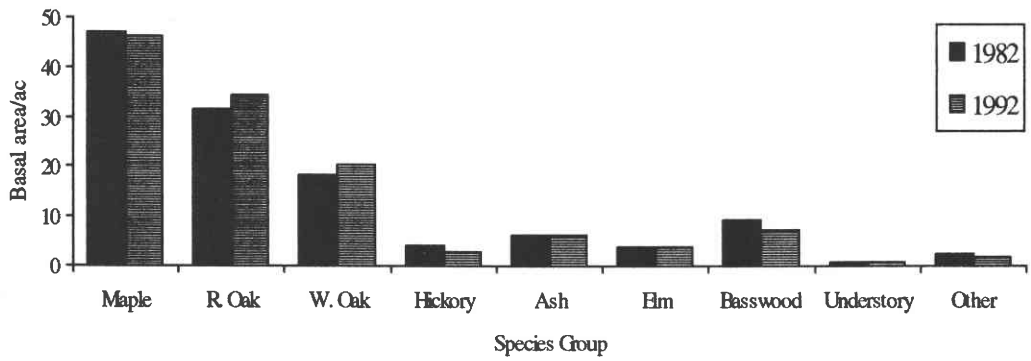

Figure 6. Basal area by species group in 1982 and 1992.

Table 7. Annual mortality rate by species group at Schnabel Woods during the 1982-1992 interval.

Species group	Annual mortality rate (%)
Understory	6.3
Basswood	6.0
Hickory	5.7
Ash	5.4
Elm	5.1
Mesic	5.0
Maple	2.3
Northern red oak	1.4
White oak	1.0
Average	2.8

Ingrowth (trees crossing the 0.8-inch dbh threshold) over the remeasurement interval consisted of 83 trees/ac and 1 ft^2/ac of basal area. Ingrowth trees were primarily sugar maple (67%), dogwood (8%), slippery elm (7%), hophornbeam (6%), and pawpaw (5%). Of these species, only maple and elm are capable of becoming part of the overstory, and elm usually succumbs to Dutch elm disease before becoming established as a dominant tree. A single chinquapin oak was the only oak ingrowth observed for the 10-year period.

Small maples, including ingrowth, were highly clumped on both aspects (Ripley's K(t), $p < 0.05$), but there was no strong spatial association (Ripley's K12(t), $p > 0.05$) with overstory trees of any species. There was a tendency for small maples (< 2.0 in dbh) to be clumped under large oaks (> 10 in dbh); however, it was only statistically significant on the northwest aspect. In contrast, small maples tended to be negatively associated with large maples (> 10 in dbh), but again, this was not significant.

DISCUSSION

The differences observed between aspects at Schnabel Woods were substantial, and greater than we would have expected prior to conducting the inventory. Although northeast- and southwest-facing slopes present large differences in environmental conditions, northwest- and southeast-facing slopes are generally expected to present more moderate or neutral environmental conditions (Hartung and Lloyd 1969). The southeast slope at Schnabel Woods had 170 more trees/ac than the northwest slope, but the basal area was 10 ft^2/ac less and the mean dbh was 1.4 in less on the southeast slope than on the northwest slope. This pattern is generally indicative of lower site quality and corresponding slower development. A further indication of site quality differences is the mass of forest floor litter which was 38% higher on the southeast slope. The lower value on the northwest slope is indicative of more mesic conditions and a higher rate of litter decomposition.

Large volumes of standing dead and down wood are often associated with old-growth forests (e.g. Parker 1989, Martin 1992). The 10-to-1 ratio of live to dead trees at Schnabel Woods is a phenomenon that has been observed in other old-growth and mature second-growth forests (Shifley and others 1995). At 563 ft^3/ac, total volume of down wood was 90 ft^3/ac greater than observed at two xeric old-growth sites in southern Missouri (Shifley and others 1995), but well below the 683 ft^3/ac (for logs > 7.9 in in diameter) observed by Mueller and Liu (1991) for a mixed mesophytic old-growth forest in Kentucky. The volume of down wood at Schnabel Woods was more than twice that observed for a second-growth oak-hickory forest in southern Missouri (Shifley and others 1995).

While the total basal area and stocking at Schnabel Woods have remained constant over the last decade, this old-growth forest is changing in many ways. The dominant trees, red oaks in particular, continue to increase in size at the expense of trees in the lower canopy strata. Approximately 3% of the total number of trees die each year, but this mortality is offset by growth on surviving trees and by annual ingrowth which amounts to approximately 2% of the number of live stems. The conspicuous absence of oaks in the ingrowth portends a substantial change in the future species composition at Schnabel Woods. This future change in composition is further indicated by the size structure of the species that are present. Red, white, and chinkapin oaks, which currently make up 44% of the basal area, are virtually nonexistent in the lower crown classes. As seen most clearly in Figure 5, red oak is the most abundant species in the dominant crown class, and white oak and sugar maple share the remainder of the class. The codominant crown class is comprised predominantly of trees in the white oak and maple species groups. Over time, we expect the red oaks, which are faster growing and shorter-lived than the white oaks, to die and be replaced by the white oaks and maples currently in the codominant crown class. The white oaks will likely be a significant component of the overstory for a century or more, but in the absence of some dramatically altered disturbance regime, the sugar maples in the intermediate and overtopped crown classes will relentlessly move their way into dominance as individual oaks die. This suggested gradual pattern of increasing maple dominance is consistent with apparent patterns of forest change inferred from early surveyor records and with patterns of change in Schnabel Woods since the earlier inventory of Kucera and McDermott (1955). The oak domination of the Schnabel Woods overstory probably reflects past disturbance regimes (e.g. burning) more than present dynamics, which are driven by fine-scale treefall gaps.

Historically, fires in mixed mesophytic forests were found primarily on ridgetops or other restricted topograhic positions (Runkle 1990). A prudent management plan for Schnabel Woods might include a careful reintroduction of fire on ridges to maintain shade intolerant species.

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METRIC CONVERSIONS

1 inch = 2.54 cm	1 ft^2 = 0.092903 m^2	1 ft^3 = 0.028317 m^3	1 U.S. ton = 0.90718 metric tons
1 foot = 0.3048 m	1 ft^2/ac = 0.2296 m^2/ha	1 ft^3/ac = 0.06997 m^3/ha	1 U.S. ton/ac = 2.241 metric tons/ha
1 mile = 1.6093 km			

STRUCTURAL AND COMPOSITIONAL DIFFERENCES BETWEEN OLD-GROWTH AND MATURE SECOND-GROWTH FORESTS IN THE MISSOURI OZARKS

Stephen R. Shifley[1], Lynn M. Roovers[2], and Brian L. Brookshire[3]

Abstract: There are currently only about 7,900 acres (3,200 ha) of remnant old-growth forest in Missouri, but public land management plans call for old-growth acreage to increase to more than 200,000 acres (81,000 ha). To develop a better quantitative understanding of the transitions that are likely as current forests mature to an old-growth state, we compared a number of characteristics measured for two old-growth sites with values for two mature second-growth sites. The stocking and the basal area of both live and dead trees were similar for the old-growth and second-growth forests. The diameter distribution (number of trees by dbh class) for all species combined had a negative exponential (reverse-J) shape that varied little from old-growth to second-growth. However, the old-growth sites consistently had more trees ≥ 17 inches (43 cm) dbh than did the second-growth sites. The absolute number of these larger trees was small; 14 per acre (35 per ha) for the old-growth sites compared to 7 per acre (17 per ha) for the mature second-growth sites. The white and red oak species groups dominated the overstories at all sites. Mean volume of down woody debris ≥ 4 inches in diameter was 476 $ft^3 \cdot ac^{-1}$ (33.3 $m^3 \cdot ha^{-1}$) on the old-growth sites *vs.* 240 $ft^3 \cdot ac^{-1}$ (16.8 $m^3 \cdot ha^{-1}$) on the second-growth comparison site.

INTRODUCTION

Old-growth forests currently occupy approximately 7,900 acres (3200 ha) in Missouri, roughly 0.05 percent of the existing forest land in the state (Shifley 1994). Most of these tracts are less than 100 acres (40 ha) and held in some form of protective status. Public land management plans in Missouri call for at least 10 percent of the publicly-owned forested acreage to be managed as old-growth. This will eventually increase the amount of old-growth forest in the state to over 200,000 acres (81,000 ha). These acreages are necessarily rough estimates because there is no precise definition of what conditions constitute an old-growth forest.

Table 1 summarizes characteristics that are typically associated with old-growth forests. While this list is comprehensive, it lacks the quantitative detail to provide a rigorous definition of the old-growth condition. Hence, the acreage of what is called old-growth forest can change when definitions are modified or when different individuals interpret the existing definitions.

Virtually every old-growth tract in Missouri has been subjected to some degree of past anthropogenic disturbance. Fire (pre- and post-European settlement) and limited livestock grazing have affected every old forest in the state. Most Missouri old-growth remnants also have had a few trees selectively harvested during the past century. Despite these disturbances, existing remnant old-growth forests provide the best available information about the likely future development of forests managed for old-growth characteristics.

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Table 1. Characteristics reported in literature to be associated with Midwestern old-growth forests[a].

Defining characteristics from Meyer (1986)	Defining characteristics from Parker (1989)	Defining characteristics from Martin (1991)
Diverse species distribution for dominant trees. Relatively high percentage of shade tolerant trees.	Tree species richness 20 to 40. Herbaceous species richness 17 to 53. Breeding bird species richness 18 to 33.	High species richness/diversity. Species richness ≥ 20 canopy trees.
Multi-layered canopy. Wide range in tree height and age.		Uneven-aged with canopy species in several size classes.
Live trees ≥ 14 inches dbh are ≥ 25% of stocking.		Several large canopy trees.
		Large, high-quality commercially important trees. (indicative of no past harvest)
Dominant trees ≥ 100 years.	Mean age of overstory 135 to 210 years. Maximum age of overstory 190 to 375 years.	Oldest trees ≥ 200 years.
	Overstory density approximately 65 to 173 trees·ac^{-1} (≥ 4 in dbh).	Overstory density approximately 100 trees·ac^{-1} (≥ 4 in dbh).
	Overstory basal area in the range of 110 - 150 ft^2·ac^{-1}.	Overstory basal area ≥ 90 ft^2·ac^{-1}.
Evidence of large tree decadence: broken and dead tops, rot, cavities. Large dead snags and large down logs. Large logs in streams and drainages.	From 8 to 18 snags·ac^{-1} (≥ 4 in dbh). Down wood 7 to 11 tons·ac^{-1}.	Logs and snags present in various sizes and stages of decay.
Variable understory density --from open to dense. Variable degree of herbaceous ground cover.	Gaps are 7-8% of forest, randomly distributed, range 0.012 - 0.09 ac in size.	Treefall gaps formed by windthrow.
	Annual mortality 0.6-0.9 %.	
		Plant and animals that prefer old- growth.
		Undisturbed soils and soil macropores.
		Little or no evidence of human disturbance.
	Volume 16,000 to 25,000 bd.ft·ac^{-1}.	

[a] cm = 2.54(inches), trees·ha^{-1} = 2.471(trees·ac^{-1}), basal area m^2·ha^{-1} = 0.2296(basal area ft^2·ac^{-1}), Mg·ha^{-1} = 2.241(tons·ac^{-1})

In this paper we compare and contrast a range of structural and compositional characteristics measured in two remnant old-growth forests with the same characteristics measured in two mature second-growth forests. This work provides new information about similarities between second-growth and old-growth forests. It also identifies some characteristics that differ sufficiently to be useful in judging where a given forest lies along the gradient from second-growth to old-growth.

STUDY SITES

The two old-growth and two second-growth sites evaluated in this study are shown in Figure 1 and described below.

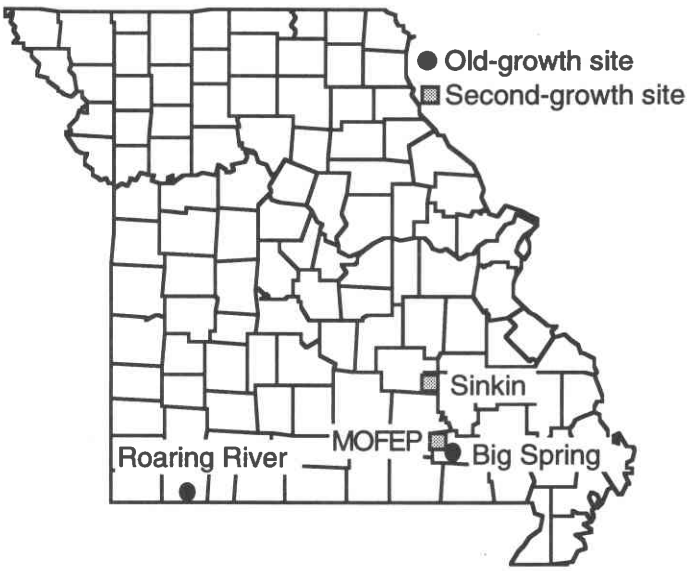


Figure 1. Location of old-growth and second-growth study sites in Missouri.

Big Spring (old-growth)

The Big Spring site is a 330-acre tract in the Ozark National Scenic Riverway in Carter County. Prior documentation for the site classified the tract as a mixture of old-growth and old second-growth oak and oak-shortleaf pine forest (Nigh and others 1992). Forest associations are xero-mesic to xeric oak-hickory and oak-pine. Our increment cores from 13 dominant and codominant trees showed ages ranging 63 to 141 years. Some trees on the tract exceed 200 years (Nigh and others 1992). There is evidence of past selective logging at a few sites along the ridges and in the bottoms. Elevation at the site ranges from 480 to 820 feet (150 to 250 m) with most slopes ranging from 20 to 40 percent. Soils are very cherty loams and silt loams that are well- to excessively-drained.

Roaring River (old-growth)

The Roaring River site is a 120-acre old-growth tract in Roaring River State Park in Barry County. A prior dendrochronological study at the site reported white oaks in the 200 to 250 year age class (Stahle and others 1985). Forest associations are principally xeric oak-hickory. With the exception of some past selective logging along the tops of the ridges that border the site, anthropogenic disturbance appears to have been limited to periodic fires that were historically common throughout the Ozarks (Ladd 1991). Elevation ranges from 1260 to 1440 feet (380 to 440 m) with slopes typically ranging from 20 to 70 percent. Most soils are cherty and moderately- to excessively-drained. Some ridgetop soils have a fragipan at approximately 14 inches (35 cm). Available water capacity and fertility are low, and rock outcrops occur at several locations on the site.

Sinkin Experimental Forest (second-growth)

The 4,100 acre (1660 ha) Sinkin Experimental Forest, located in Dent and Reynolds Counties, was used as a second-growth comparison site. Prior to establishment as an experimental forest in 1950, the tract was treated much like other forests in the area. It was extensively logged for shortleaf pine[4] between 1900 and 1920; grazing and burning were common in the following years. Since 1950 grazing and wildfire have been excluded from the Sinkin. Numerous silvicultural studies have been established on the experimental forest, but the majority of the acreage is well-stocked, second-growth, oak-hickory and oak-pine forest in the 70- to 90-year age class. Elevation ranges from 1000 to 1350 feet (300 to 410 m) with slopes typically from 10 to 35 percent. Soils are cherty loams and cherty silt-loams For this study, we excluded areas that had been harvested since 1950.

MOFEP (second-growth)

MOFEP, the Missouri Forest Ecosystem Project is a large-scale study of the impacts of silvicultural treatments on an array of ecosystem attributes (Brookshire and Hauser 1993). The MOFEP study includes 9 large, contiguous experimental tracts or compartments. We used the 3 compartments in the Peck Ranch Wildlife Area (MOFEP compartments 7, 8, and 9 in Carter County) as second-growth comparison sites. These three compartments range in size from 825 to 1240 acres (334 to 502 ha), lie in a single township, are the closest ones to the Big Spring old-growth site, and generally have been excluded from harvest for longer than the other MOFEP sites (approximately 40 years). All utilized data were collected prior to implementation of experimental treatments planned for the sites. With the exception of the exclusion of harvesting, these mature second-growth oak-hickory forests are fairly typical of Ozark forest sites. More than ninety percent of the MOFEP comparison plots were on ecological land types (ELTs) that support dry to xeric chert or limestone forest similar to the old-growth sites. Soils on the MOFEP sites are generally well- to excessively-drained and cherty. Site conditions for MOFEP compartments 7, 8, 9 are described in great detail by Brookshire and Hauser (1993).

METHODS

In 1992, thirty 0.25 ac (0.1 ha) circular inventory plots per tract were systematically established at the Big Spring and Roaring River sites. On each main plot all trees $\geq$ 4 inches (10 cm) dbh were inventoried. Trees $\geq$ 1 inch (2.5 cm) and < 4 inches (10 cm) dbh were inventoried on a concentric 0.025 ac (0.01 ha) circular subplot. Species and dbh were recorded for each tree. The number of cavities with smallest dimension $\geq$ 0.8 inch (2 cm) was also recorded for each tree at the Big Spring site and for trees on every second plot at the Roaring River site. The length and mid-point diameter of each piece of down wood with minimum diameter $\geq$ 4 inches (10 cm) was measured on the 0.25 ac (0.1 ha) main plot. Each piece of down wood was classified by decay stage using a system described by Maser and others (1979) for western conifers (Table 2). Volume of each down log was computed as the volume of a cylinder of known length and midpoint diameter. The ground area covered by each down log was computed as the product of its length and diameter.

In 1992-93, nearly identical protocols were used to sample 96 plots systematically distributed across the Sinkin Experimental Forest. The only deviation was that cavities were not measured. Of the 96 plots on the Sinkin, 73 were utilized in this comparison. The remaining 23 were excluded because they fell in areas that had been subjected to harvest or other disturbance since the experimental forest was established in 1950.

[4]Scientific names for all species are given in Table 4.

Table 2. Decomposition classes used to classify down logs. From Maser and others (1979).

Decomposition Class	Defining characteristics
Class 1	Round log, bark intact, small twigs present, log elevated on support points
Class 2	Round log, bark intact, twigs absent, log elevated on support points
Class 3	Round log, traces of bark, twigs absent, log sagging near ground
Class 4	Round to oval log, bark absent, twigs absent, log entirely on ground
Class 5	Oval log, bark absent, twigs absent, log entirely on ground

The MOFEP sites were sampled with a total of 210 plots during 1990-91. Plot location was random subject to the constraint that at least one plot had to fall in each identified forest stand. The 0.5 ac (0.2 ha) circular main plots were used to sample trees $\geq$ 4.5 inches (11.4 cm) dbh. Trees $\geq$ 1.5 inches (3.8 cm) and < 4.5 inches (11.4 cm) dbh were sampled on four circular 0.05 ac (0.02 ha) subplots. The MOFEP samples included species and dbh for each tree. Percent of the ground covered by down wood $\geq$ 2 inches (5 cm) in diameter on the MOFEP sites was sampled along four 56.5-foot (17.3 m) transects originating at plot center and oriented along the cardinal directions. Percent cover was computed as the percent of the transect length actually covered by down logs larger than the 2-inch (5 cm) size threshold.

Per acre values for each characteristic were computed by plot and summarized to obtain means and variances for each tract. Stocking percent was computed from the tree area ratio equations of Gingrich (1967), Rogers (1983), and Stout and others (1987). Importance values for each tract (all plots combined) were computed as [0.5(relative number of trees + relative basal area)]. Two types of statistical comparisons were made among characteristics observed at the study sites. The first test was a one-way analysis of variance by site for the 4 sites with all plots per site used in the comparison (n = 343). The null hypothesis is that the characteristic of interest (e.g. basal area, number of trees, etc.) is equal on all sites. Although this is an appropriate test for differences among the specific sites, the procedure introduces the problem of pseudoreplication (Hurlbert 1984) if multiple observations per tract are used to make inferences concerning the more general comparison of old-growth *vs.* second-growth sites. A more appropriate and more conservative comparison of old-growth and second-growth forest characteristics can be made by using a t-test to evaluate the null hypothesis that the means observed for the two old-growth sites were equal to those for the two second-growth sites (n=4). Whenever possible, this second statistical test was also performed and reported.

RESULTS

Structural Characteristics

The most notable trend that emerges from a comparison of composite stand characteristics for the old-growth and mature second-growth forests is the similarity across all sites. (Table 3). Although the MOFEP site had substantially fewer trees per acre than the other sites, the MOFEP sampling scheme excluded live trees between 1 and 1.5 inches (2.5 and 3.8 cm) dbh. Observations on the Sinkin indicate that live trees between 1 and 1.5 inches (2.5 and 3.8 cm) dbh would add approximately 150 trees•ac^{-1} (350 trees•ha^{-1}) and 1.3 ft^2•ac^{-1} (0.3 m^2•ha^{-1}) of basal area to the MOFEP means. This addition would bring the mean number of trees•ac^{-1} on the MOFEP sites in line with the observed values for the other sites, although the mean basal area would still be somewhat lower.

Table 3. Comparison of composite stand characteristics for old-growth (Big Spring and Roaring River) and second-growth (Sinkin and MOFEP) study areas. Values in parentheses are standard deviations for values by plot at that site. Values are for trees ≥ 1 inch (2.5 cm) dbh except as noted[a].

Site	Sample size	Live trees number•ac^{-1}	Live basal area ft^2•ac^{-1}	Stocking percent	Dead trees number•ac^{-1}	Dead basal area ft^2•ac^{-1}
Big Spring	30	582 (209)	104 (12)	88 (12)	68 (44)	8.3 (6.3)
Roaring River	30	623 (191)	108 (23)	96 (18)	77 (57)	12.5 (8.4)
Sinkin	73	675 (297)	102 (21)	91 (17)	90 (86)	9.1 (6.9)
MOFEP[b]	210	439 (161)	90 (13)	82 (9)	15 (11)	7.4 (5.8)

[a] Trees•ha^{-1} =2.471(trees•ac^{-1}) ; basal area m^2•ha^{-1}= 0.2296(basal area ft^2•ac^{-1}).

[b] MOFEP data includes only live trees ≥ 1.5 inches dbh and only standing dead trees ≥ 4.5 inches dbh.

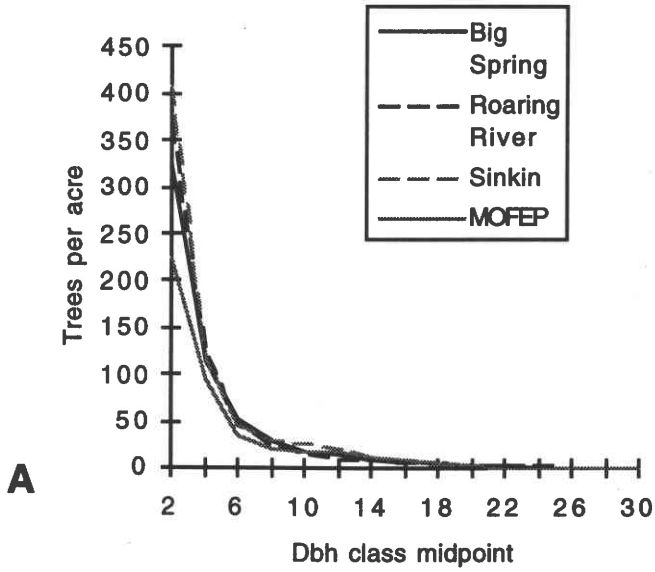
One-way ANOVA using all plots to test for differences among the 4 sites showed significant differences in the mean number ($p < 0.001$, 339 d.f.), basal area ($p < 0.001$, 339 d.f.), and stocking percent ($p < 0.001$, 339 d.f.) for live trees per acre. However, the differences did not follow patterns that could be used to readily distinguish the old-growth sites from the second growth sites. MOFEP means were uniformly smaller than the other three sites which were statistically indistinguishable from one another (Tukey-Kramer HSD, $\alpha = 0.05$) for all three dependent variables. Adjusting for trees between 1 and 1.5 inches (2.5 and 3.8 cm) dbh that were excluded from the MOFEP sampling, eliminates the significant difference among sites in the number of live trees per acre.

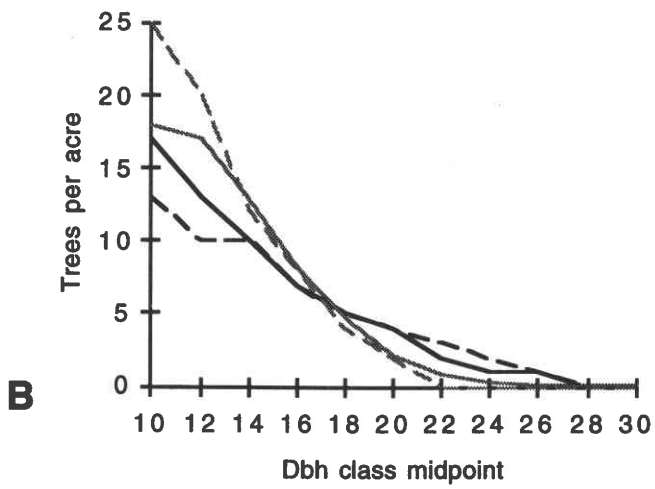
The number of standing dead trees per acre was also significantly different among sites (one-way ANOVA by site, $p < 0.001$, 339 d.f.) with the MOFEP mean smaller than the other three sites (Tukey-Kramer HSD, $\alpha = 0.05$). Again this was at least partially due to sampling procedural differences at the MOFEP site. Basal area of standing dead trees followed a somewhat different pattern. Means were significantly different among sites (one-way ANOVA by site, $p < 0.001$, 339 d.f.), but in this case the Big Spring and MOFEP means were significantly larger than the Roaring River and Sinkin sites (Tukey-Kramer HSD, $\alpha = 0.05$).

A comparison of the means of the two old-growth sites with that of the two second-growth sites showed no significant differences in the number, basal area, or stocking of live trees, nor for the number or basal area of standing dead trees (2 d.f. with p-values of 0.74, 0.25, 0.46, 0.65, and 0.44, respectively). Nor were the mean differences of sufficient magnitude to generally be considered of practical importance, statistical tests notwithstanding.

At first glance, a comparison of the number of live trees by diameter class also shows little difference among the old-growth and second-growth sites (Figure 2A). At all sites the diameter distribution (based on all tree species) had a negative exponential shape commonly associated with uneven-aged forests. The greatest absolute difference in the diameter distributions occurred for trees in the 2-inch (5 cm) dbh class. The MOFEP site had fewer trees of this size than the other three sites, due primarily to the fact that trees < 1.5 inches (3.8 cm) were not included in the MOFEP sample. The greatest *relative* difference in the number of trees per acre, however, occurred for trees larger than 8 inches (20 cm) dbh. That portion of the diameter distribution is redrawn at higher resolution to highlight those differences (Figure 2B).

The second growth sites had 1.5 to 2 times as many trees between 9 and 17 inches (23 and 43 cm) as the old-growth sites, but this relationship shifts for larger diameter classes. On average the old-growth sites had 14 trees per acre (35 trees per ha) larger than 17 inches (43 cm) dbh compared to 7 trees per acre (17 trees per ha) for the second-growth sites. This difference may appear rather minor, but in terms of basal area (rather than the number of trees) these large trees have much greater impact (Figure 2C). A graphic based on tree volume or biomass would further accentuate the influence of the largest diameter trees.





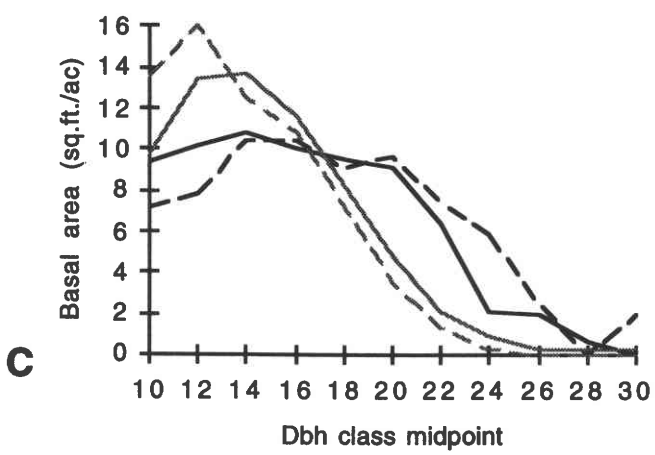


Figure 2. Three views of the diameter distribution (all species) for old-growth and second growth sites showing (A) number of trees ≥ 1 inch dbh (≥ 1.5 inches for MOFEP), (B) number of trees ≥ 9 inches dbh, and (C) basal area of trees ≥ 9 inches dbh.

The functional importance of these relatively few large trees must not be overlooked. One feature that inevitably stands out when walking through an old forest is the large trees, which people tend to notice and recall disproportionate to their low frequency. If humans tend to ascribe high relative importance to large individual trees, other organisms may also do so, either individually or collectively. The largest trees provide the highest and strongest perches. They provide the greatest surface area and volume and the largest cavities. Larger trees are also more likely to have cavities that can be used as nesting sites by wildlife. Observations of cavities by tree size on the Big Spring and Roaring River tracts showed a 15 percent probability of a 10-inch (25 cm) dbh tree having a cavity; that probability increased linearly to over 60 percent for a 25-inch (64-cm) dbh tree.

For all sites the diameter distribution of standing dead trees followed the same general shape as the diameter distribution of live trees, at approximately 10 percent the frequency of live trees. This result was consistent across all sites and might serve as a rule of thumb for the amount and size distribution of dead-standing wood in the maturing oak forests.

Species Composition

The total number of tree species (tree species richness) was greatest on the Sinkin and MOFEP sites. The number of tree species observed by site was 43, 40, 36, and 33 for Sinkin, MOFEP, Big Spring and Roaring River, respectively. This difference of seven in the tree species richness between the two old-growth and two second-growth sites was near the margin of statistical significance (t-test, $p = 0.08$, 2 d.f.) However, the higher species richness values on the second-growth sites were consistent with expectations given the larger sample sizes on those sites. All other factors being equal, the greatest number of species would be expected on these two tracts with the most samples. Because each site had only a single observation for species richness, statistical differences in species richness among all four sites could not be analyzed using one-way ANOVA.

It is informative to compare importance values for species by site. In Table 4, importance values and their ranks are given for all woody species that attained a diameter of at least one inch (2.5 cm) dbh (1.5 inches or 3.8 cm dbh for the MOFEP site). White oak, black oak, scarlet oak and dogwood have high importance values at all sites. Shortleaf pine has a high value at sites other than Roaring River where it did not occur. The high importance value for dogwood is strictly a function of the large number of dogwood stems that are common in the lower canopy strata at these sites. Species importance rank order at a given site does not change if some species or groups of species (e.g. species generally relegated to the understory) are dropped from consideration. Also, the values in Table 4 can be used to compute the importance values for subsets of species.

It is interesting to note that both maple and pawpaw were minor components in these relatively dry old-growth forests. This is in sharp contrast to values reported for more mesic old-growth sites (e.g. Kucera and McDermott 1955, Shotola and others 1992, Wuenscher 1967) where sugar maple is typically more than 20 percent of the total basal area and papaw may constitute more than 5 percent of the woody stems in the understory.

Some patterns in forest composition are more easily discerned by examining species groupings by size class. Figure 3 illustrates how the relative importance of various species groups changes with increasing diameter class. Although relatively rare in the smaller diameter classes, the white and red oak groups dominate the overstories at all sites. On the two old-growth sites, the white oaks have greater relative importance in the largest diameter classes than do the red oaks.

Although the composite diameter distribution for all species combined has a negative-exponential shape (as shown in Figure 1), individual species do not necessarily follow that form. For white oak, red oak, and shortleaf pine groups which dominate the overstory at these sites, the composite number of trees by diameter class has a unimodal distribution on all sites (Figure 4). However, on the old-growth sites the peak of the diameter distribution is shifted to the right and the number of trees in the smaller size classes is substantially fewer than observed for the second-growth

Table 4. Importance values and their ranks for woody plants on old-growth (Big Spring and Roaring River) and second-growth (Sinkin and MOFEP) sites. Importance values are computed as {(relative number of trees + relative basal area)/2} for all live trees sampled. Species are arranged in decreasing rank order for the Big Spring site. Nomenclature follows Little (1953).

		Big Spring		Roaring River		Sinkin		MOFEP	
Common name	Scientific name	IV	Rank	IV	Rank	IV	Rank	IV	Rank
Flowering dogwood	*Cornus florida*	24.0	1	17.6	2	16.7	2	8.8	4
White oak	*Quercus alba*	22.5	2	20.8	1	22.7	1	17.9	3
Black oak	*Quercus velutina*	6.9	3	15.3	3	15.0	3	18.3	2
Scarlet oak	*Quercus coccinea*	6.5	4	0.4	24	3.9	6	18.8	1
Shortleaf pine	*Pinus echinata*	5.5	5	<0.1	--	12.9	4	5.9	6
Mockernut hickory	*Carya tomentosa*	5.3	6	10.8	4	2.3	8	4.5	8
Black hickory	*Carya texana*	4.4	7	2.2	9	1.5	12	5.1	7
Post oak	*Quercus stellata*	4.2	8	1.5	11	1.4	14	8.4	5
Blackgum	*Nyssa sylvatica*	3.8	9	2.3	8	6.4	5	3.0	9
S. red oak	*Quercus falcata*	2.9	10	<0.1	--	<0.1	--	<0.1	--
N. red oak	*Quercus rubra*	1.6	11	5.9	6	1.7	10	<0.1	--
Red maple	*Acer rubrum*	1.6	12	3.0	7	2.1	9	0.8	13
Chinkapin oak	*Quercus muehlenbergii*	1.2	13	1.1	14	0.8	16	0.4	14
Winged elm	*Ulmus alata*	0.9	14	0.7	20	0.1	30	0.2	21
Grape	*Vitis* spp.	0.8	15	1.9	10	<0.1	--	0.2	18
Slippery elm	*Ulmus rubra*	0.8	16	0.4	23	3.5	7	0.2	19
Serviceberry	*Amelanchier* spp.	0.7	17	7.4	5	0.1	32	0.1	23
Bitternut hickory	*Carya cordiformus*	0.7	18	<0.1	35	0.7	17	0.1	26
Pignut hickory	*Carya glabra*	0.7	19	0.3	26	1.5	11	2.0	11
Sassafras	*Sassafrass albidum*	0.6	20	1.0	15	1.2	15	2.5	10
Buckthorn	*Rhamnus* spp.	0.6	21	0.7	19	0.1	29	0.3	15
Ironwood	*Ostrya virginiana*	0.4	22	0.7	18	1.4	13	<0.1	36
Green ash	*Fraxinus pennsylvanica*	0.4	23	<0.1	--	0.2	24	<0.1	31
Black walnut	*Juglans nigra*	0.4	24	0.1	34	0.6	20	0.2	22
Blackjack oak	*Quercus marilandica*	0.4	25	0.1	28	<0.1	34	1.3	12
White ash	*Fraxinus americana*	0.4	26	1.3	13	0.2	27	0.1	24
Eastern redbud	*Cercis canadensis*	0.3	27	0.7	17	0.4	22	0.2	17
Blueberry	*Vaccinium* spp.	0.3	28	<0.1	--	<0.1	--	<0.1	37
Persimmon	*Diospyros virginiana*	0.3	29	<0.1	--	<0.1	41	0.2	16
Bluebeech	*Carpinus caroliniana*	0.3	30	<0.1	--	0.7	18	<0.1	--
Paw paw	*Asimina triloba*	0.2	31	<0.1	--	<0.1	--	<0.1	40
N. pin oak	*Quercus palustris*	0.1	32	<0.1	--	<0.1	--	<0.1	--
Basswood	*Tilia americana*	0.1	33	0.2	27	<0.1	37	<0.1	--
Viburnum	*Viburnum* spp.	0.1	34	<0.1	--	<0.1	--	<0.1	35
Gumbemelia	*Bumelia lanuginosa*	0.1	35	<0.1	--	<0.1	39	<0.1	32
Black cherry	*Prunus serotina*	<0.1	36	<0.1	36	0.2	28	0.1	27
Wild plum	*Prunus americana*	<0.1	37	0.9	16	<0.1	--	0.1	28
American elm	*Ulmus americana*	<0.1	38	0.1	33	<0.1	--	0.1	30
Sugar maple	*Acer saccharum*	<0.1	--	1.3	12	<0.1	38	<0.1	42
Spicebush	*Lindera benzoin*	<0.1	--	0.5	21	<0.1	--	<0.1	--
Shagbark hickory	*Carya ovata*	<0.1	--	0.5	22	<0.1	--	<0.1	43

Table 4 (continued)

Eastern redcedar	*Juniperus virginiana*	<0.1	--	0.3	25	0.1	31	0.2	20
Red mulberry	*Morus rubra*	<0.1	--	0.1	29	0.2	26	0.1	29
Ozark chinkapin	*Castanea ozarkensis*	<0.1	--	0.1	30	<0.1	--	<0.1	--
Greenbrier	*Smilax* spp.	<0.1	--	0.1	32	<0.1	--	<0.1	--
Sycamore	*Platanus occidentalis*	<0.1	--	<0.1	--	0.7	19	<0.1	41
Hackberry	*Celtis occidentalis*	<0.1	--	<0.1	--	0.4	21	<0.1	33
Butternut	*Juglans cinerea*	<0.1	--	<0.1	--	0.3	23	<0.1	--
Hawthorn	*Crataegus* spp.	<0.1	--	<0.1	--	0.2	25	0.1	25
Shellbark hickory	*Carya laciniosa*	<0.1	--	<0.1	--	0.1	33	<0.1	--
Honeylocust	*Gleditsia triacanthos*	<0.1	--	<0.1	--	<0.1	35	<0.1	38
Chokecherry	*Prunus virginiana*	<0.1	--	<0.1	--	<0.1	36	<0.1	--
Blue ash	*Fraxinus quadrangulata*	<0.1	--	<0.1	--	<0.1	42	<0.1	--
Swamp chest. oak	*Quercus michauxii*	<0.1	--	<0.1	--	<0.1	43	<0.1	--
Shumard oak	*Quercus shumardii*	<0.1	--	<0.1	--	<0.1	--	<0.1	34
Sumac	*Rhus* spp.	<0.1	--	<0.1	--	<0.1	--	<0.1	39
Poison ivy	*Rhus radicans*	<0.1	--	<0.1	--	<0.1	--	<0.1	44

tracts. The mean diameters of the combined white oak, red oak, and shortleaf pine species groups was 7.7 and 8.8 inches (19.6 and 22.4 cm) for the Big Spring and Roaring River sites, respectively, while those for the Sinkin and MOFEP sites were 6.3 and 6.0 inches (16.0 and 15.2 cm), respectively. The unimodal shape of the diameter distributions in Figure 3 are largely due to the white oak group. When considered separately, the diameter distribution for the red oak group has a unimodal shape on the Big Spring and MOFEP sites. The diameter distribution of shortleaf pine by itself did not have a distinctly unimodal shape at any of the four sites.

Down Woody Debris

Mean volume of down woody debris at least 4 inches (10 cm) in diameter on the Big Spring and Roaring River sites was about twice the volume observed at Sinkin (Table 5). There were no distinguishable patterns within decomposition classes, but the classes had been developed for western conifers (Maser and others 1979). Using these class definitions, the majority of the volume consistently fell into classes 3 and 4.

Within each tract the variability in the volume of down wood among plots was high. The coefficient of variation for the total volume of down wood by plot was in the range of 70 to 75 percent. Nevertheless, analysis of variance for the mean volume of down wood per acre for the Big Spring, Roaring River, and Sinkin sites (one-way ANOVA based on all plots per site) showed significant differences among sites ($p < 0.001$, 130 d.f.). Pairwise comparison of means for these three sites (Tukey-Kramer HSD, $\alpha = 0.05$) showed no significant difference between Big Spring and Roaring River, but values for those two sites were significantly larger than for the Sinkin. Because of procedural differences in sampling, compatible estimates of down wood volume were not available for the MOFEP site. The mean volume of down wood at least 2 inches (5 cm) in diameter on the MOFEP site is in the approximate range of 200 to 300 $ft^3 \cdot ac^{-1}$ (14 to 21 $m^3 \cdot ha^{-1}$), depending on the set of assumptions used to estimate the volume of down logs. But even at the high end, these estimates are markedly lower than observed at Big Spring and Roaring River. Because a comparable estimate of down wood volume was not available for the MOFEP sites, it was not possible to use a t-test to evaluate differences in down wood volume between the two old-growth and the two second-growth sites.

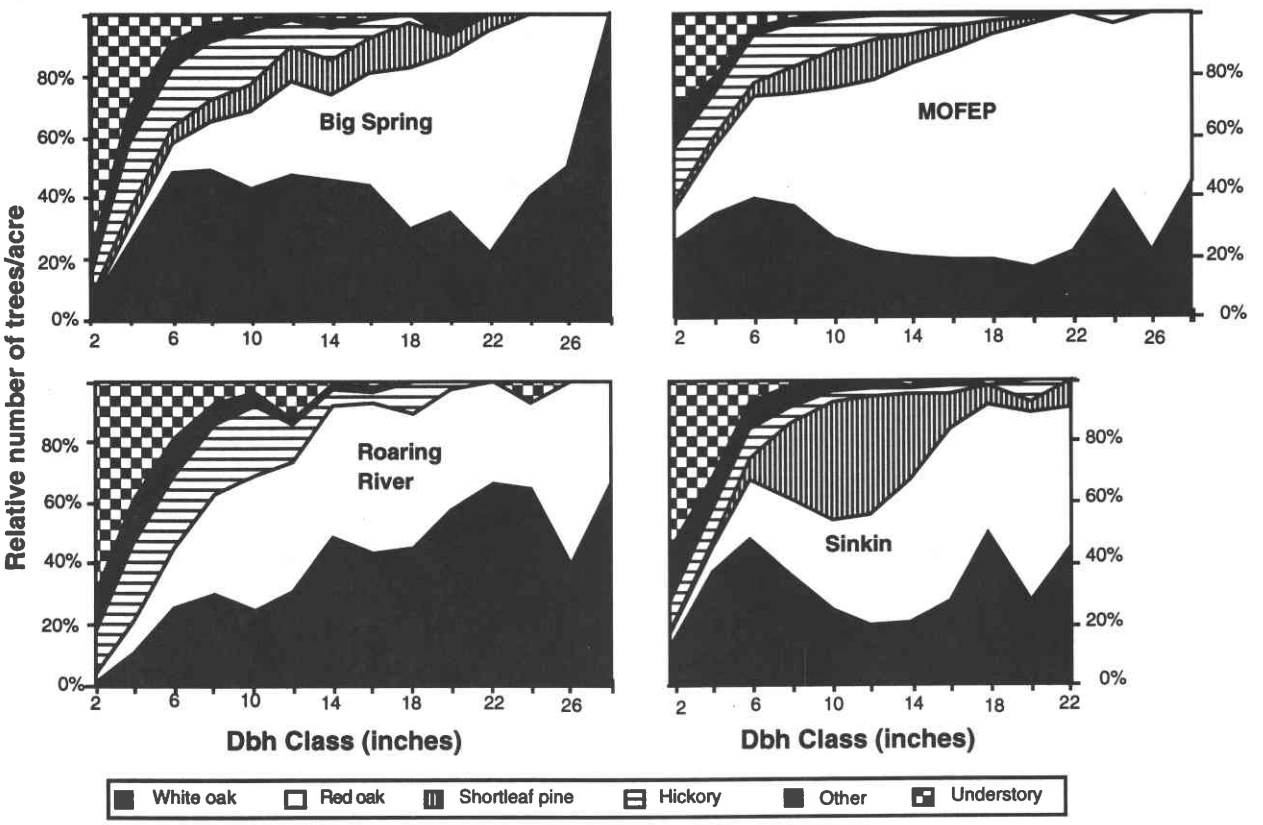


Figure 3. Relative number of trees per acre by 2-inch dbh class midpoint and major species groups. The white oak group includes white, post, chestnut, and chinkapin oak. The red oak group includes northern red, southern red, scarlet, black, and blackjack oaks. Hickory includes all *Carya* spp. Understory species include dogwood, blackgum, ironwood, bluebeech, serviceberry, pawpaw, buckthorn, redbud, vines, and shrubs.

The mean percent cover for the two old-growth and two second-growth sites (Table 5) was 1.56 and 1.29 percent, respectively. This difference was not statistically significant ($p = 0.33$ for t-test with 2 d.f.).

Table 5. Mean volume of down woody debris and percent of ground covered by down wood[a]. Values include pieces ≥ 4 inches (10 cm) diameter for Big Spring, Roaring River and Sinkin; ≥ 2 inches (5 cm) diameter for MOFEP. Class definitions are from Maser and others (1979) and summarized in Table 2.

	(newly fallen) ⟶				(decomposed)	Total	Percent of ground
Site	Class 1 $ft^3 \cdot ac^{-1}$	Class 2 $ft^3 \cdot ac^{-1}$	Class 3 $ft^3 \cdot ac^{-1}$	Class 4 $ft^3 \cdot ac^{-1}$	Class 5 $ft^3 \cdot ac^{-1}$	all classes $ft^3 \cdot ac^{-1}$	covered by down wood
Big Spring	15.6	28.1	192.6	180.1	40.7	457.1	1.50
Roaring River	3.1	74.9	265.1	104.1	47.6	494.8	1.61
Sinkin	16.0	45.5	104.9	57.1	16.7	240.2	1.09
MOFEP	--	--	--	--	--	--	1.49

[a]Volume $m^3 \cdot ha^{-1} = 0.06997$(volume $ft^3 \cdot ac^{-1}$)

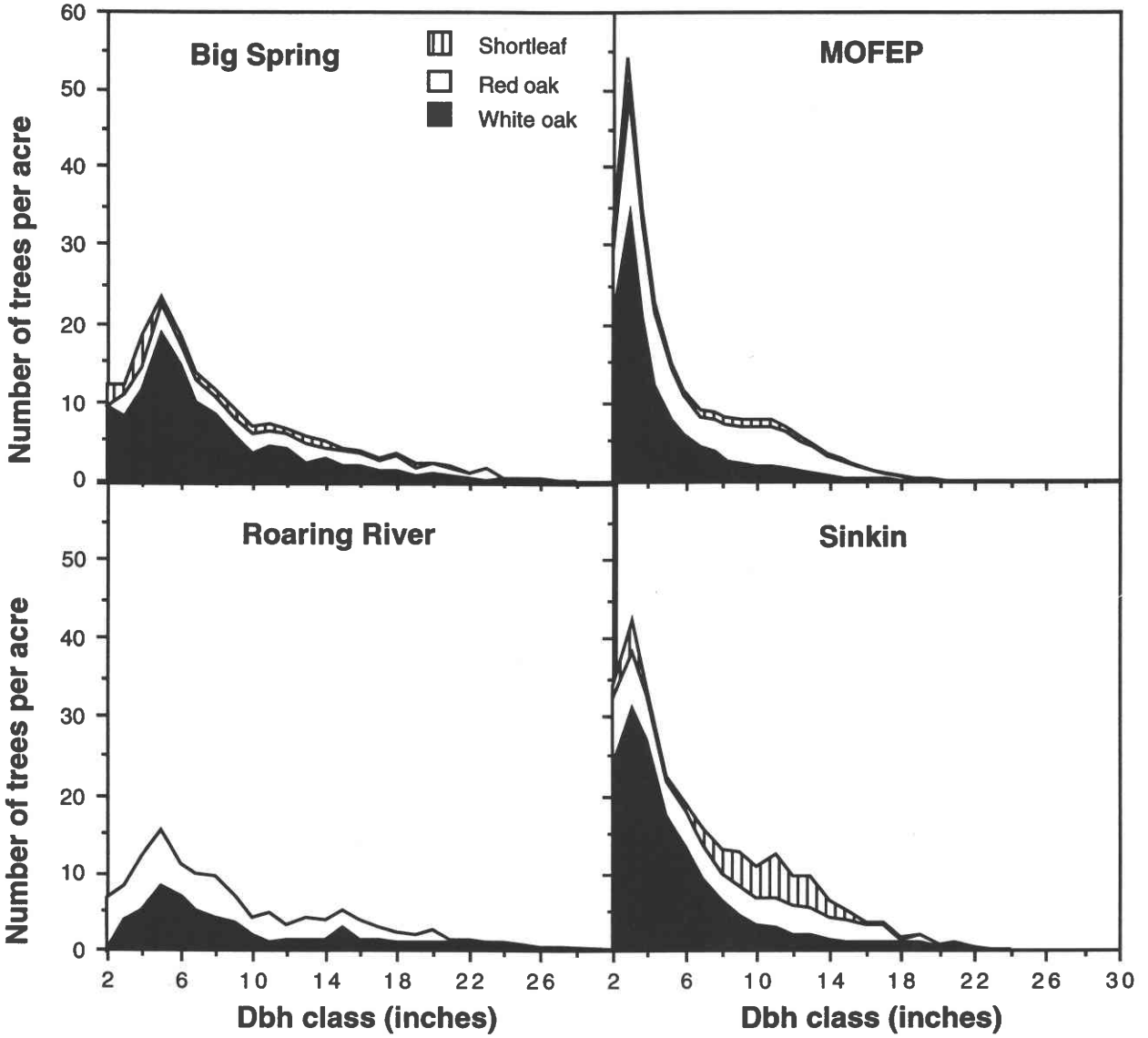


Figure 4. Number of trees per acre by major species groups. The white oak group includes white, post, chestnut, and chinkapin oaks. The red oak group includes northern red, southern red, scarlet, black, blackjack, and pin oaks.

CONCLUSIONS

Composite stand characteristics such as basal area and stocking were remarkably similar for the old-growth and second-growth stands compared. Likewise, the diameter distributions for all species combined had a negative exponential shape that was similar for the old-growth and second-growth tracts.

However, the old-growth tracts consistently had more trees per acre ≥ 17 inches (43 cm) dbh than did the second-growth sites. The absolute number of these large trees was small at all sites--an average of 14 per acre (35 per ha) for old-growth sites and 7 per acre (17 per ha) for second-growth sites. The overstories at all four sites were dominated by the red and white oak groups, and shortleaf pine was a substantial overstory component for all sites except Roaring

River. Dogwood was the dominant understory species at all sites. When considered jointly, the white oaks, red oaks, and shortleaf pine had unimodal diameter distributions at all sites, although the old-growth sites had fewer and larger trees of these species. Down wood on the old-growth sites was between 450 and 500 $ft^3 \cdot ac^{-1}$ (31 and 35 $m^3 \cdot ha^{-1}$), about double that observed for the Sinkin Experimental Forest.

The data we examined indicate that the composite density of second-growth Ozark oak forests may change relatively little as they mature to an old-growth state. Some of the most readily observed changes should be shifts in the number and basal area of trees larger than 17 inches (43 cm) dbh, corresponding changes in the diameter distribution of the dominant overstory species, and large increases in the volume of down wood.

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DEVELOPMENT OF A CENTRAL HARDWOOD STAND FOLLOWING WHOLE-TREE CLEARCUTTING IN CONNECTICUT

C. Wayne Martin[1]

Abstract: Little information is available concerning the initial stages of forest regeneration following intensive harvesting of central hardwood stands in the northeastern part of the range. Establishment of commercial species, density, and rate of biomass accumulation of the regeneration are of major concern to both foresters and landowners contemplating a harvest. To help provide this information, a 6-ha forest stand dominated by the central hardwood forest type of Connecticut was mechanically whole-tree clearcut in 1981. Intensive sampling of the regeneration including herbs, shrubs, tree seedlings, and stump sprouts was conducted the first, third, fifth, and tenth years after cutting. This paper will discuss the development of the stand over the first ten years following clearcutting. Prior to harvesting, the stand contained 1,163 stems ha^{-1} ≥2.0 cm d.b.h.; at year 10 there were 9,500. At year 10, the trees and shrubs ≥2.0 cm d.b.h. had accumulated 20% of the preharvest levels of biomass, 52% of nitrogen, and 21% of calcium. Based on data from the first 10 years after clearcutting, the future forest should be similar to the preharvest forest with red maples, oaks, and birches being the most numerous trees.

INTRODUCTION

The central hardwood forest type is extensive in southern New England. As of 1985, more than 2.1 million acres of Massachusetts, Rhode Island, and Connecticut were classified as the oak-hickory forest type (Brooks et al. 1993). In Connecticut alone, there were 966,000 acres of the oak-hickory forest in 1985 (Dickson and McAfee 1988). Eighty four percent of the oak-hickory type in Connecticut is in non-industrial private ownership. In 1984, 722,000 cubic meters of hardwood wood products were produced in Connecticut including sawlogs, roundwood, and fuelwood (Dickson and McAfee 1988).

Landowners, foresters, loggers, and the general public are concerned about the future productivity and biodiversity of these forest lands. For long-term planning, landowners and foresters need information on the characteristics of the early stages of regeneration, the development of stands, and the expectations for the future mature stands. Little information is available concerning the regeneration and early successional stages of forest development following logging in the northeastern part of the central hardwood range. The first 5 to 10 years after harvest are critical in the development of the new stand and are not well understood (Hornbeck et al. 1990).

The effects of calcium depletion on the long-term productivity of these forests is another reason for studying the regeneration of these stands. Hornbeck and others (1990) found that 10% of the total calcium pool was removed from a central hardwood stand in Connecticut in wood products removed from the site and in soil water leached to streams following mechanical whole-tree clearcutting. They also predicted that calcium uptake by 3 year-old regeneration would exceed the extractable calcium in the soils of the site. They studied the calcium dynamics of 3 forest types in New England and concluded that calcium is potentially a limiting plant nutrient in the forests of Connecticut. Data on calcium uptake by regeneration through year 10 would add new insight to this hypothesis.

During the energy crisis of the late 1970s and early 1980s, the forests of southern New England were viewed as alternative sources of fuel. Fuelwood cutting on a commercial scale was proposed, much of it on state forest lands. To

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test the potential impacts of fuelwood cutting on state forest lands, the State Forester of Connecticut, requested a study of the effects of mechanical whole-tree clearcutting on site productivity, regeneration, and development of future stands. The purpose of this paper is to report on the development of a southern New England stand over the first 10 years following clearcutting.

METHODS

The Study Site

The study site is a 6 ha forested stand located in the Cockaponset State Forest in Chester, CT. Prior to harvesting, the watershed supported an 80- to 110-year old oak-birch-maple forest typical of Connecticut and southern New England. Oaks (*Quercus Prinus* L., *Q. rubra* L., *Q. alba* L., and *Q. velutina* Lam.) accounted for 47% of the basal area; birches (*Betula lenta* L., and *B. alleghaniensis* Britt.) accounted for 27%; and maples (*Acer rubrum* L. and *A. saccharum* Marsh.) accounted for 13% of the basal area. The remainder included miscellaneous hardwoods. The total basal area of all stems larger than 2 cm d.b.h. was 23 m^2 ha^{-1}, with a density of 1,163 stems ha^{-1} (Tritton et al. 1982).

The soils of the upper slopes of the study site are of the Hollis-Chatfield-Rock association which are loamy, mixed, mesic Lithic Dystrochrepts. These soils consist of shallow, somewhat excessively drained soils with occasional outcrops of exposed bedrock. The bedrock is of the upper Middletown formation, an assemblage of gneisses and schists with inclusions of sillimanite quartz and pegmatite. The soils of the lower slopes are deep, well-drained, coarse, loamy, mixed, mesic, Typic Dystrochrepts of the Chatfield-Canton association. The soils of the valley floor are poorly drained, acidic, coarse-loamy, mixed, mesic Aeric Haplaquepts of the Leicester series (USDA Soil Conserv. Serv. 1981).

In the winter of 1981-82, the entire watershed was whole-tree clearcut. All living and dead trees greater than 5 cm d.b.h. were cut. Trees less than 30 cm stump diameter were cut with a rubber-tired feller buncher. Larger trees were felled with chain saws. Whole trees of all diameters were skidded to the landing outside the watershed by rubber-tired, articulated skidders. Ninety one percent of the aboveground biomass was removed from the site by the harvesting operation (Tritton et al. 1987). Harvesting was completed in March of 1982 and was confined to the dormant season.

Regeneration Surveys

Prior to harvesting, a 25 X 25 m grid was surveyed on the watershed with the corners permanently marked. From this grid, plots were selected at random for the regeneration surveys. These plots were also permanently marked. Three sizes of plots were selected for vegetation sampling: 1 X 1 m plots for herbs, shrubs, and tree seedlings <130 cm tall; 1 X 25 m plots for saplings 0.1 to 9.9 cm d.b.h.; and 25 x 25 m plots for trees ≥10 cm d.b.h. and for following the fate of stump sprouts. Prior to cutting, all trees ≥10 cm d.b.h. were measured and their location mapped.

The first survey was conducted in August 1982, the first year after harvest. Since all vegetation was <130 cm tall, 50 1 X 1 m plots were sampled. Sprouts from the stumps of the mapped trees were also measured. The second survey was conducted in August 1984, the third year after cutting. The third survey was conducted in August 1986, the fifth year after harvest. On both of these occasions, twenty five of the 1 X 1 m plots were remeasured. In addition, 16 of the 1 X 25 m plots were measured.

The fourth survey was conducted in August 1991, the 10th year after cutting. The twenty five 1 X 1 m plots were remeasured. Twenty five paired plots were harvested for biomass and nutrient analyses. The sixteen 1 x 25 m plots for saplings were remeasured, but because saplings in this size range represented a majority of the vegetation on the area, 20 more plots were added, at random, for a total of 36 plots. Because of the density of the foliage, the stump sprouts on four 25 X 25 m plots were remeasured in November of 1986 and 1991.

Biomass

Three individuals of each species were clipped, on separate plots, for biomass and nutrient analyses the first year after cutting. These samples were dried at 65°C to constant weight. The average of the 3 samples was multiplied by the number of stems of that species per hectare. Biomass of the vegetation <130 cm tall the 3rd and 5th year after cutting was estimated by clipping twenty one 1 X 1 m plots selected at random using ranked-set sampling techniques (McIntyre 1952). Biomass of the vegetation <130 cm tall in the 10th year was estimated by taking a simple average of the oven-dry weights of twenty five plots paired with the inventory plots. The first year after cutting there were no stems large enough to have a d.b.h. During the 3rd and 5th year inventories, 3 stems of each species in each 1-cm d.b.h. class were clipped in various areas of the watershed. Each stem was weighed in the field, subsampled, and oven dried at 65°C. The average oven-dried weight of each species and size class was then multiplied by the estimated number of stems of that species and size per hectare. The 10th year after cutting, several individuals over a range of d.b.h. were sampled and dried by species. Then allometric equations were developed for each species for predicting oven-dry weight from d.b.h.

Nutrient Analyses

Vegetation samples collected for biomass estimates were dried to a constant weight at 65°C, and then analyzed chemically. Subsamples were digested in a Technicon BD-20[2] block digestor. Total Kjeldahl nitrogen was measured on the digested samples with colorimetric analyses performed on the Technicon Autoanalyzer. Calcium concentrations of the digested samples were determined by flame atomic absorption spectroscopy (Franson 1975).

RESULTS

Density

The first year after cutting, there were 49 genera of plants inventoried on the clearcut watershed. These genera included 14 tree species in 11 genera, 12 species of shrubs in 10 genera, and 28 genera of herbs. By the 10th year after cutting, the total number of genera had dropped to 47. However, trees had increased to 22 species in 13 genera. Shrubs had increased to 18 species in 15 genera, and herbs had dropped to 19 genera.

By August of the first year after cutting, there were about 788,000 stems ha^{-1} of all types of plants on the watershed with trees, shrubs, and herbs occurring in similar amounts (Fig. 1). By the 5th year after cutting, there were more than 1,000,000 stems ha^{-1} on the watershed with about 60% of them herbs and only 10% of them trees. By year 10, the total number of stems had dropped to less than 500,000 stems ha^{-1} with herbs still more than 60% and trees down to about 6% of the total number of stems. Of the tree genera, birches, primarily *Betula lenta*, dominated the population through year 5 (Fig. 2). By year 10, birches accounted for 45%, maples 33%, oaks 14%, and all other genera of trees combined only 8%.

A decade after harvest, the density of stump sprouts was low -- only 5% of the total number of stems of trees. Red maple was the most prolific sprouter, but sprouts were only 7% of all red maple stems at year 10. Oak sprouts were only 11% of all oak stems at the 10th year after cutting. At year 10, there were about 23,000 tree saplings tall enough to have a d.b.h. and only 7% of them were sprouts.

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[2]The use of trade, firm, or corporation names does not constitute an official endorsement or approval by the U.S. Department of Agriculture or the Forest Service of any product or service to the exclusion of others that might be suitable.

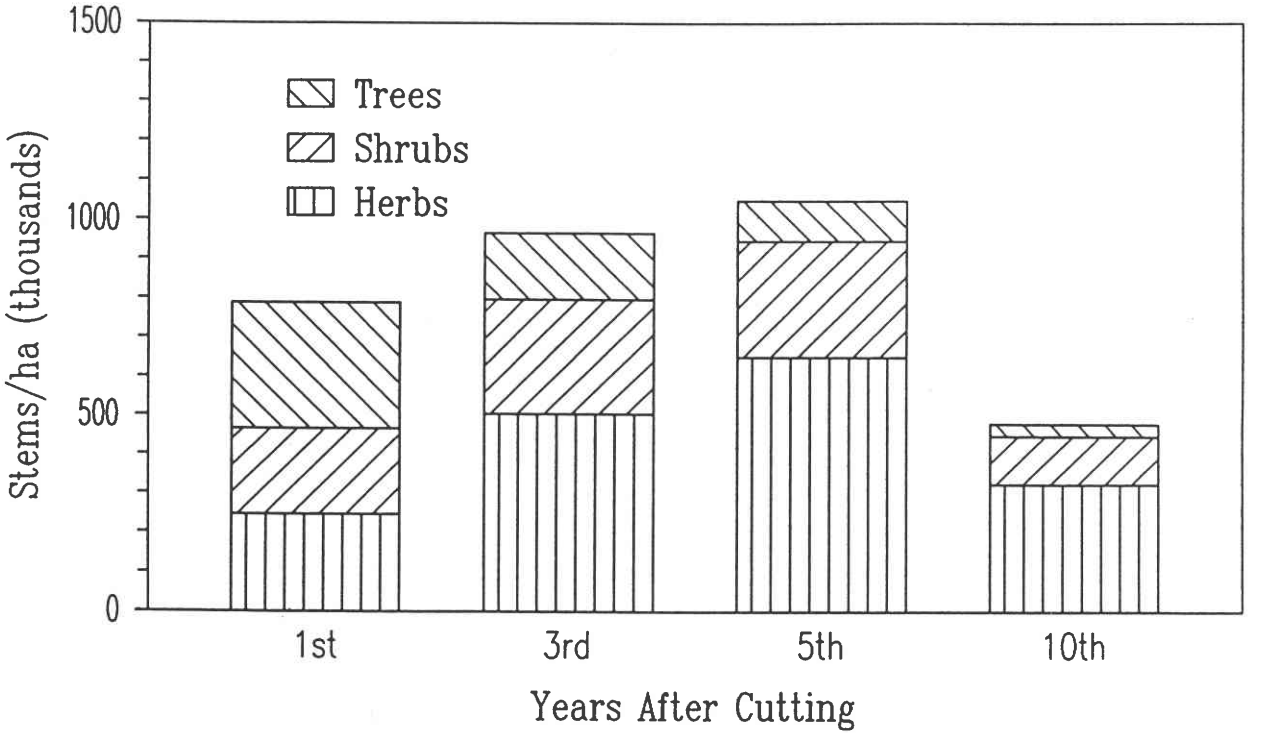

Figure 1. Density of all plants on the whole-tree clearcut at the Cockaponset State Forest, Chester, CT.

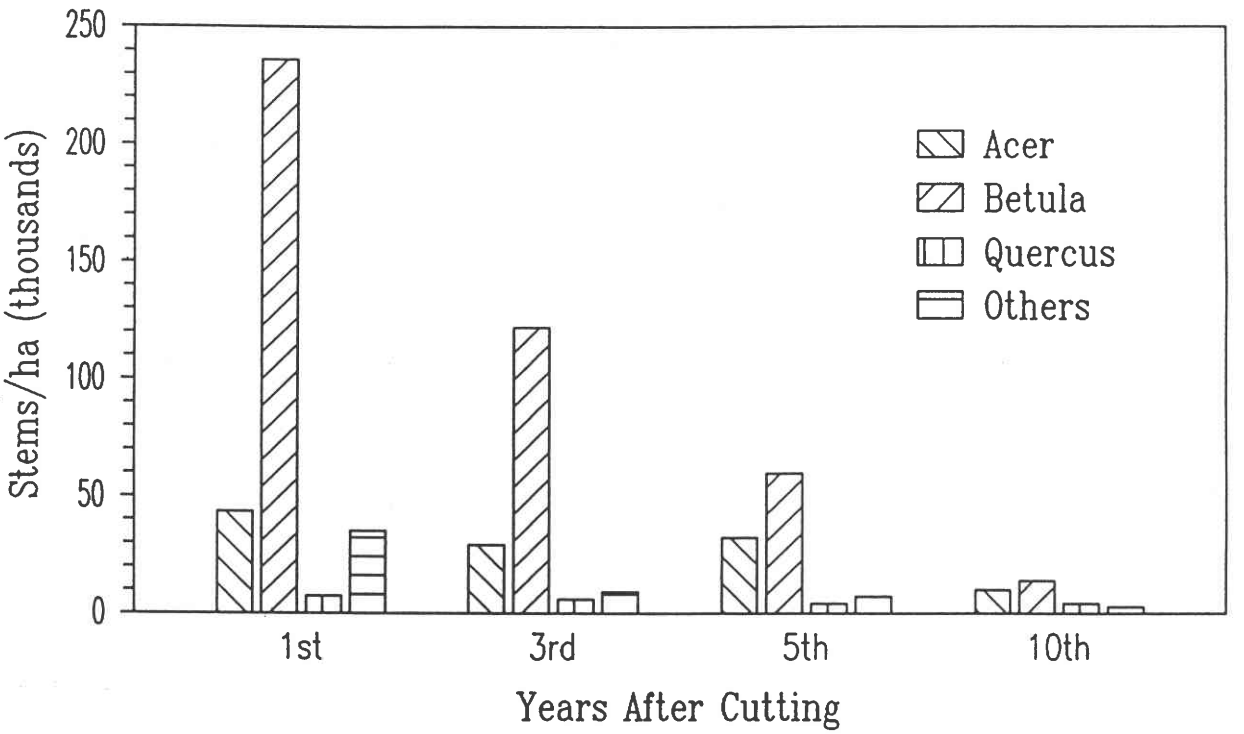

Figure 2. Density of tree genera on the whole-tree clearcut at the Cockaponset State Forest, Chester, CT.

Biomass

By August of the 1st year after cutting, there were nearly 2 tonnes ha^{-1} of aboveground dry weight of biomass with trees accounting for most of it (Fig. 3). By year 5, biomass had increased to 18 tonnes ha^{-1} with trees accounting for two-thirds of it. In the next 5 years the biomass more than doubled to nearly 42 tonnes ha^{-1}, with 88% in trees.

Of the tree genera present one growing season after cutting, the maples produced the most biomass, followed by chestnut (*Castanea dentata*), followed by the oaks, followed by the birches. By year 3, the maples still had accumulated the most biomass, followed by the birches, then by chestnut, and then by the oaks. By year 5, the order had changed again with the maples still dominant, in terms of biomass, followed by the birches, with the oaks in 3rd place and chestnut moved to 4th (Fig. 4). At year 10, the maples, birches and oaks accounted for 90% of the tree biomass with the maples at 45%, and chestnut relagated to 0.2% due to mortality from the chestnut blight. Sprouts accounted for more than 20% of the tree biomass. Maple sprouts were 20% of the maple population, but oak sprouts were nearly 50% of the oak population in terms of biomass. Of the total biomass in tree genera, 99% was in trees tall enough to have a d.b.h. at year 10.

Nutrients

The 1st year after cutting, 9 kg ha^{-1} of Ca accumulated in the above ground biomass with three-quarters of it in trees. Nearly 30 kg ha^{-1} of N accumulated in plant tissue the first year with about two-thirds in trees (Fig. 5). By year 10, about 150 kg ha^{-1} of Ca had accumulated in aboveground living plant tissue with 86% of it in tree genera. Nearly 200 kg ha^{-1} of N was sequestered in plant tissue at year 10 with about 86% of it in trees.

At year 10, the maples accounted for nearly 40% of all of the above ground biomass and contained about one-third of the Ca. The oaks contained only 20% of the biomass of the watershed, but they contained 27% of the Ca (Fig. 6). The maples contained 3.1 kg of Ca per 1000 kg of biomass, while the oaks contained 4.8 kg of Ca per 1000 kg of biomass. Of the total Ca in trees, 24% was in sprouts, but 43% of the Ca in oaks was in sprouts.

At year 10, maples had sequestered 35% of the N, and oaks 26% (Fig. 7). The maples contained 4.0 kg of N for each 1000 kg of biomass. The oaks contained 6.0 kg of N for each 1000 kg of biomass. Sprouts of all genera contained 24% of the N in all tree genera.

DISCUSSION

Prior to harvesting, the central hardwood forest at the Cockaponset State Forest in Chester, CT was a diverse forest with 22 species of trees and shrubs (Tritton et al. 1982). The first year after whole-tree clearcutting with 91% of the above-ground living biomass removed, there were 26 species of trees and shrubs in the regeneration. By the 10th year after cutting, the number of species of trees and shrubs had increased to 40.

The total numbers of stems of all plants increased through the 5th year after cutting and then by year 10 had dropped considerably as the canopy closed (Fig. 1). Trees were the most numerous the 1st year after cutting followed by herbs and then shrubs. Over the 10 year period the numbers of trees declined steadily; shrubs and herbs increased through year 5 and then declined by year 10. Of the tree species, sweet birch (*Betula lenta*) was clearly the most numerous throughout the 10 years of the study followed by red maple (*Acer rubrum*). The oaks were always important and were the 3rd most numerous tree by year 10 (Fig. 2).

The above ground living biomass of all plants on the watershed increased steadily over the 10 years after cutting, more than doubling between years 5 and 10. Tree species have clearly dominated the biomass throughout the post-harvest period inreasing from 67% of the biomass the first year after cutting to 88% by year 10 (Fig. 3). Herbs were a major part of the biomass through year 5. Shrub biomass has increased steadily throughout the period.

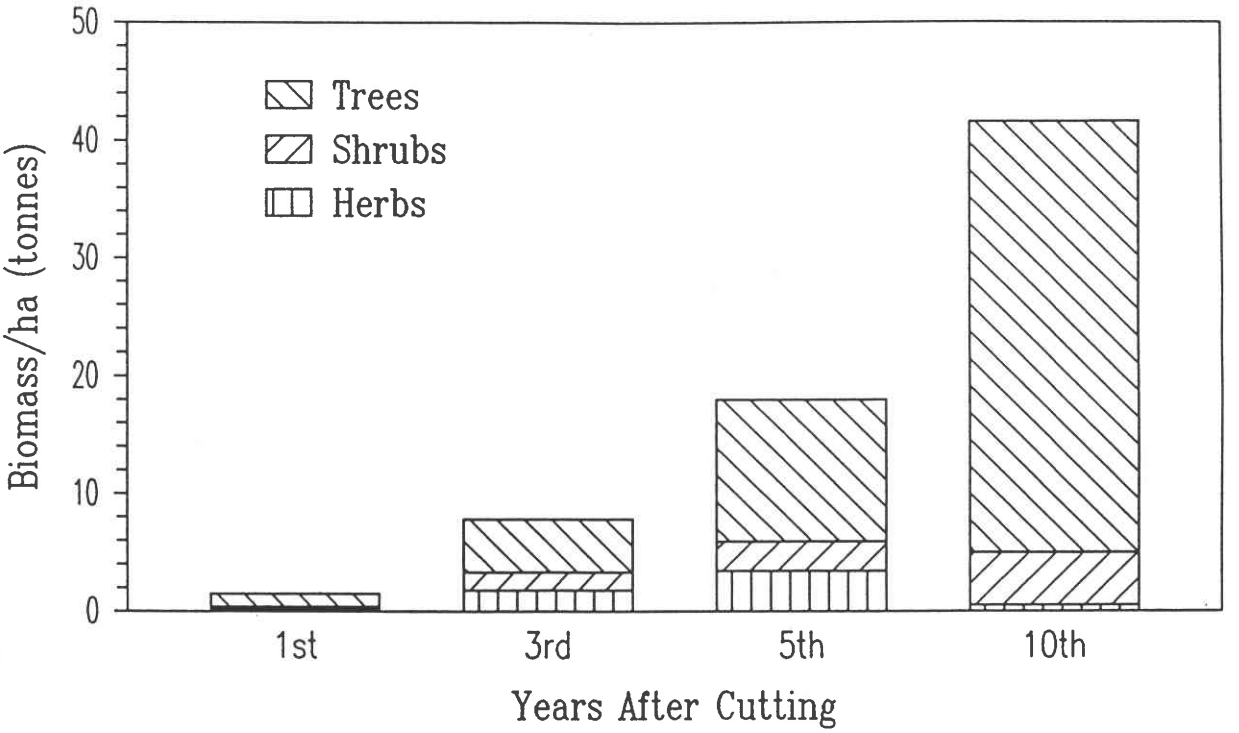

Figure 3. Biomass of all plants on the whole-tree clearcut at the Cockaponset State Forest, Chester, CT.

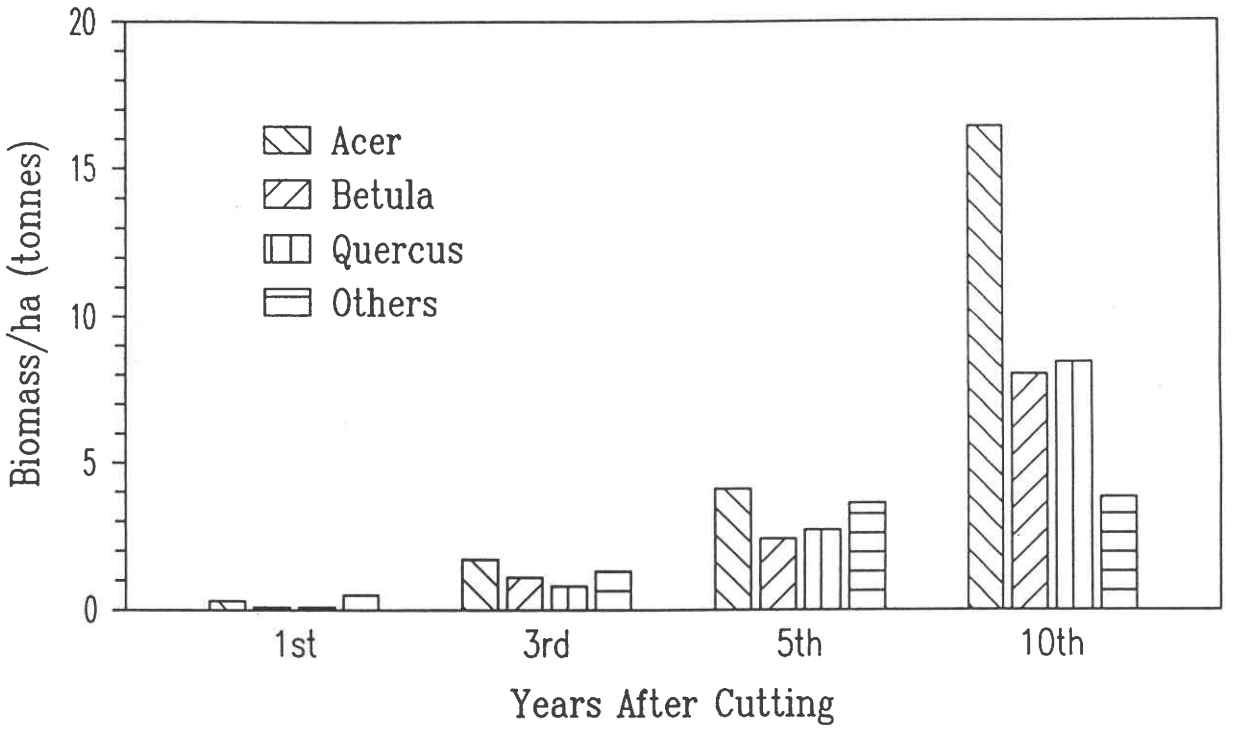

Figure 4. Biomass of tree species on the whole-tree clearcut at the Cockaponset State Forest, Chester, CT.

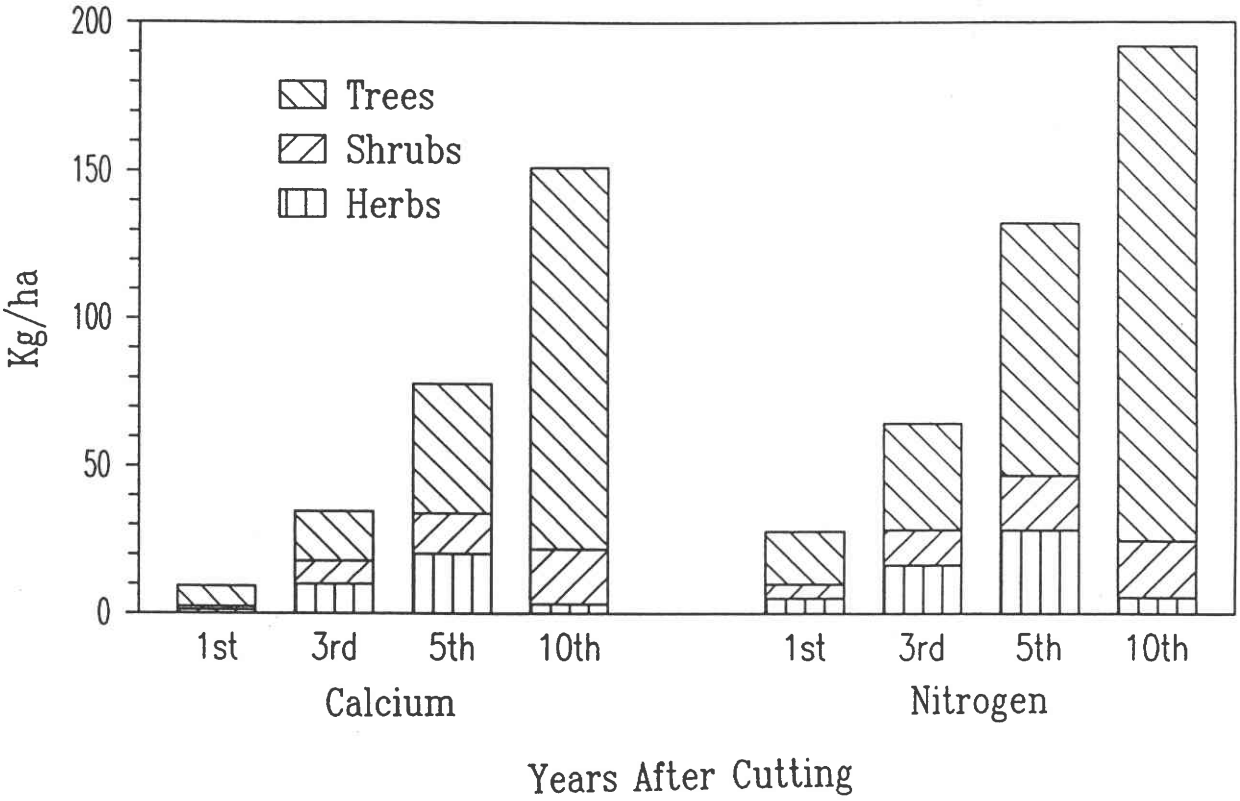

Figure 5. Calcium and nitrogen sequestered in all plants on the whole-tree clearcut at the Cockaponset State Forest, Chester, CT.

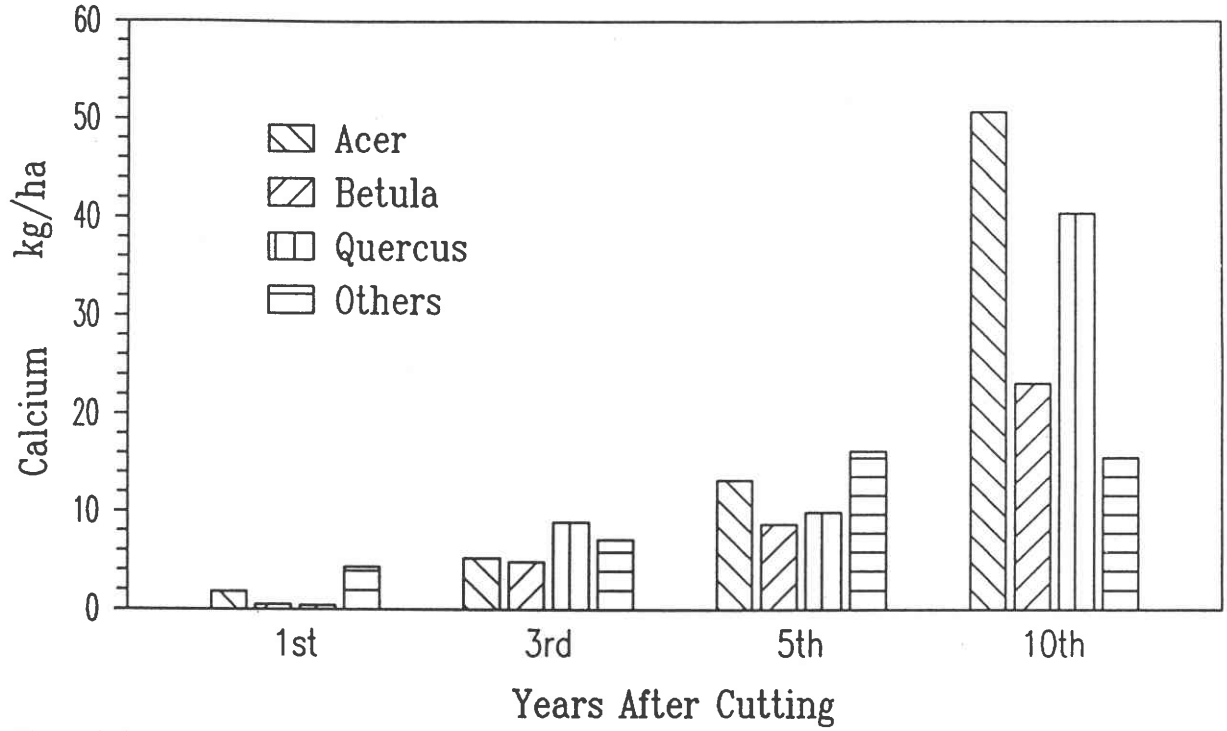

Figure 6. Calcium sequestered in tree genera on the whole-tree clearcut at the Cockaponset State Forest, Chester, CT.

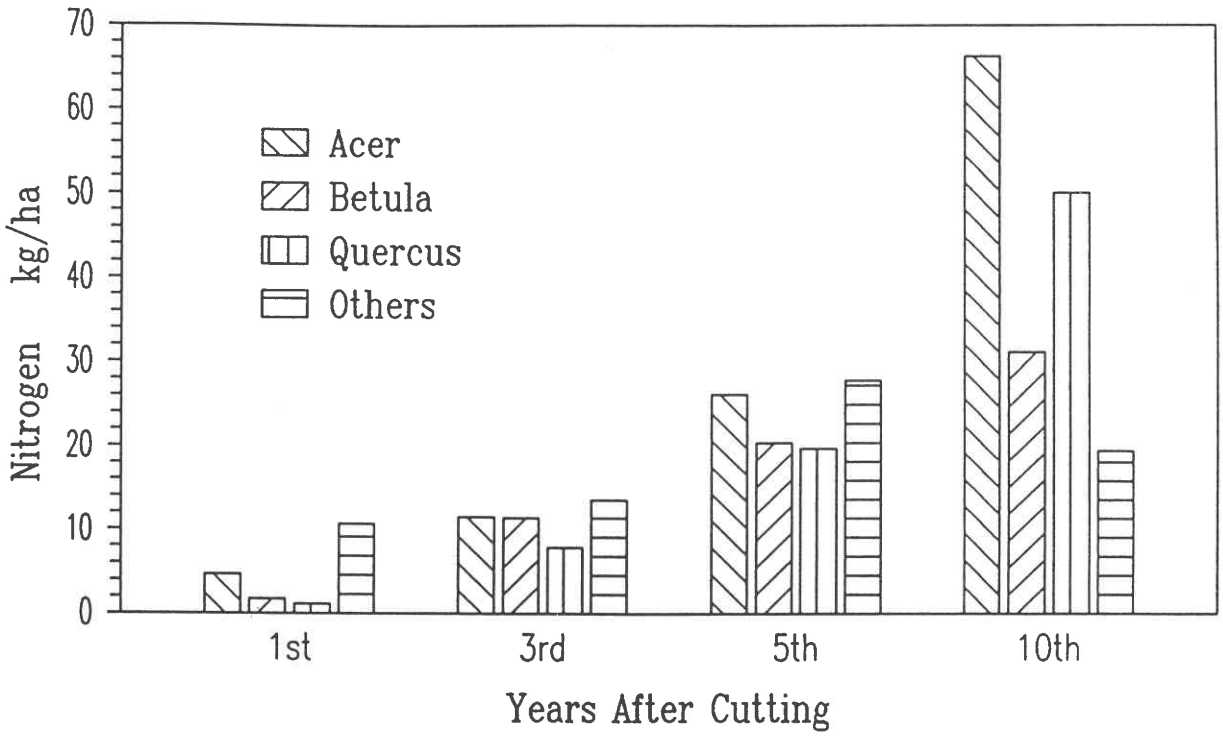

Figure 7. Nitrogen sequestered in tree genera on the whole-tree clearcut at the Cockaponset State Forest, Chester, CT.

Of the trees, red maple accumulated the most biomass of any single species throughout the sampling period followed by sweet birch (Fig. 4). After year 5, biomass of all of the oak species combined exceeded that of the birches.

The accumulation of nutrients in all plants on the site followed the same pattern as the biomass (Fig. 5). Trees sequestered much more Ca and N than did shrubs and herbs from the very first year after cutting. However, no one genus clearly dominated nutrient uptake until year 10, when maple and oaks sequestered the greatest amounts (Figs. 6 and 7).

The Future Forest

Comparisons of living trees and shrubs with stems ≥2.0 cm d.b.h. indicate that prior to cutting, dogwood (*Cornus florida*) was the most numerous tree on the watershed, but was an understory tree with very little biomass (Table 1). Of the major tree species, the maples were most numerous followed by the oaks and then the birches. At 10 years after cutting, the maples were the most numerous trees ≥2.0 cm d.b.h. followed by the birches and then by the oaks. Prior to cutting, the oaks had accumulated more than 50% of all of the biomass, Ca, and N in stems greater than 2.0 cm d.b.h. At 10 years after cutting, red maple had assumed that role. However, the oaks accounted for 24% of the biomass, 31% of the N, and 32% of the Ca.

Prior to cutting, there were more than 1,100 stems ha^{-1}; at year 10 there were more than 9,500 in trees and shrubs ≥ 2.0 cm d.b.h. which are likely to survive. This seems to be adequate stocking. Already by year 10, these trees and shrubs have accumulated 20% of the preharvest biomass; 52% of the preharvest N; and 21% of the preharvest Ca.

Table 1. Living trees and shrubs ≥2.0 cm d.b.h. on a 6 ha watershed on the Cockaponset State Forest, Chester, CT before and ten years after clearcutting.

Species	Before Cutting				Ten Years After Cutting			
	Density Stems/ha	Biomass Tonnes/ha	N ---kg/ha---	Ca	Density Stems/ha	Biomass Tonnes/ha	N ---kg/ha---	Ca
Quercus	202	93.2	176.0	399.3	1,350	8.1	48.4	38.5
Betula	174	39.6	65.7	89.2	2,058	4.2	18.4	13.4
Acer	305	14.4	29.8	51.0	3,717	15.5	62.5	47.2
Carya	58	4.9	9.1	18.6	400	1.3	7.6	5.0
Cornus	339	0.9	7.3	9.6	293	0.7	3.0	3.7
Other	85	15.4	14.8	20.8	1,720	4.0	16.6	14.0
Total	1,163	168.4	302.7	588.5	9,538	33.8	156.5	121.8

CONCLUSIONS

Clearcutting, especially whole-tree clearcutting, is a major disturbance of the forest ecosystem. The rate and success of the regeneration are of concern to forest landowners, foresters, and the general public. The Cockaponset site revegetated rapidly with herbs, shrubs, and tree seedlings during the first growing season after harvesting. Species density increased rapidly for the first 5 years and then declined due to crown closure. Biomass increased exponentially through year 10.

Even though clearcutting disrupts the forest nutrient cycles, regeneration including trees, shrubs, and herbs on the Cockaponset site accumulated nearly 25% of the preharvest levels of biomass, more than 60% of nitrogen, and about 25% of calcium by year 10.

There is always concern about the adequate stocking of commercial species. More than 9,500 stems of trees and shrubs ≥2.0 cm d.b.h. at year 10 seems to be adequate stocking, especially with the oaks playing a prominant role in both density and biomass.

Based on data from the first 10 years after clearcutting, it would seem that the future forest would be similar to the preharvest forest with red maple, oaks, and birches being the most numerous trees.

ACKNOWLEDGMENTS

This study was conducted in cooperation with the Office of the State Forester, Connecticut Department of Environmental Protection, Hartford, CT. The author wishes to thank Jane Hislop for performing the chemical analyses of the plant tissue, and Ian Halm for assistance in plot sampling and the grinding of the plant tissue.

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IMPACTS OF ELECTRIC DEER EXCLUSION FENCING AND SOILS ON PLANT SPECIES ABUNDANCE, RICHNESS, AND DIVERSITY FOLLOWING CLEARCUTTING IN PENNSYLVANIA

Jonathan Lyon and William E. Sharpe[1]

Abstract: Electric deer exclusion fencing has become a widely used management tool to promote hardwood regeneration in high deer browsing intensity areas. To assess the impacts of deer browsing on hardwood regeneration and vegetation patterns on clearcuts, six clearcuts with paired electric fenced and unfenced treatments were investigated. Additional data were collected on 10 sites with only fenced clearcut treatments. Overall browse damage on fenced treatments was lower than unfenced treatments, but there were substantial species-specific variability. The vegetation patterns on clearcuts were strongly influenced by pre-harvest ground covers of herbaceous and woody species. Comparisons between fenced and unfenced treatments showed that species richness was not significantly different for any site and that woody species diversity (Shannon Index) was significantly higher on a fenced versus unfenced clearcut on only one site. Observed vegetation patterns may also be influenced by soil nutrient and acidity status. Species richness was significantly related to exchangeable Ca and Al levels in the subsoil and organic horizon pH.

INTRODUCTION

There has been much concern surrounding the difficulties in regenerating desirable hardwood species following clearcutting in Pennsylvania and throughout the eastern hardwood region. There is also growing interest in evaluating clearcut plant communities in light of recent efforts to manage forests for biodiversity (Gove and others 1992; Hansen and others 1991; Niese and Strong 1992). Interpreting causal factors responsible for observed vegetation patterns, however, requires detailed analysis of an array of biotic and abiotic factors that may be influencing the structure and composition of vegetation on any given site. In Pennsylvania, excessive browsing by white-tailed deer (*Odocoileus virginianus virginianus* Boddaert) has been widely cited as the primary agent responsible for depleting advance regeneration stocks, reducing hardwood regeneration on clearcuts, and influencing species composition (Jordan 1967; Marquis 1981). However, ground covers of competing species (Drew 1988; Horsley 1977; Horsley and Marquis 1983; Kolb and others 1990), rodent damage and seed predators (Marquis and others 1976), poor seed production (Tryon and Carmean 1958), low levels of soil N and P (Auchmoody 1982), and soil acidity and soil nutrient imbalances (Tomlinson and Tomlinson 1990) may all play a role in determining the success of hardwood regeneration and the overall community that is regenerated after clearcutting.

Deer exclusion fencing has been used for many years as a research and management tool to both ascertain and ameliorate the impacts of deer browse on regeneration and revegetation of clearcuts (Frontz 1930; Grisez 1959; Marquis and Brenneman 1981; Marquis and Grisez 1978; Shafer and others 1961; Tilghman 1987; Trumbull and others 1989). Research results indicating the potential value of wire mesh fences in promoting hardwood regeneration have led to the development and widespread use of less expensive electric deer fencing in high deer density areas. Despite the expanded use of electric exclusion fencing , however, there have been few quantitative studies on the value of these fences in promoting hardwood regeneration (George and others 1991; Kochel and Brenneman 1987) and fewer studies on plant species richness and diversity. The current study was undertaken to obtain additional information on the effectiveness of electric exclusion fencing in reducing deer influence on the composition of plant

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communities, species richness, and woody plant diversity on clearcuts. The study also focused on the assessment of soil chemical factors at each site to determine if and how soil chemistry might be influencing vegetation patterns. Specifically, the study was designed to analyze three components of clearcut revegetation: (1) comparisons of woody and non-woody species density, height, and cover across a range of different sites, (2) comparisons of overall vegetation patterns, species richness, and woody species diversity, and (3) the potential influence of soil acidity factors in influencing hardwood regeneration and vegetation patterns.

METHODS

Study Areas

One of the aims of this study was to undertake a field evaluation of the effectiveness of electric fencing on clearcut sites that were representative of typical fencing efforts currently employed in Pennsylvania. The study areas were carefully chosen from a large pool of clearcuts located on Pennsylvania Bureau of Forestry lands. The 16 study sites supported even-aged stands, were approximately the same age when cut, had minimal slopes (0-5%), and were from 2 to 4 years old. Six primary sites had paired fenced and unfenced clearcut treatments, of which four were located on oak forest sites and two on northern hardwood sites. Ten secondary clearcut sites scattered across north-central Pennsylvania with only fenced treatments were used to provide supplemental information. The fencing used on each clearcut was a multi-strand electric fence after the design of Brenneman (1982). Fences were constructed soon after harvest operations and were maintained by Bureau of Forestry personnel. At each site, all vegetation plots were located on the same soil series and soil sample results are based on analysis of the specific soil series. To ascertain any potential influence of soil chemical factors, the sites chosen also represented a range in soil conditions. General characteristics of the study sites are given in Table 1.

Study Design and Field Measurements

Each of the six primary study sites was divided into three treatments: a fenced clearcut, an unfenced clearcut, and an understory of an adjacent uncut forest stand similar in composition to the original, pre-clearcut stand. A stratified sampling design was used for vegetation analysis. Nine, 1.828 m radius (6-ft radius), sample plots were located within each treatment at each site (3 treatments x 9 plots = 27 plots per site). Bureau of Forestry field data on the composition of the over- and understories at each site prior to clearcutting were used to verify the validity of using vegetation data from understory plots adjacent to clearcuts as a proxy for pre-clearcut vegetation data in this study. The locations of all vegetation plots were restricted to a single soil type within each site. Soil types at each site were identified using SCS soil maps with ground verification using small, hand excavated soil pits. Vegetation plots were laid out on a 25 m x 25 m grid over the specific soil type on each of the 3 treatments with certain restrictions; plots had to be at least 30 m from the exclusion fences and not to be located on areas with major disturbance such as skid trails, uprooted trees or exposed mineral soil.

Vegetation plots were established during the spring of 1992. Full scale quantitative sampling was conducted on each site in August of 1992. On each of the fenced and unfenced clearcut plots, the number, height, and browse status of each species of woody plant were recorded. On the uncut, unfenced forest plots, the number, height, and browse status of each species of woody seedling and sapling were recorded. Cover of both shrubs and herbaceous species was also determined using the Braun-Blanquet system (7 cover categories). Browse damage on hardwood seedlings and saplings was noted if the top or lateral stems showed browse damage. No separation was made between deer browse and other mammal browse due to difficulties in distinguishing browse damage in the field. However, the high population of deer, and widespread evidence of deer activity at each site, suggest that deer were likely to have been responsible for most of the browse damage observed.

Table 1. General characteristics of six primary, and ten secondary study sites.

Site	Location State Forest	Size (ha)	Elev (m)	Stand Age When Cut	Year Cut	Date Fenced	Preharvest Overstory	Soil Series
				PRIMARY SITES				
1	Bald Eagle	10.1	510	80	1988	5/89	CO,RO,WO	Laidig
2	Bald Eagle	19.8	520	82	1990	6/90	RO,CO,WO	Hazleton
3	Forbes	6.1	830	65	1989	7/89	RO,RM	Dekalb
4	Moshannon	40.5	695	78	1989	8/89	BC,RO,RM	Cookport
5	Susquehannock	29.9	680	84	1990	7/90	BC,BE,RM	Clymer
6	Tioga	23.8	640	90	1989	4/89	SM,BC,BE	Lordstown
				SECONDARY SITES				
7	Moshannon	19.0	695	82	1989	8/89	RO,RM	Hazleton-Clymer
8	Moshannon	12.1	600	78	1989	9/89	RM,RO	Hazleton-Clymer
9	Moshannon	8.1	620	80	1989	9/89	RO,RM	Hazleton-Clymer
10	Moshannon	8.2	525	88	1990	7/90	RO,WO	Berks
11	Moshannon	21.9	680	85	1988	5/89	RO,RM	Cookport
12	Tioga	6.9	720	90	1990	9/90	RO,WO	Clymer
13	Tioga	12.5	730	80	1987	5/88	RO,RM	Oquaga
14	Elk	15.8	520	68	1989	8/89	RM,RO,WO	Wharton
15	Susquehannock	26.3	660	84	1989	5/90	BC,RM,SM	Leetonia
16	Tiadaghton	27.5	680	75	1989	5/90	RM,BE	Wurtsboro

Overstory Species Codes:

RO - Red Oak (*Quercus rubra* L.)
CO - Chestnut Oak (*Quercus prinus* L.)
SM - Sugar Maple (*Acer saccharum* Marsh.)
RM - Red Maple (*Acer rubrum* L.)
WO - White Oak (*Quercus alba* L.)
BC - Black Cherry (*Prunus serotina* Ehrh.)
BE - American Beech (*Fagus grandifolia* Ehrh.)

Soil samples from all horizons to a depth of 30 cm were collected from a random sub-sample of 4 of the 9 vegetation plots. Soils were analyzed by horizon, and soil analysis was performed at The Pennsylvania State University and included pH (glass electrode with 1:1 [w:v] H_2O), soil texture, CEC and base saturation (North Dakota State University 1988), organic matter, total N , and exchangeable Ca, Mg, K, Mn, and Al using a 0.01M $SrCl_2$ extraction (Joslin and Wolfe 1989). The latter analysis was performed in order to estimate the "plant available" fraction of these cations. Table 2 provides a summary of the mineral horizon soil conditions (15-30 cm depth) at each of the six primary sites.

Data Analysis

Differences in vegetation and soil parameters across treatments for each site were analyzed by ANOVA and mean separations were performed using Fisher's LSD test. Herbaceous cover was analyzed after the methods of Bannister (1966). Species richness comparisons were based on the mean number of species per plot within each treatment. Total species richness was measured as the sum of all species found within a treatment. Woody plant diversity was measured using the Shannon-Weiner Index (Magurran 1988). Herbaceous species diversity was not assessed because the Braun-Blanquet cover estimates employed in the study were not amenable to diversity measures (Magurran 1988). All significant differences are reported at the alpha ≤ 0.05 level. Regression and multiple regression analyses were used to determine relationships between plant composition and the soil parameters being studied.

Table 2. Soil characteristics (15-30 cm depth) of the six study sites.

SITE	pH[1]	C:N	%BS	CEC[2]	ppm[3] Ca	Mg	K	Al	molar Ca:Al
1 Understory	4.13	44.3	7.2	6.4	2.0	0.3	3.1	9.7	0.14
Clearcut[4]	4.20	36.3	7.5	8.9	2.4	0.4	2.7	9.2	0.18
2 Understory	4.05	30.0	8.1	9.8	4.4	3.2	8.2	8.1	0.37
Clearcut	4.13	31.3	6.1	11.9	5.6	4.4	9.4	7.3	0.52
3 Understory	4.02	26.7	6.5	11.3	3.6	3.7	7.4	21.3	0.11
Clearcut	4.08	30.9	5.2	14.3	3.2	4.1	7.8	19.7	0.11
4 Understory	4.65	32.2	12.3	8.9	5.2	4.9	16.0	11.1	0.32
Clearcut	4.59	27.9	14.6	7.6	6.0	4.6	17.2	10.5	0.38
5 Understory	4.60	22.2	11.1	10.1	8.4	7.1	15.2	15.6	0.36
Clearcut	4.56	28.3	13.0	11.3	11.6	7.3	17.2	11.3	0.69
6 Understory	4.51	21.3	6.9	16.1	14.0	6.8	21.9	13.8	0.67
Clearcut	4.49	24.6	7.5	16.1	12.4	6.1	22.7	12.7	0.66

[1] pH in 1:1 (w:v) H_2O [2] CEC in cmol(+) kg^{-1}
[3] Using 0.01 M $SrCl_2$ extraction (Joslin and Wolfe 1989)
[4] Clearcut soil values refer to fenced clearcut sites

RESULTS

Pre-Clearcut Vegetation Versus Understory Vegetation

In most cases, the composition of the over- and understory on the pre-clearcut stands paralleled the 1992 sampling plots located adjacent to the existing clearcuts (Table 3). Based on this information, the assumption was made that the pre-harvest clearcut vegetation was similar to the adjacent uncut, unfenced plots before cutting. Nonetheless, caution is warranted in interpreting the results based on this assumption.

Fencing Impacts on Browse Damage

Based on the single year (1992) browse damage survey, the overall level of browse damage on woody species was significantly lower within fenced clearcut treatments than on either the unfenced or non-clearcut, unfenced treatments. Species-specific browse damage, however, exhibited substantial variation across sites and treatments. Northern red oak (*Quercus rubra* L.) and sweet birch (*Betula lenta* L.) were the only species that consistently had significantly higher percentages of browsed stems in unfenced treatments (61-82% and 45-100% for red oak and sweet birch respectively), compared to fenced treatments (16-43% and 6-38%). Browse on pin cherry (*Prunus pensylvanicum* L.) and black cherry (*Prunus serotina* Ehrh.) was not significantly different on fenced versus unfenced treatments indicating deer penetration through the fence or other mammal browsing inside the fence. Red maple (*Acer rubrum* L.) browse damage was highly variable ranging from 0-100% on both fenced and unfenced treatments across the six study sites. There were no significant correlations between species-specific stem densities and the proportion of those stems browsed.

Table 3. Comparison of pre-clearcut regeneration information[1] versus 1992 understory vegetation survey.

	SITE 1		SITE 2		SITE 3	
Seedlings/ha	pre-cut	1992 survey	pre-cut	1992 survey	pre-cut	1992 survey
Hardwood seedlings	11600	18890	10280	14850	<3000	1810
Oak seedlings	3700	3890	7410	8100	<500	100
Vaccinium cover	67%	82%	37%	49%	sparse	0%
Kalmia cover	40%	46%	53%	44%	<5%	5%
grass cover	3%	1%	sparse	2%	<5%	5%
fern cover	sparse	1%	sparse	0%	70%	95%

	SITE 4		SITE 5		SITE 6	
Seedlings/ha	pre-cut	1992 survey	pre-cut	1992 survey	pre-cut	1992 survey
Hardwood seedlings	25700	18500	29100	23000	140000	195000
Oak seedlings	2500	3200	0	0	0	0
Prunus serotina L.	15000	7500	12000	8530	24000	29100
Vaccinium cover	15%	23%	sparse	0%	sparse	0%
Kalmia cover	0%	0%	sparse	0%	0%	0%
grass cover	10%	12%	>15%	20%	0%	0%
fern cover	65%	81%	>20%	19%	0%	1%

[1] Based on Bureau of Forestry records. All data presented are based on mean densities or mean % cover on vegetation plots except for values with < or > signs which indicate maximum or minimum estimates.

Woody Stem Densities and Importance Values

Total woody stem densities varied across sites and treatments (Table 4). Stem densities on fenced treatments differed significantly from unfenced treatments on sites 2 and 6; the former with lower density on the unfenced treatment and the latter with higher density on the unfenced treatment. Woody stem densities on the 10 supplemental sites ranged from $41x10^3$ ha^{-1} to $351x10^3$ ha^{-1} in the adjacent forest understory and from $23x10^3$ ha^{-1} to $369x10^3$ ha^{-1} on fenced clearcuts. Based on pooled data from the 6 primary and 10 supplemental sites, understory stem densities were a poor, nonsignificant predictor of stem densities on fenced clearcuts (r^2 = 38.2 %). Table 4 shows calculated importance values (IV = relative density + relative frequency) for all woody species encountered in the sampling on the six primary sites. Site specific comparisons across treatments show widespread similarity in IV's for most woody species, with some key exceptions. The IV's for red maple (*Acer rubrum* L.) in the adjacent understory were significantly greater than those in the clearcut treatment for sites 3, 4, and 6. The same pattern was observed for American beech (*Fagus grandifolia* Ehrh.) on site 5. IV's for pin cherry were significantly higher on unfenced versus fenced treatments on sites 3, 5, and 6 while sweet birch showed the reverse trend on those same sites. The IV's for laurel (*Kalmia latifolia* L.) and *Vaccinium* spp. varied little; however, on site 2, laurel showed a decrease in IV on the clearcut treatments and on site 4, exhibited an increase in IV on the clearcut treatments.

Table 4. Comparison of importance values[1] (IV) for woody vegetation by treatment.
N (not cut, unfenced), F (fenced clearcut), U (unfenced clearcut).

SITE	1			2			3			4			5			6		
	N	F	U	N	F	U	N[2]	F	U	N	F	U	N	F	U	N	F	U
TREES																		
Quercus alba L.	2	-	-	5a	2a	5a	-	-	-	3a	4a	3a	-	-	-	-	-	-
Quercus prinus L.	16a	10a	13a	21a	19a	17a	-	-	-	-	-	-	-	-	-	-	-	-
Quercus rubra L.	12a	9a	6a	5a	5a	3a	*23*	-	-	25a	22a	15a	-	-	-	-	-	-
Quercus velutina Lam.	-	-	-	-	2	-	-	-	-	-	2	-	-	-	-	-	-	-
Acer rubrum L.	22a	19a	17a	35a	24a	39a	*115a*	14b	16b	99a	57b	77b	44a	32b	19b	55a	30b	13c
Acer saccharum Marsh.	-	-	-	-	-	-	-	-	-	-	-	-	5a	4a	7a	-	-	-
Fagus grandifolia Ehrh.	-	-	-	-	-	-	-	-	-	-	-	-	97a	23b	26b	15a	24a	18a
Sassafras albidum (Nutt.) Nees	-	13a	14a	-	-	-	*40a*	82a	84a	-	2a	2a	-	-	-	-	-	-
Liriodendron tulipifera L.	-	-	-	-	-	-	-	10a	5a	-	-	-	-	-	-	-	-	-
Acer pensylvanicum L.	-	-	-	-	-	-	-	-	4	-	-	-	-	8a	4a	12a	31a	29a
Prunus serotina Ehrh.	-	-	-	-	-	-	-	-	-	24	32a	29a	-	19a	8a	34a	20a	25a
Prunus pensylvanica L.f.	-	-	-	-	7	-	*23a*	41a	58b	-	7	-	19a	66b	103c	68a	44b	82a
Betula lenta L.	-	-	-	-	-	-	-	28a	4b	-	-	-	38a	40a	20b	14a	45b	31a
Betula alleghaniensis Britton	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	-
Pinus strobus L.	-	-	-	-	-	3	-	-	-	-	-	-	-	-	-	-	-	-
Tsuga canadensis (L.) Carr.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	7	-	-	-
Populus tremuloides Michx.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-
SHRUBS																		
Kalmia latifolia L.	21a	24a	23a	30a	14b	11b	-	19a	21a	-	-	-	-	-	-	-	-	-
Vaccinium spp.	107a	107a	107a	91a	111a	102a	-	-	5	14a	28b	33b	-	-	-	-	-	-
Amelanchier spp.	-	-	-	3	-	3	-	6	4	19a	23a	28a	-	3a	4a	-	-	-
Myrica asplenifolia L.	-	-	4	-	-	5	-	-	-	-	2	-	-	-	-	-	-	-
Hamamelis virginiana L.	20a	19a	16a	12a	15a	13a	-	3	-	-	2a	2a	-	5a	2a	-	-	-
Viburnum spp.	-	-	-	-	-	-	-	-	-	17a	20a	12a	-	-	-	-	-	-
Hardwood seedling density[3]	19a	14a	6a	22a	23a	32a	*1*	4a	3a	78a	52a	59a	23a	78b	99b	195a	41b	71b
Total stem density[3]	29a	22a	7b	32a	27a	38a	*1*	5a	4a	87a	78a	84a	23a	79b	100b	196a	41b	71b

Different letters denote significant differences ($P \leq 0.05$) in means within each site using Fisher's LSD test.
[1] IV = (relative density + relative frequency)
[2] IV values for the uncut forest are of limited value due to the extremely low stem density
[3] (1000 / ha)

Fencing Impacts on Seedling Heights

Figure 1 illustrates differences in mean heights of the dominant (tallest), non-stump sprout origin, tree seedling stems of each tree species found on fenced versus unfenced treatments. On sites 1 and 3 the dominant seedling heights were greater in the fenced than in the unfenced treatments. The relatively tall dominant seedling heights on site 4 reflected the dominance of black cherry growth at the site. However, if black cherry was not included in the mean determinations, there were still no significant differences in dominant seedling heights between the fenced and unfenced treatments. Overall growth performance of hardwood seedlings appeared to be poor relative to that of other published height data. On the oak forest sites, northern red oak seedling height was also found to be positively and significantly related to covers of *Vaccinium* spp. ($r^2 = 32.4\%$).

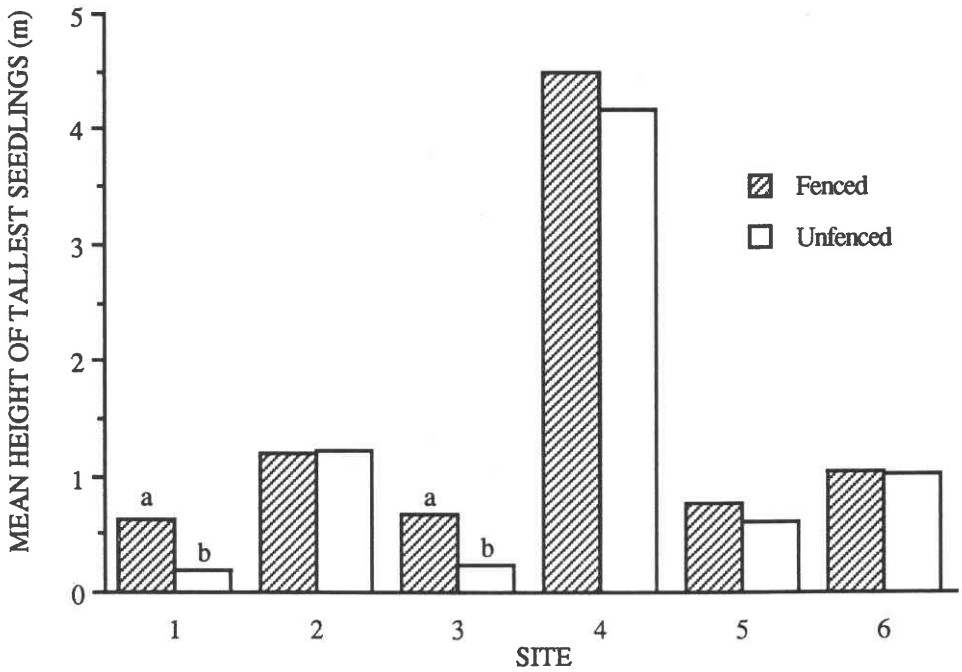

Figure 1. Mean height of the dominant commercial tree seedlings in fenced vs. unfenced clearcuts. Letters denote significant differences ($P \leq 0.05$).

Fencing Impacts on Herbaceous Species

Table 5 provides a condensed summary of herbaceous species cover on the six primary study sites. Ground covers of fern, grass and sedge were found across all sites and treatments. Fern cover was generally higher on unfenced versus fenced treatments, although significant differences were found only on sites 3 and 6. Site 6 also had significantly greater fern cover on the clearcut treatments compared to the adjacent understory. Grass and sedge covers on clearcut treatments were generally higher than in the understory. The only significant difference in grass and sedge cover on clearcut treatments was observed on site 5. Cover of *Rubus* on unfenced treatments was significantly lower than the fenced treatment on sites 3 and 5. The remaining covers of herbaceous species were highly variable across sites and treatments, yet the herbaceous cover at each site was generally dominated by a single herbaceous species.

Fencing Impacts on Species Richness and Diversity

Total and woody species richness comparisons are summarized in Table 6. Total species richness ranged from 9 to 26 species across sites and treatments. Total species richness on the 10 supplemental sites ranged from 9 to 29 species. Comparisons of the mean number of species per plot between fenced and unfenced clearcut treatments showed no significant differences in either total or woody species richness. Sites 4 and 5 did have significantly lower total species richness in the understory compared to the clearcut treatments. Regression analysis was performed to predict species richness using herbaceous covers on fenced clearcuts (pooled from the six primary and 10 supplemental study sites). Significant negative relationships were found between total species richness and grass and sedge cover ($r^2 = 31.6\%$) and fern cover ($r^2 = 34.4\%$), although the variation explained by these variables is modest.

Table 5. Percent cover[1] of dominant herbaceous species.
N (not cut, unfenced), F (fenced clearcut), U (unfenced clearcut).

SITE	1			2			3			4			5			6		
	N	F	U	N	F	U	N	F	U	N	F	U	N	F	U	N	F	U
Ferns[2]	0.5	4.0	1.0	0.0	1.7	0.0	95.1	74.2	91.5	81.0	47.7	73.0	18.9	18.7	28.1	5.0	5.8	29.0
Grass/Sedge	0.7	1.6	0.0	1.7	3.3	1.0	5.1	16.3	14.5	12.1	48.6	53.3	20.0	27.6	14.1	2.5	8.1	14.1
Rubus	1.0	0.0	0.0	0.0	7.0	0.4	0.0	31.4	16.3	3.1	12.3	12.1	2.0	27.6	14.5	4.0	19.7	17.2
Other Herbs	12.0	64.9	18.6	16.3	19.9	53.6	51.7	16.3	57.3	27.5	44.3	51.7	4.2	9.1	5.3	3.7	9.6	9.7

[1] % covers listed have been back transformed from Domin values (Bannister 1966).
[2] predominantly hay-scented fern (*Dennstaedtia punctilobula* [Michx.] Moore)

Table 6. Species richness[1] comparisons for three treatments at each site.

SITE	1	2	3	4	5	6
			total species richness			
Not cut (adjacent forest understory)	9	14	16	19	9	12
Fenced Clearcut	13	16	19	26	14	15
Unfenced Clearcut	11	13	18	26	18	17
			woody species richness			
Not Cut (adjacent forest understory)	7	8	4	7	4	7
Fenced Clearcut	7	9	8	12	8	6
Unfenced Clearcut	8	10	9	9	10	7

[1] Species numbers represent total number of species tallied at each site

Computed Shannon-Weiner diversity index values (H') are listed in Table 7. Comparison of diversity indices showed no significant difference in diversity between fenced and unfenced treatments on five of the six primary sites. Only site 5 had a significantly lower H' value on the unfenced treatment. This low H' value can largely be attributed to the influence of a high proportion of pin cherry seedlings on the unfenced site.

Table 7. Shannon diversity values (H') for treatments on all sites.

SITE	1	2	3	4	5	6
Not Cut (adjacent forest understory)	0.52a[1]	0.96a	0.95a	0.89a	0.98a	1.15a
Fenced Clearcut	0.55a	0.47b	1.20a	1.67b	1.54b	1.40a
Unfenced Clearcut	0.41a	0.68b	1.18a	1.26b	0.68a	1.24a

[1]Letters denote significant differences ($P \leq 0.05$) in means within each site.

Potential Impacts of Soil Parameters

To test the potential direct and/or indirect influence of soil parameters on species richness, soil and species richness data from the 10 supplemental sites were added to the six primary study site data. Only data on fenced clearcuts were used in this analysis due to the lack of unfenced treatments on the supplemental sites. Regression and multiple regression analysis indicate that species richness on fenced clearcuts covaried with some of the soil parameters investigated. As shown in Figure 2, a significant positive relationship was found between species richness and exchangeable Ca levels in the subsoil ($r^2 = 61.0\%$). A significant positive relationship was also found between species richness and organic horizon pH. ($r^2 = 22.4\%$). Significant negative relationships between species richness and exchangeable Al ($r^2 = 33.4\%$) and the molar Ca:Al ratio in the mineral soil horizons ($r^2 = 35.1\%$)

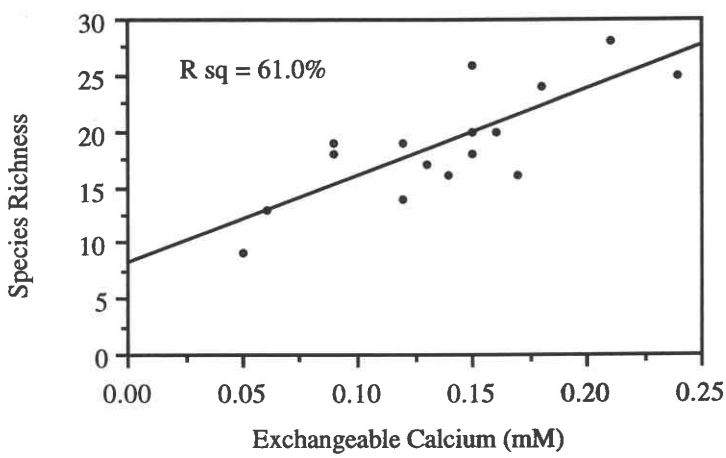

Figure 2. Regression between species richness on fenced clearcuts and exchangeable Ca in the mineral subsoil (15-30 cm depth).

DISCUSSION

No accurate data were available as to deer densities at the different study sites during the study period. Thus, comparisons with previous deer exclusion and enclosure studies must take into account potential differences in deer browsing pressure as well as differences in foraging patterns. Historically, most exclusion studies have employed woven wire mesh fencing. It is likely that the five- and six-strand electric exclusion fencing present on the study sites discussed here provided less protection than woven mesh fencing. George and others (1991) reported increased deer penetrations through electric fencing after two years of service. The results of the present study also are based on a single sampling period (1992); and the results must be interpreted within that limited framework. These caveats notwithstanding, comparisons of the results of the present study with woven mesh studies can provide useful information in ascertaining the effectiveness of the electric fencing.

Overall, electric exclusion fencing treatments provided limited improvement in promoting increased stocking of desirable hardwood species, enhancement of total species richness or woody species diversity compared to non-fenced treatments. Based on the assumption that the vegetation data collected on adjacent, uncut stands in the current study is representative of the pre-clearcut vegetation on the site, the composition of clearcut vegetation was strongly influenced by the composition of species prior to clearcutting. This result implies that the influence of fencing must be assessed within the framework of vegetation cover prior to harvest. Similar patterns in post-clearcut vegetation composition were noted by Hughes and Fahey (1991). In addition, the results of the current study showed few differences in shrub densities and IV's between understory, fenced and unfenced treatments at each site similar to results presented by Martin and Hornbeck (1989). Few significant differences in absolute woody stem densities were found between fenced and unfenced treatments in this study and by other investigators (Jordan 1967; Marquis 1981). The present study also found significant browse damage and IV declines for sweet birch on unfenced versus fenced treatments as reported by Marquis (1981) and Shafer and others (1961). Other tree species densities and IV's showed variable patterns across treatments and sites. Lack of consistent increases in fern cover in unfenced versus fenced treatments in the current study was also reported by Trumbull and others (1989). Also, paralleling their results, we found a general increase in grass cover following clearcutting on half the study sites.

Contrary to previous studies that found pin cherry nearly eliminated by browse in unfenced clearcut treatments (Marquis 1981), the current study found no differences in mean seedling height or dominant seedling height for pin cherry between fenced and unfenced treatments. In addition, the IV of pin cherry significantly increased in unfenced clearcut treatments on 3 of the 5 sites where it was present. We also found no significant increases in beech or striped maple density, height or IV on unfenced plots as reported by Marquis (1981). These results indicate first, that browse impacts can be highly site specific and second, that vegetation within electric exclusion fencing did not differ as dramatically from unfenced treatments as reported in previous woven mesh fencing studies. The latter point is further reinforced by comparing our results with the deer enclosure study of Tilghman (1989). Tilghman reported that the mean heights of the tallest woody stems on clearcuts were significantly shorter under higher deer densities. While we do not have accurate deer density estimates in the current study, our results show significantly lower heights for dominant woody stems on only 2 of 6 unfenced plots. Thus, despite increased browse pressure on unfenced seedlings, on 4 of the 6 study sites we investigated, some stems were able to reach heights comparable to those within the exclusion fencing.

The present study sites were from 2 to 4 years old and the low woody species richness and diversities observed could be considered an artifact of clearcut age (Gove and others 1992). The improbable recruitment of new seedlings into the clearcuts, however, suggests that woody species richness will show few gains in the short-term. It is difficult to make meaningful comparisons in species richness and diversity across geographical regions which support very different assemblages of species and where different sampling designs have been employed. Nonetheless, the range in woody species richness (4-12 species) and the range in Shannon diversity values reported here (0.41-1.67), although low, fall in the range of values reported by other authors (Locasio and others 1991; Niese and Strong 1992; Tilghman 1989). In terms of herbaceous species richness, the current study showed few differences on fenced versus unfenced clearcuts and only modest gains in numbers of herbaceous species on 2 to 4 year old clearcuts when compared with the understory. This is in contrast to Stransky and others (1986) who reported a greater than doubling

of herbaceous species following the first year after clearcutting of a loblolly pine-hardwood forest and a peak in species richness after 3 years.

Regression analysis of soil variables and species richness on clearcuts in this study provides some preliminary evidence that soil acidity factors may be directly or indirectly influencing plant species growth and composition. Similar to the present study, studies on the North Carolina Piedmont have noted a strong correlation between calcium and magnesium levels in soil and species richness (Palmer 1990; Peet and Christensen 1980). Sharpe and others (1991) also reported increased diversity following liming in a forest understory in western Pennsylvania. In the current study, the low pH's and the low base saturations observed point to potential increased solubility of Al which might play a role in inhibiting cation availability and uptake and ultimately reducing root and shoot growth. In fact, we found significant negative relationships between species richness and both exchangeable Al in the subsoil and the molar Ca:Al ratio in the subsoil. The potential negative impacts of high Al and low Ca levels in subsoils on root uptake, root growth, and overall vigor have been addressed by Tomlinson and Tomlinson (1990). The combined impact of the predisposing stress of extremely acidic soils and the contributing stress of herbivory by deer may be too severe for regenerative success on some sites and indicate that soil evaluation should be considered as an integral component when formulating forest management strategies on extremely acidic forest soils.

SUMMARY

The influence of electric deer exclusion fencing on vegetation patterns, species richness, and woody species diversity were investigated on 16 clearcut sites across Pennsylvania. Many site specific and interacting factors are likely to have influenced hardwood regeneration success and the growth and development of plant communities on these sites. The results of this study indicated that hardwood seedling density, species richness, and woody species diversity were not enhanced by the use of electric deer fencing on the hardwood clearcuts studied. Some additional conclusions can be drawn from the current study.

1. Pre-harvest (understory) ground covers of both woody and herbaceous species appeared to have had an impact on species composition in clearcuts regardless of the presence of electric exclusion fencing as evidenced by a high degree of overlap between pre- and post-harvest plant communities. In addition, comparisons of the importance values of woody species (especially shrubs) across fenced versus unfenced treatments on all sites showed few significant differences.

2. Electric deer fencing did reduce browse damage but did not promote increased species richness on clearcuts. No significant differences in total or woody species richness were observed across treatments. Computed Shannon diversity values for woody species on fenced, versus unfenced, clearcuts were significantly higher on only 1 of 6 fenced clearcut treatments.

3. On sites with extremely acidic soils, soil acidity/nutrient factors may be influencing both hardwood regeneration success and the overall composition of plant communities. Because of the variable effectiveness of electric exclusion fencing, it was difficult in this study to separate the effects of deer browse pressure from acidity/nutrient problems. It was clear that use of electric exclusion fencing did not guarantee acceptable hardwood regeneration in terms of relative density and frequency of seedlings or diversity of woody stems. Furthermore, significant positive relationships were found between species richness and soil Ca levels, organic horizon pH, and Ca:Al ratios in the mineral subsoil. A significant negative relationship was found between species richness and exchangeable Al levels in the subsoil.

4. The combined impact of the predisposing stress of extremely acidic soil and the contributing stress of herbivory by mammals (primarily deer), may be too severe for regeneration success on some sites. This indicates that soil chemical evaluation, and possibly remediation through liming and fertilization, should be added to the management equation when formulating forest management strategies on sites with extremely acidic soils, even when deer exclusion fencing is to be utilized.

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VARIABILITY IN OAK FOREST HERB LAYER COMMUNITIES

J. R. McClenahen and R. P. Long[1]

Abstract: This study evaluates forest herb-layer sensitivity to annual-scale environmental fluctuation. Specific objectives were to determine the between-year variation in herb-layer community biomass, and to contrast and evaluate the temporal stability of spatial relationships in herb-layer community structure and composition between successive years. Aboveground dry weights of herbs and of woody species <2.5 cm dbh were estimated in June of 1988 and 1989 in 13 ecologically analogous, oak-dominated stands along a 170-km east-west transect in north-central Pennsylvania. An approximately 50% reduction in total herb-layer biomass was measured from 1988 to 1989, presumably due to a residual or lagged effect of the early summer drought in 1988. Herb biomass reductions were inferred to largely result from reduced plant size rather than decreased spatial distribution. Smaller and (or) less-consistent biomass reductions for woody species may have resulted from high first-year seedling mortality in the case of *Acer rubrum* L., and possibly from intensified deer browsing of shrubs and saplings as a consequence of reduced herb forage resulting from the drought. Classification and ordination also indicated a 1989 reduction in importance of mesic-type species and an increase in similarity of herb-layer communities among stands. Conversely, greater spatial differences in species diversity measures were evident in 1989. Comparisons among herb layer communities based on single or infrequent measurements may be misleading due to large temporal variability and the spatially differential responses that broad-scale environmental factors such as climate may superimpose. Monitoring herb layer biomass in this oak forest ecosystem offers a sensitive means for detecting short-term environmental fluctuations such as annual climate variation, and additional data may enable identification of vegetation trends with statistical confidence.

INTRODUCTION

Eastern North American deciduous forests receive a multitude of natural and anthropogenic stresses, including climatic extremes, air pollution, and disease and insect outbreaks. These stresses can eventually alter ecosystem composition, structure and function (Graham et al., 1990; Hinrichsen, 1987). A national research priority is to identify and monitor sensitive forest ecosystem indicators that portend such changes.

The forest herb-layer can be a sensitive indicator of microsite (Pregitzer and Barnes, 1982), and is useful for classifying forest ecosystem types (Host and Pregitzer, 1991; Hix, 1988). An altered understory community is theorized to be an early response of forested landscapes to global change (Davis, 1989). However, most long-term studies involve measurement intervals of several years to decades (Rodenkirchen, 1992; Mueggler, 1992), thus masking any effects of climate or other high-frequency environmental fluctuation.

Few annual-scale studies of forest understory have been published. A six-year study of vegetation cover and frequency in a Pinyon-Juniper community demonstrated the high variability to be expected among both perennial and annual herbs, with little fluctuation among shrubs and trees (Treshow and Allan, 1979). It thus appears that spatial and temporal monitoring of herb-layer community dynamics may yield sensitive bioindication of both environmental fluctuations and trends. Further, with appropriate measurements of forest overstory, site, climate, atmospheric pollution and other intervening factors, the influence of these environmental variables can also be assessed.

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This paper focuses on evaluating the spatial and temporal variability of herb-layer biomass in mixed oak forests. Specific objectives were: (1) to determine the between-year variation in herb-layer community biomass and, (2) to contrast and evaluate the temporal stability of spatial relationships in herb-layer community biomass between successive years.

Background

Ecological monitoring can be a useful approach for investigating long-term processes such as effects of climate change and atmospheric pollution (Likens, 1989). Substitution of space for time is one technique employed to investigate long-term processes on shorter time scales (Pickett, 1989). We used a spatial gradient of wet sulfate and nitrate deposition across north central Pennsylvania to capitalize on an existing range of atmospheric inputs as a substitute of space for time (Likens, 1989). Data reported here were collected as part of the broader study of oak forest condition and response along this gradient (Nash et al., 1992).

Ecologically similar sites are often used to reduce experimental variation, but in studies such as ours where space is substituted for time, different site conditions or histories can lead to erroneous conclusions (Pickett, 1989). Therefore, initial emphasis was placed on selecting ecologically analogous sites. The selection process included ordination and classification analyses of candidate sites using preliminary data on soil physical properties, physiography, canopy stratum species abundances, and evaluation of disturbance history as determined from insect defoliation surveys, forest management records, and on-site evidence of fire and cutting (Long et al, 1991). Our rationale was to confine selection criteria to factors unlikely to have been influenced by chemical deposition. Subsequent intensive sampling of herb-layer biomass on the selected sites in 1988 and 1989 form the basis for this report.

METHODS

Study Area

The study region lies within the Appalachian Plateaus Physiographic Province (Ciolkosz et al., 1983). Soils of the study sites are derived from residual sandstone and are mostly Typic Dystrochrepts with small areas of Ultisols and Alfisols. Thirteen ecologically analogous sites (6-50 ha) were selected within four approximately equally-spaced core areas (clustered sites) across the 170-km gradient (Long et al., 1991). The core areas, from west to east and corresponding to the state forests in which they are located, are referred to as Clear Creek (C), Moshannon (M), Sproul (S) and Tiadaghton (T). The 13 sites occupied gently sloping, mostly broad uplands at a mean elevation of 576 m (range: 490-661).

Forest canopy trees on the sites were 70-80 yr old, even-aged, and dominated by *Quercus rubra* L). Common associated canopy species were *Acer rubrum* L., *Quercus alba* L., *Quercus prinus* L., and *Betula lenta* L.

Basal areas for canopy trees on the 13 sites averaged 25.1 (19.7-28.9) m^2/ha, with a mean overstory density of 291 (243-334) stems/ha. *Sassafras albidum* (Nutt.) Nees, *Hamamelis virginiana* L., *Amelanchier* spp, *Cornus florida* L., and in the Tiadaghton core area, *Acer pensylvanicum* L., were common understory trees.

Sampling

Herb-layer above-ground biomass was destructively sampled on the 13 sites in 1988 and 1989. Pairs of 3.1 m^2 (1-m-radius) plots were located 14 m north and south (1988) or east and west (1989) of permanent, randomly-established reference stakes. Depending on area of the site, 10 or 20 plots per site were sampled, for a total of 160 plots each year.

Sampling was conducted after the initial growth flush by clipping at the ground surface all herbaceous plants and all woody stems <2.5 cm dbh. In 1988 sampling was completed between June 6 and June 23, and in 1989 between June

5 and June 16. Clipped material was placed in paper bags by species. In the laboratory, samples were temporarily refrigerated, dried at 70° C to constant weight, and weighed to 0.01 g. A voucher collection was prepared. Species identification of *Carex*, *Viola* and other difficult genera was verified by consultation with personnel at The Ohio State University Herbarium.

Data Analysis

A split-plot ANOVA fixed effects model was one method used to evaluate spatial and temporal differences in herb layer biomass. Stands were tested as spatial (main) effects, and years and the years by species groups interaction terms were subunits used to test for temporal differences and possible interactions between species and years (Neter et al., 1985). Means of factors with significant F-values ($P \leq .05$) were separated by Student-Newman-Keuls' test. Species were combined into groups to avoid non-normality problems resulting from high frequencies of zero's for individual species. The groupings were based mostly on species abundances and taxonomic relationship. One analysis tested two groups consisting of total woody biomass and total herbaceous biomass. A second analysis used the following nine species groups: (1) *Dennstaedtia punctilobula*, (2) other fern species, (3) *Viola* spp., (4) *Gramineae* and *Carex* spp., (5) other herbaceous species, (6) *Vaccinium* spp. and *Gaylussacia* spp., (7) other *Ericaceae*, (8) other shrubs and, (9) tree seedlings. *Kalmia latifolia* was omitted from the second analysis due to its extreme variability in biomass, a result of the wide range in plant size and clumped spatial distribution.

The herb-layer communities were compared on the basis of species biomass by classification and ordination techniques to examine geographical and temporal trends. Plots were classified using Hill's (1979a) TWINSPAN algorithm, and ordinated by detrended correspondence analysis (DCA) using computer program DECORANA (Hill, 1979b).

Species diversity was compared geographically (among core areas) on the basis of Hill's (1973) diversity numbers: N0, N, and N2 (Ludwig and Reynolds, 1988). N0 is the number of species. N1 is calculated as $e^{H'}$, where e is the base of natural logarithms and H' is Shannon's diversity index, yielding the biomass-weighted or *effective* number of relatively well-represented species. N2 is $1/\lambda$, where λ is Simpson's diversity index, which provides a biomass-weighted *effective* number of relatively dominant species. The modified Hill's ratio (Alatalo, 1981), a measure of evenness (E), was calculated for core areas as $(N2 - 1)/(N1 - 1)$. Thus, E achieves a maximum of 1.0 when all species are equally abundant, whereas values of E approaching the minimum of 0 indicate increasing dominance by a single species.

Biomass distribution among species was also compared between years and among core areas on the basis of dominance-diversity curves (Bonham, 1989).

RESULTS

Biomass for each species sampled on plots is presented in Table 1. Analysis of variance detected no significant differences in biomass among the 13 sites ($P \leq .05$) on the basis of the nine species groups; nor were site differences found when total woody and total herbaceous biomass groupings were used.

However, there was a significant difference in biomass between years when the nine species groups were used in ANOVA, with mean biomass decreasing nearly 50% from 1988 (283.5 kg/ha) to 1989 (151.1 kg/ha). Between-year biomass reductions were consistent for the nine species groups. The relative decreases were close to 50% (40.4% - 56.6%) for all four herbaceous groups and the miscellaneous shrubs, while the smallest decrease (12.1%) occurred within the tree seedling group. A significant species group by year interaction resulted from the comparatively large biomass reduction of *Dennstaedtia punctilobula*. For other individual species, primarily those with low plot frequencies (e.g., *Dryopteris spinulosa*, *Podophyllum peltatum*, *Vaccinium vacillans*; Fig. 1) between-year biomass differences were less consistent (Table 1).

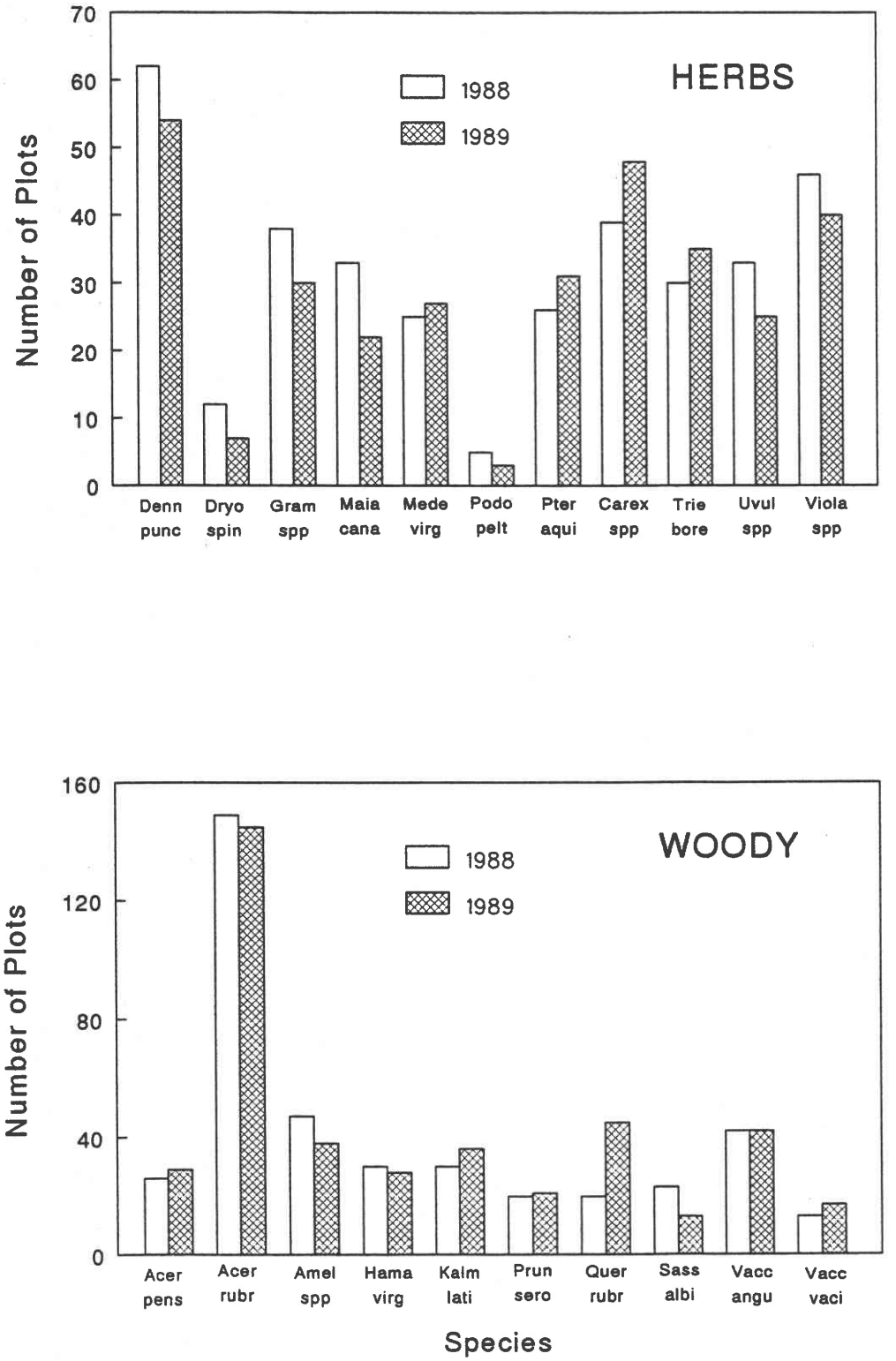

Figure 1. Plot frequencies of selected herbaceous and woody species. Complete species names are found in Table 1.

Table 1. Mean herb-layer species aboveground dry matter by core area and year. Core areas are Clear Creek (C; n=40), Moshannon (M; n=50), Sproul (S; n=40), and Tiadaghton (T; n=30).

Species	Year	Core Area			
		C	M	S	T
		-kg/ha-			
Acer pensylvanicum L.	1988	-	-	-	340.5
	1989	-	2.25	0.06	193.9
Acer rubrum L.	1988	1.84	9.94	5.66	1.91
	1989	1.07	7.24	0.86	1.95
Amelanchier spp.	1988	1.98	1.40	0.66	0.84
	1989	0.38	2.28	0.47	1.64
Aralia nudicaulis L.	1988	0.19	-	-	-
	1989	0.59	1.14	-	2.01
Aster spp.	1988	-	-	-	-
	1989	0.13	-	0.04	-
Azalea spp.	1988	5.14	0.25	-	-
	1989	0.73	0.57	-	-
Betula lenta L.	1988	-	2.88	0.11	0.10
	1989	-	2.73	0.10	2.01
Brachyeletrum erectum (Schreb.) Beauv.	1988	4.78	1.92	2.23	-
	1989	-	-	-	-
Carex debilis Michx.	1988	-	42.1	-	-
	1989	0.44	1.14	-	-
C. digitalis Willd.	1988	4.78	0.19	5.76	1.40
	1989	2.50	-	1.38	2.26
C. pensylvanica Lam.	1988	47.2	7.27	25.2	15.3
	1989	10.6	8.66	17.3	-
C. swanii (Fern.) MacKenz.	1988	-	-	0.03	5.82
	1989	-	-	-	-
Carex spp.	1988	5.14	1.54	-	3.85
	1988	0.06	-	-	4.31
Carya spp.	1988	-	-	-	0.38
	1989	-	-	-	0.01
Conopholis americana (L.) Wallr.	1988	23.0	-	-	-
	1989	-	-	-	-

Table 1 (continued)

Species	Year	C	M	S	T
			-kg/ha-		
Cornus florida L.	1988	-	0.41	-	-
	1989	-	-	-	-
Craetegus spp.	1988	1.46	-	0.35	-
Danthonia compressa Aust.	1988	-	41.0	55.1	-
	1989	-	3.44	3.08	-
Dennstaedtia punctilobula (Michx.) Moore	1988	112.2	200.6	239.0	504.8
	1989	58.3	67.3	89.1	398.2
Dryopteris spinulosa (O.F. Muell.) Watt	1988	5.15	6.22	0.16	28.1
	1989	1.61	12.5	0.13	7.64
Euonymous spp.	1988	-	-	-	0.16
	1989	-	-	-	-
Eupatorium rugosum Houtt.	1988	-	-	-	1.56
	1989	-	-	-	-
Fagus grandifolia Ehrh.	1988	0.22	26.7	-	8.50
	1989	-	104.4	-	4.96
Fraxinus americana L.	1988	-	-	-	0.84
	1989	0.03	0.08	-	0.30
Galium spp.	1988	0.52	-	-	-
	1989	0.03	0.32	-	-
Gaultheria procumbens L.	1988	2.83	29.7	1.31	2.70
	1989	3.86	6.22	1.28	2.26
Gaylussacia spp.	1988	50.2	-	-	-
	1989	244.	-	-	-
Hamamelis virginiana L.	1988	10.9	1.68	0.13	83.1
	1989	6.70	1.38	0.32	9.32
Hepatica acutiloba DC.	1988	-	-	-	-
	1989	-	0.06	-	-
Hypericum spp.	1988	-	0.03	-	-
	1989	-	-	-	-
Ilex verticillata (L.) Gray	1988	2.29	0.13	-	-
	1989	-	1.02	0.32	-

Table 1 (continued)

Species	Year	C	M	S	T
			-kg/ha-		
Kalmia latifolia L.	1988	0.89	3690	2120	2394
	1989	110.6	65.0	1091	2993
Liriodendron tulipifera L.	1988	0.03	-	-	-
	1989	-	-	-	-
Lycopodium obscurum Var. *dendroideum*	1988	-	-	-	-
(Michx.) D.C. Eat.	1989	0.35	-	-	-
Lysimachia quadrifolia L.	1988	2.96	0.11	0.06	0.60
	1989	0.89	0.48	-	0.70
Magnolia acuminata L.	1988	-	-	-	-
	1989	-	2.16	-	-
Maianthemum canadense Desf.	1988	0.90	1.00	0.06	0.29
	1989	0.70	0.20	-	0.48
Medeola virginiana L.	1988	2.38	3.21	0.21	0.23
	1989	0.78	2.96	0.34	1.05
Mitchella repens L.	1988	2.59	4.68	-	-
	1989	0.40	3.50	0.67	1.75
Monotropa uniflora L.	1988	-	1.92	1.95	-
	1989	-	-	-	-
Nyssa sylvatica marsh.	1988	0.83	-	1.53	-
	1989	0.53	5.57	0.10	-
Onoclea sensibilis L.	1988	1.59	-	-	-
	1989	-	-	-	-
Oryzopsis asperifolia Michx.	1988	3.41	15.2	17.7	-
	1989	-	5.51	2.51	-
Panax trifolius L.	1988	-	-	0.32	-
	1989	-	-	-	-
Panicum spp.	1988	0.64	0.16	-	-
	1989	0.06	8.54	-	-
Phlox spp.	1988	-	-	-	-
	1989	-	-	0.03	0.37

Table 1 (continued)

Species	Year	C	M	S	T
			-kg/ha-		
Pinus strobus L.	1988	0.16	-	0.03	-
	1989	-	-	-	-
Poa spp.	1988	-	-	-	1.37
	1989	-	-	-	-
Podophyllum peltatum L.	1988	6.12	-	-	-
	1989	24.4	-	-	-
Polygala paucifolia Willd.	1988	-	-	-	-
	1989	-	0.13	0.16	-
Populus grandidentata Michx.	1988	2.20	-	-	-
	1989	-	-	-	-
Potentilla spp.	1988	1.81	-	-	-
	1989	-	-	-	-
Prenanthes sp.	1988	0.60	0.09	-	0.42
	1989	0.03	0.14	0.17	0.95
Prunus serotina Ehrh.	1988	0.67	0.40	-	0.40
	1989	0.30	0.21	-	4.22
Pteridium aquilinum (L.) Kuhn	1988	8.59	111.7	-	-
	1989	1.02	66.5	-	-
Quercus alba L.	1988	-	0.58	-	-
	1989	0.29	0.53	0.10	0.80
Q. coccinea Muenchh.	1988	1.81	-	-	-
	1989	-	-	-	-
Q. prinus L.	1988	1.29	1.20	-	1.69
	1989	-	-	-	13.8
Q. rubra L.	1988	0.54	0.54	1.02	1.09
	1989	0.80	1.74	0.93	0.55
Q. velutina Lam.	1988	0.83	0.40	0.22	0.68
	1989	-	-	-	-
Rubus spp.	1988	0.38	0.57	-	-
	1989	-	0.10	-	0.06
Sassafras albidum (Nutt.) Nees	1988	0.44	4.51	2.65	-
	1989	-	0.66	0.37	-

Table 1 (continued)

Species	Year	C	M	S	T
			-kg/ha-		
Smilacina racemosa (L.) Desf.	1988	-	-	-	-
	1989	-	-	0.89	-
Smilax herbacea L.	1988	-	-	-	-
	1989	2.43	-	-	-
S. rotundifolia L.	1988	0.60	0.92	3.47	-
	1989	0.38	0.29	0.13	-
Solidago spp.	1988	-	-	-	-
	1989	0.32	0.04	-	-
Streptopus spp.	1988	-	0.06	-	0.10
	1989	0.21	0.43	-	0.76
Thelypteris noveboracensis (L.) Nieuwl.	1988	-	41.2	2.58	-
	1989	-	7.25	-	-
T. palustris Schott	1988	-	62.3	-	-
	1989	0.13	-	-	-
Trientalis borealis Raf.	1988	2.05	0.75	-	1.27
	1989	1.15	1.49	-	1.40
Trillium undulatum Willd.	1988	-	0.65	-	0.51
	1989	0.37	0.41	0.03	0.60
Uvularia perfoliata L.	1988	0.22	1.05	-	-
	1989	0.70	-	0.06	1.46
U. sessilifolia L.	1988	1.25	0.80	0.14	0.11
	1989	0.70	0.97	-	1.72
Vaccinium angustifolium Ait.	1988	127.1	133.4	28.3	24.1
	1989	103.9	100.2	14.2	10.3
V. vacillans Torr.	1988	62.3	48.2	-	6.68
	1989	23.8	73.5	2.39	-
Viburnum acerifolium L.	1988	5.49	-	-	2.10
	1989	-	-	-	20.2
Viola hastata Michx.	1988	0.46	0.13	-	-
	1989	0.32	1.24	-	1.24
V. pallens (Banks) Brainerd	1988	19.2	0.03	-	-
	1989	-	-	-	-

Table 1 (continued)

Species	Year	C	M	S	T
			-kg/ha-		
V. rotundifolia Michx.	1988	4.06	-	-	0.28
	1989	-	-	-	-
V. spp.	1988	1.29	0.48	0.06	0.76
	1989	1.74	-	-	-
Vitis spp.	1988	0.04	0.08	0.06	8.12
	1989	0.05	0.20	-	0.12
Total Woody	1988	279.2	3924.	2164.	2875.
	1989	496.0	372.0	1111.	3257.
Total Herbaceous	1988	266.3	576.7	351.9	569.5
	1989	112.4	200.7	117.2	429.2
TOTAL	1988	545.5	4500.	2516.	3445.
	1989	608.4	572.7	1228.	3686.

Spatial Relationships

Classification. TWINSPAN analysis revealed considerable overlap in species composition among core areas (Fig. 2). With the notable exception of Tiadaghton, plots were not strongly segregated by core areas in either year. The most important indicator species in both years were *K. latifolia*, *D. punctilobula* and *A. rubrum*, with high relative abundances of *K. latifolia* and *A. pensylvanicum* distinguishing most Tiadaghton plots from those of other core areas. *A. rubrum* and *Vaccinium angustifolium* were additional consistent indicator species between years.

Other indicators in 1988 were mostly representative of mesic sites, including *Viola pallens*, *Amelanchier* spp., *Trientalis borealis*, *Maianthemum canadense*, *D. spinulosa* and *Brachyeletrum erectum*. In contrast, additional indicators in 1989, *Carex pensylvanica* and *Gaultheria procumbens*, were typical of dry or acidic sites, except for the unique occurrence of *Podophyllum peltatum* as an indicator on two plots (Fig. 2).

The shift between years from mesic to xeric indicator types and associated species (preferentials) was also evident at more refined divisions (not shown). Mesic herbs (e.g., *Viola* spp.) in the 1988 analysis tended to be replaced by xeric shrubs and herbs (e.g., *Vaccinium pallens*, *V. angustifolium*, *Pteridium aquilinum*, *G. procumbens*) in 1989. Moreover, compared to 1989, the 1988 analysis identified more indicator species, and revealed more detailed vegetational trends.

Ordination. The 1988 DCA ordination generally confirmed the considerable overlap of plots from Clear Creek, Moshannon, and Sproul, and the distinction of most Tiadaghton plots as indicated by TWINSPAN (Fig. 3). The 1989 ordination reflected a more homogeneous vegetation structure than did that for 1988, especially regarding the Tiadaghton samples (Fig. 3). The three outlying Moshannon plots seen in the 1988 and 1989 ordinations and classifications are due to the presence of *Fagus grandifolia* Ehrh.

The 1988 species ordinations (Fig. 4) suggest a weak moisture gradient on the first axis. The gradient extends from species mostly associated with well-drained soils (*A. pensylvanicum*, *F. grandifolia*, *Fraxinus americana* L., *Carex swanii* (Fern.) MacKenz. and *K. latifolia*) to those associated with more mesic sites (*Viola* spp., *P. peltatum*) and finally to species typical of very moist sites (*Thelypteris noveboracensis* (L.) Nieuwl. and *Thelypteris palustris* Schott).

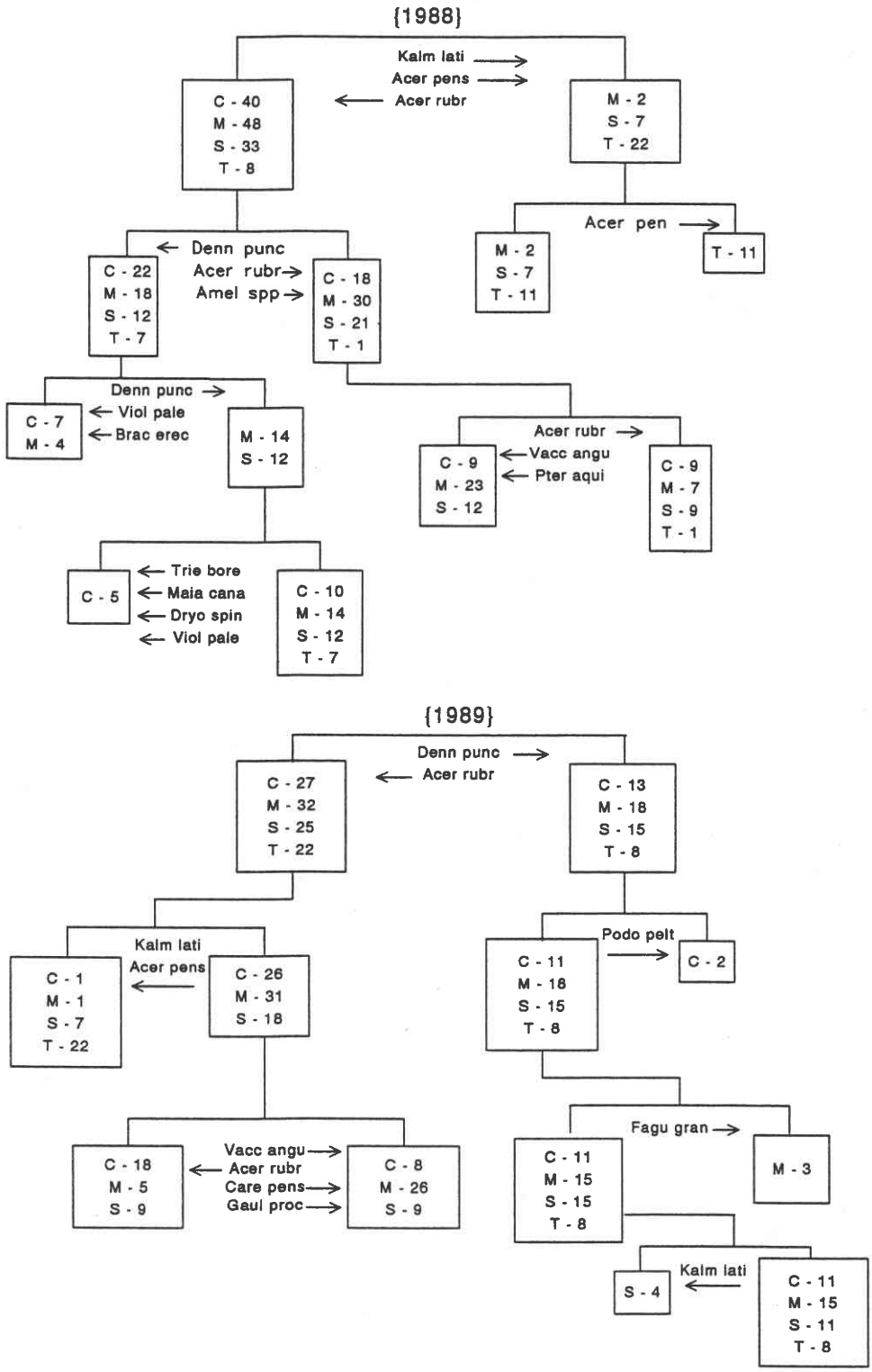

Figure 2. Hierarchal (TWINSPAN) classification of plots accoring to core areas in 1988 and 1989. Major indicator species and their directional indication are shown by arrows.

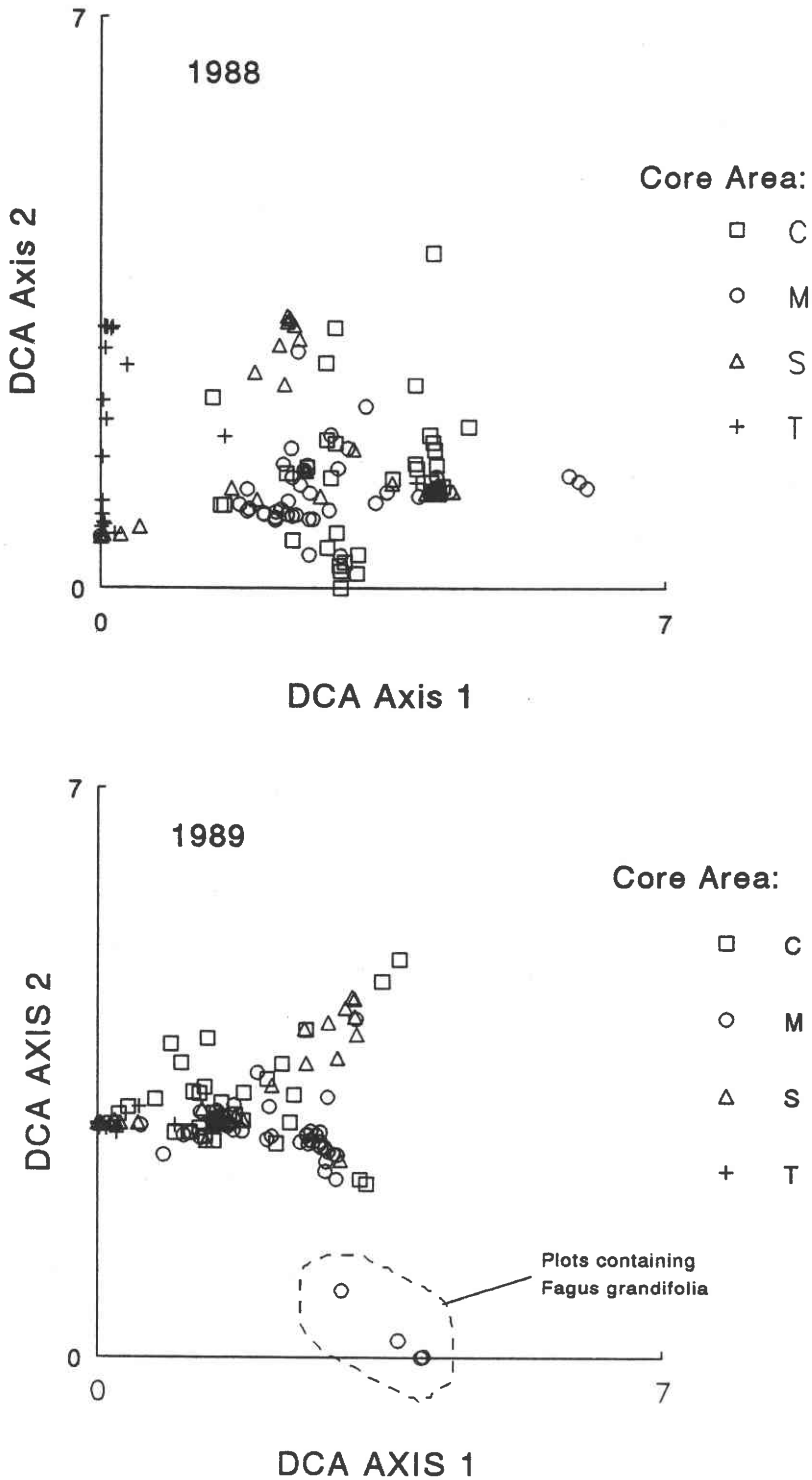

Figure 3. Detrended correspondence analysis (DCA) ordinations of plots in 1988 and 1989 in relation to core areas.

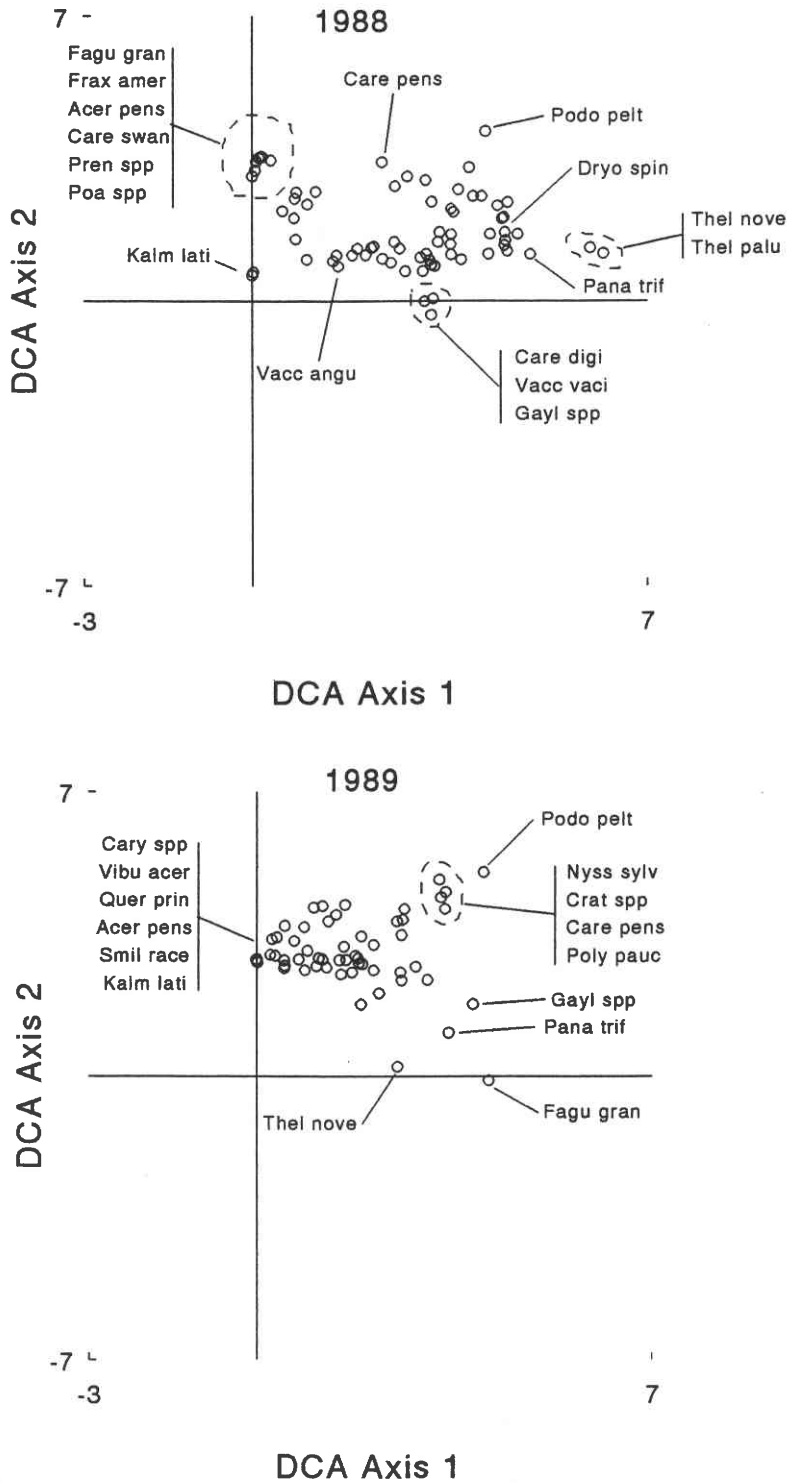

Figure 4. Detrended correspondence analysis (DCA) ordinations of species in 1988 and 1989 in relation to core areas. Species names are found in Table 1.

The second DCA axis revealed a slight trend from mostly ericaceous shrubs typical of infertile, acidic soils (e.g., *K. latifolia*, *Gaylussacia* spp., *Vaccinium* spp.) to more fertile, mesic site species such as *P. peltatum*, *C. swanii*, *Poa* spp., *A. pensylvanicum*, *F. grandifolia*, and *F. americana*. In 1989, these relationships were not evident and, as was the case for the plot ordinations, there was little differentiation except for a few outliers (Fig. 4).

Differences in herb-layer composition on the Tiadaghton core area suggested by classification and ordination are evident in the comparative species abundances (Table 1). Seven species were sampled almost exclusively at Tiadaghton in 1988, while four species sampled on most or all other core areas were absent at Tiadaghton. Particularly notable were the comparative abundances of *A. pensylvanicum* and *C. swanii* at Tiadaghton, and the absence of three species well-represented on other core areas: *S. albidum*, *Smilax rotundifolia* L., and *Oryzopsis asperifolia* Michx.

Species Diversity. On the basis of Hill's diversity numbers, the western pair of core areas (Clear Creek and Moshannon) had greater species richness (N0) and more abundant (N1) and very abundant (N2) species than the two eastern core areas in both years (Table 2). Conversely, evenness was similar among all four core areas in 1988; however, between 1988 and 1989 it increased at Clear Creek and Moshannon, but decreased at Sproul and Tiadaghton. This same temporally diverging pattern is evident for N1 and N2 between the western and eastern pairs of core areas. However, within the western and eastern pairs of core areas these diversity measures were consistently similar.

The dominance-diversity curves indicate a lognormal distribution typical of relatively complex, species-rich communities (Fig. 5). They clearly illustrate the distributional differences in species abundances between the western and eastern pairs of core areas noted above. A shift toward greater similarity in species distributions is also revealed between Clear Creek and Moshannon from 1988 to 1989, making each half of the gradient appear even more distinct in 1989. These spatial patterns reveal distinctly lower species diversity in the eastern half of the gradient.

DISCUSSION AND CONCLUSIONS

Lacking site-specific environmental monitoring data, reasons for the large between-year differences in herb-layer biomass cannot be definitely determined. However, a severe drought developed over the 1988 growing season. Palmer Drought Severity indices revealed mild drought conditions across the study area by the end of April (Figure 6). Drought conditions were rated moderate (-2.3 to -2.5) by mid-June, becoming severe (-3.3 to -3.5) in July. Despite increased rainfall in mid-July, drought conditions prevailed through August (Figure 6). There was little difference in drought severity across the study transect. In contrast, growing season precipitation in 1987 and 1989 was at or above normal.

We suggest that the 1988 drought could have produced a lagged decrease in herb layer production as measured in 1989. Such a lag might occur among perennial herbs with large belowground storage organs if initial growth flush is largely determined by stored reserves. For herbaceous plants, this decrease would likely have occurred primarily through a reduction in the size of individual plants and, for rhizomatous species, perhaps the numbers of stems as well. Both of these factors could have been operative in the case of *D. punctilobula*, the most abundant and widely distributed herbaceous species (Table 1, Fig. 1). The nearly 50% reduction in total biomass between years cannot be attributed entirely to the small reduction in distribution (Fig. 1). Mortality or decreased reproductivity probably contributed relatively little to the herbaceous biomass decrease, as suggested by the comparatively small reductions or even increases in plot frequencies. For example, there was a large between-year decrease in *Carex* spp. biomass (Table 1), yet plot frequency increased (Fig. 1). Thus, herb-layer biomass reductions between years are inferred to result more from reduced plant size rather than decreased spatial distribution.

The above hypothesis cannot, however, be directly invoked to explain the apparent 1989 woody biomass reduction. The large but non-significant decrease in *K. latifolia* biomass is evidently an artifact of extreme variability in plant size and patchy distribution. Seedlings of many tree species such as *Q. rubra*, *Amelanchier* spp., *H. virginiana*, and *Prunus serotina* Ehrh. also did not change perceptibly in frequency (Fig. 1), but were relatively uncommon and highly variable in individual plant size, all of which contributed to inconsistent between-year differences.

Table 2. Species diversity measures summarized for core areas and years.

Core Area	Year	n[1]	NO[2]	N1[2]	N2[2]	Evenness[3]
Clear Creek	1988	40	58	6.53	3.74	0.50
	1989	40	51	7.43	5.44	0.69
Moshannon	1988	50	55	4.95	3.24	0.57
	1989	50	49	7.65	5.28	0.63
Sproul	1988	40	31	1.83	1.41	0.49
	1989	40	33	1.66	1.31	0.47
Tiadaghton	1988	30	41	2.31	1.78	0.59
	1989	30	38	1.75	1.36	0.48

[1]Number of 3.1 m^2 plots
[2]Hill's diversity numbers (Hill, 1973); includes unidentified species.
[3]Modified Hill's ratio (Alatalo, 1981).

In contrast, *A. rubrum* seedling density can fluctuate markedly even within a growing season (McClenahen and Hutnik, 1979). This species was widely distributed (Fig. 1), but newly germinated seedlings predominated. Prolific seed production often results in large *A. rubrum* populations in late spring when the current seed crop is dispersed and germinates. Many of these seedlings disappear by autumn (McClenahen and Hutnik, 1979).

A. pensylvanicum was well-represented at Tiadaghton, but not elsewhere (Table 1), and significantly decreased by almost 50% in biomass between years. The most abundant and well-distributed shrub, *V. angustifolium*, also declined in biomass (by 27%) between years (Table 1). Plot frequency did not change (Fig. 1), but it is unclear from the data how this biomass reduction occurred. An interesting conjecture, however, is that deer browsing and drought may have interacted. Deer browsing has had a major, long-term impact on forest understory vegetation in northern Pennsylvania (Hough, 1965; Marquis, 1974). In order to compare deer browsing intensities on the study sites, we used the pellet group count method to estimate deer population densities for each site in 1992 (Eberhardt and Van Etten, 1956). Sampling was confined within the site boundaries, so deer densities reflect comparative deer use of the sites during the dormant season (October - April); they cannot be interpreted as actual population estimates. Densities averaged 9.4 deer per km^2 (range: 1.8 - 28.3). Except for one Sproul site, all density estimates were above levels believed to reduce understory richness and cover (D. deCalesta, personal communication).

Thus, intensive deer browsing has probably altered the herb-layer over the long term at virtually all sites and may have exacerbated the impact of other stresses on existing vegetation. As a result of drought-reduced herbage by early summer of 1988, deer may have more intensively browsed woody vegetation. Woody species most clearly reduced in biomass were *V. angustifolium* and, at Tiadaghton, *A. pensylvanicum*. These two species are utilized by deer (Marquis, 1974; Bramble and Goddard, 1953), and may have been more heavily browsed if herbaceous forage was less available. The potential for interactions between deer browsing and other environmental stresses to impact forest understory community dynamics is poorly understood and deserves further invesigation.

Classification and ordination revealed no clear geographical separation of herb layer communities in either year. Although these analyses indicated that Tiadaghton communities were less like those in other core areas, this was only evident in 1988. The relatively different vegetation at Tiadaghton was also detected by similar analyses of the overstory community (and site factors) during site selection (Long et al., 1991). This tends to validate our original approach that focused on the dominant vegetative stratum along with disturbance history, physiography and soil physical properties as a sufficient and practical basis for choosing ecological analogs.

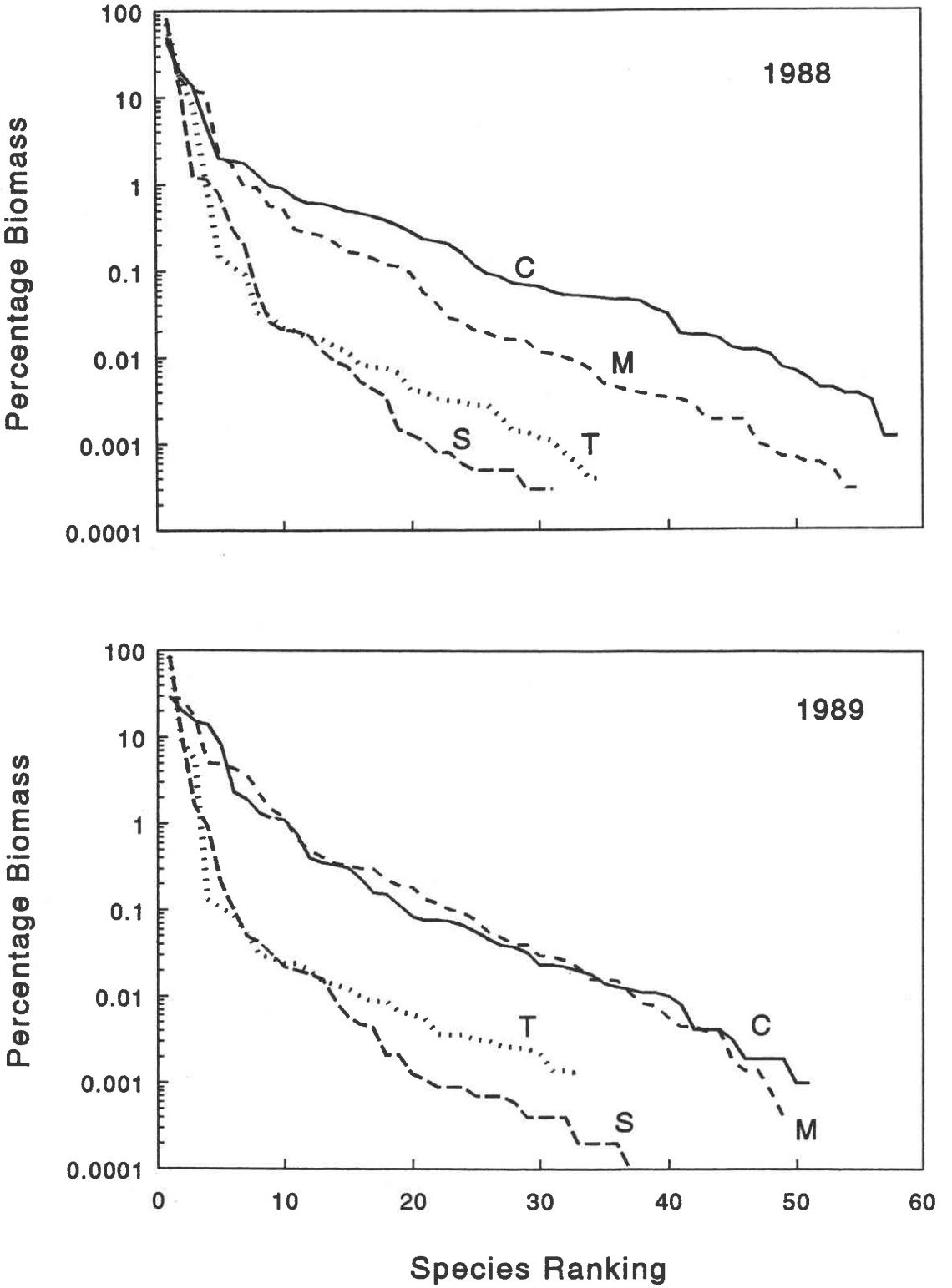

Figure 5. Dominance-diversity curves for core areas in 1988 and 1989.

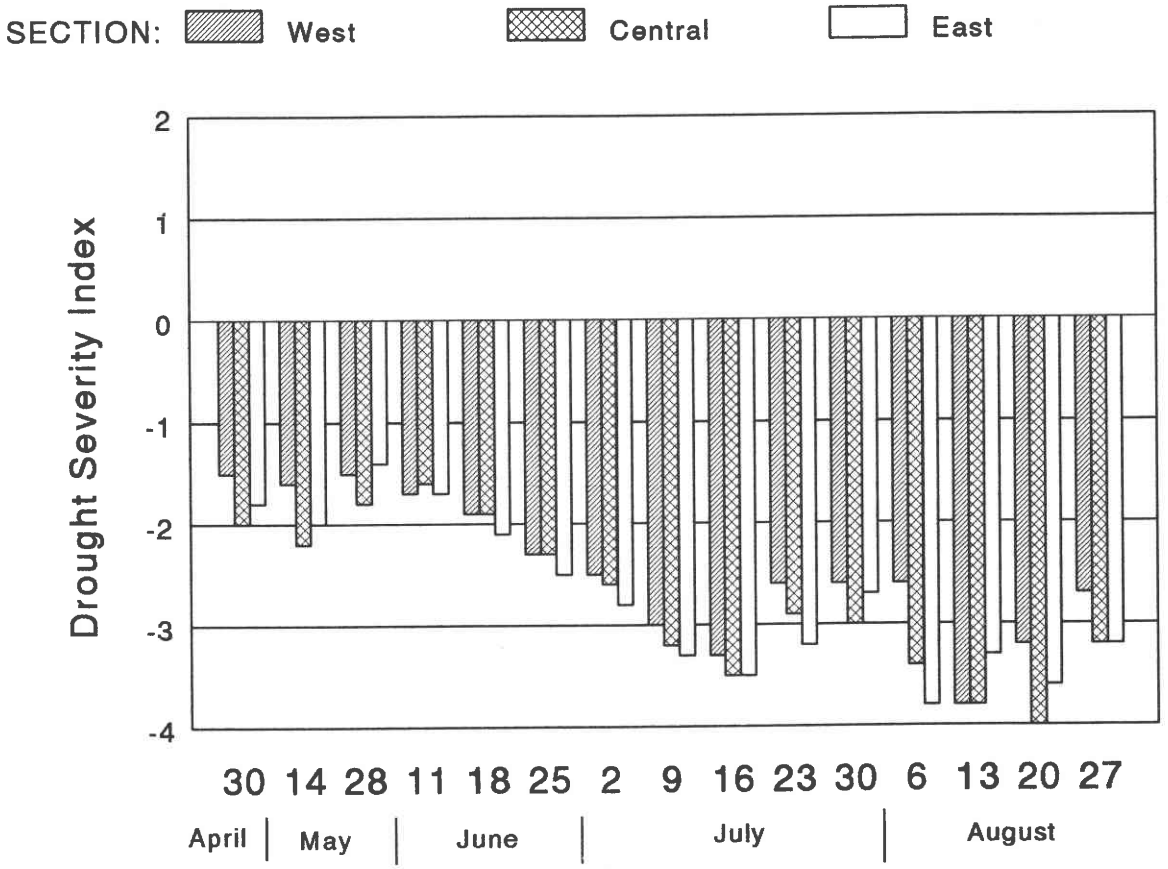

Figure 6. Palmer Drought Severity Indices during the 1988 growing season for climatic divisions corresponding to western, central and eastern sections of the study area (National Weather Service, 1988).

Two important temporal differences detected by classification and ordination were a reduction in mesic site indicator species and an increase in apparent vegetative similarity, particularly regarding Tiadaghton. Several investigators found inherent resistance to changes in spatial distribution among certain of the same species within herb-layer communities similar to ours (Hughes and Fahey, 1991; Collins and Pickett, 1988). Distributions of these species, especially the "stress-tolerators" (Grime, 1979), are hypothesized to develop under favorable conditions, and are maintained by a capacity to tolerate subsequent environmental stresses (Hughes and Fahey, 1991). Our results suggest that, despite any such resistance to broad-scale (successional) changes, this forest herb community can exhibit large annual-scale biomass variation, presumably in response to environmental fluctuation such as the 1988 drought.

Whatever the cause(s) of the annual biomass variation, mesic species were reduced in biomass to the extent that they were less useful for classifying (and ordinating) plots in 1989. The 1989 analyses were therefore less revealing of subtle variations in herb-layer community structure among these relatively analogous sites. This underscores the importance of considering the impact of recent climatic or other environmental perturbations when using the herb stratum to detect experimental effects or to compare sites at a single point in time. Also revealed is the capacity for large annual variability of this herb layer community.

Monitoring herb layer biomass in this oak forest ecosystem offers a sensitive means for detecting short-term environmental fluctuations such as annual climate variation. This short-term variability could have a large effect on interpretations of experiments that are based on single or infrequent measurements. Conversely, annual monitoring will eventually yield estimates of the short-term variability that can potentially be used to identify vegetation trends with statistical confidence.

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CARBON AND NITROGEN POOLS IN OAK-HICKORY FORESTS OF VARYING PRODUCTIVITY

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Abstract: Carbon (C) and nitrogen (N) storage capacities are critical issues facing forest ecosystem management in the face of potential global climate change. The amount of C sequestered by forest ecosystems can be a significant sink for increasing atmospheric CO_2 levels. N availability can interact with other environmental factors such as water availability or temperature to control potential forest productivity. This in turn, may determine the amount of C that can be sequestered in forest ecosystems.

This study was designed to quantify C and N pools in oak-hickory dominated forest stands of varying productivity. Sites selected were second-growth upland oak forests with site indices ranging from approximately 55 to 90 (base age of 50 years of white oak). This wide range of potential site productivities was selected to determine if patterns of C and N storage vary with changing site productivity. Carbon and N pools were determined for the following components: living aboveground biomass, root biomass, annual litterfall, forest floor layers, and the mineral soil. Results indicate that total C and N storage increases as site productivity increases, but that the relative importance of these components may vary with changes in site productivity. In addition, the annual nutrient inputs from the various pools may vary directly with changes in site productivity.

INTRODUCTION

Under changing environmental conditions, forest ecosystems become increasingly more important as both sources and sinks for nutrients. In particular, the role of forest ecosystems to serve as potential sinks for increasing atmospheric CO_2 concentrations has become increasingly more important to develop global carbon budgets. There has also been concern about the possible impacts of increasing atmospheric nitrogen deposition in forest ecosystems and the ability of forest ecosystems to adapt to potential increasing nitrogen inputs in highly industrialized regions. Conversely, as utilization and management of forest resources continue to increase, questions have arisen concerning the long-term sustainability of these practices. In particular, as intensive harvesting becomes more widely practiced, will these forest systems continue to be self-sustaining or will nutrient limitations become apparent? If the potential impacts of environmental changes and management activities are to be determined, the carbon and nitrogen storage characteristics of forest ecosystems under current conditions must be determined.

Most previous studies examining nutrient storage patterns in forest ecosystems have shown that the nutrient storage potential of the living biomass and the soil at a site are often high and that in the long-term, these pools may contain the majority of nutrients within a forest stand. In mature undisturbed forest stands, the short-term availability of nutrients within a site is determined by the annual inputs of nutrients via litterfall, fine root turnover, mineral weathering, symbiotic nitrogen fixation, and atmospheric inputs. In undisturbed old-growth forest stands, the addition and turnover of nutrients through the mortality and decay of senescent trees can represent a highly significant input of nutrients into these systems. However, in younger, aggrading forest ecosystems, the addition of organic matter and nutrients through tree death is usually a relatively minor component of inputs. Following large scale disturbance such as timber harvesting, extensive fires, or mortality caused by insects or disease, the potential nutrients

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returned to the site from the aboveground biomass can be significant in young stands. However, in young forest stands lacking catastrophic disturbance events, the yearly inputs of aboveground leaf and litter fall and belowground root production and turnover will be significant in determining the amount of nutrients available to the forest stand.

There have been a host of studies that have examined the carbon and nitrogen storage patterns of different forest ecosystems, but few studies have determined if nutrient storage patterns differ within a single ecosystem distributed along a productivity gradient. In this context, this study was designed to determine the carbon and nitrogen storage patterns of oak-hickory dominated forests distributed along a productivity gradient. The study concentrated on quantifying and determining the relative importance of pools in these forest stands. Greatest emphasis was placed on those pools that undergo relatively rapid turnover in these second-growth forest stands; therefore, sampling intensity was greatest for annual leaf-fall, forest floor organic layers, and fine root components.

METHODS

Five study sites distributed throughout the state of Indiana were selected for this study. All study locations were in second-growth oak-hickory dominated forests having overstory trees ranging from 80-120 years of age. The oak-hickory species group comprised at least 55% of the basal area in each stand, but associated species differed among the various sites. Sites were chosen that represented a broad range of potential site productivities ranging from xeric, nutrient poor to mesic, nutrient rich sites. On the two more xeric sites, the oak-hickory group comprised 95% of the total stand basal area, while on the more mesic sites, the oak-hickory group dominance decreased to 55-70% and mesophytic species increased in importance. Species composition, basal area by species, and relative dominance for each species is listed in Table 1. The study sites, all located in the state of Indiana, are described in order of productivity from highest to lowest productivity.

(1) Nelson-Stokes (NS) is located in Putnam County in the Entrenched Valley Section of the Central Till Plain Natural Region (Homoya and others 1985). The Russell silt-loam soil (Fine-silty, mixed, mesic Typic Hapludalf) developed from moderately thick loess deposits overlying glacial till. The overstory vegetation is dominated by white oak, *Quercus alba*, northern red oak, *Quercus rubra*, assorted hickories, *Carya* spp., and has a subcanopy dominated by *Acer saccharum*, sugar maple. The site index for oak is approximately 85-90 feet at a base-age of 50 years.

(2) Feldun Purdue Agricultural Center (Feldun) is located in Lawrence County in the Mitchell Karst Plain Section of the Highland Rim Natural Region (Homoya and others 1985). The soil is a Caneyville silt-loam (Fine, mixed, mesic Typic Hapludalf) developed from a thin layer of loess covering an older paleosol. The site is underlain by limestone bedrock within 60 cm of the surface of the soil. The vegetation is dominated by black oak, *Quercus velutina*, northern red oak, tulip poplar, *Liriodendron tulipifera*, and beech, *Fagus grandifolia*. The site index for oak is approximately 75-80 and the site index for tulip poplar is approximately 95.

(3) Southern Indiana Purdue Agricultural Center (SIPAC) is located in DuBois County in the Crawford Upland Section of the Shawnee Hills Natural Region (Homoya and others 1985). The soil is a Wellston silt-loam (Fine-silty, mixed, mesic Ultic Hapludalf) derived from moderately thick loess deposits over paleosols derived from weathered sandstone and siltstone. The overstory vegetation is dominated by white oak, hickory, and northern red oak with the subcanopy dominated by sugar maple. The site index for white oak is approximately 65-70.

(4) Clark State Forest is located in Clark County in the Knobstone Escarpment Section of the Highland Rim Natural Region (Homoya and others 1985). The soil is Rarden silty-clay loam (Fine, mixed, mesic Aquultic Hapludalf) derived from a very thin layer of loess over a subsoil derived from weathered shale. The heavy clay texture of the subsoil restricts water infiltration through the profile during the winter and early spring, but also has a low plant available water holding capacity when water becomes limiting during summer and early fall. During the growing season, soil water potentials typically become very low. The overstory vegetation is comprised primarily of white and black oak. The site index for oak on this xeric, nutrient poor site is approximately 55-60.

Table 1. Species composition of the five study stands. The first row under each species gives its basal area (m^2/ha) and the second row under each species gives its relative dominance in each respective stand (%).

	Site				
Site Index	Nelson-Stokes 85-90	Feldun 75-80	SIPAC 65-70	Clark 55-60	Jasper-Pulaski 55-60
Species					
White Oak	20.6	1.7	22.0	16.0	15.2
	56.7	4.8	68.0	77.5	67.1
Black Oak	0.0	11.9	0.2	2.7	6.4
	0.0	33.9	0.6	12.9	28.4
Northern Red Oak	3.5	3.1	2.4	0.0	0.0
	9.6	8.9	7.4	0.0	0.0
Hickory Spp.	4.3	3.5	3.4	1.1	0.0
	11.9	10.1	10.6	5.2	0.0
Sugar Maple	4.6	2.3	4.0	0.1	0.0
	12.7	6.6	12.3	0.4	0.0
Beech	0.3	3.7	0.0	0.0	0.0
	0.9	10.4	0.1	0.0	0.0
Yellow Poplar	0.5	7.5	0.0	0.0	0.0
	1.3	21.4	0.0	0.0	0.0
Misc.	2.5	1.3	0.3	0.7	1.0
	6.9	3.9	1.0	4.0	4.5
Total Basal Area	36.3	35.0	32.3	20.6	22.6

(5) Jasper-Pulaski is located at the Jasper-Pulaski Fish and Wildlife Area in Jasper County. This area is in the Kankakee Sand Section of the Grand Prairie Natural Region (Homoya and others 1985). This soil is a Plainfield fine sand (Mixed, mesic Typic Udipsamment). The site is located in an area of outwash sand reworked into dunes. The vegetation on this xeric site is composed almost entirely of white and black oak. The site index for white and black oak on this site is approximately 55-60.

At each site, three replicated blocks were established at random locations within the stand. Woody vegetation was sampled at each site using 5, 1/20 ha circular plots within each block. All woody vegetation with a diameter at breast height (DBH) of 5 cm or greater was identified and its DBH recorded. Vegetation less than 5 cm in diameter at

breast height was not measured. The composite information from these 15 plots at each stand was used to determine the basal area/ha (m^2) and the relative dominance of each species.

Standing aboveground biomass for living trees was determined using equations developed by Hahn and Hansen (1991) and Smith (1985). These equations utilize species specific modified Wiebull volume regressions to determine the volume of the tree. The calculated volume is a function of the diameter of the tree and the site index for that species on the site. Once the volume for each individual tree is determined, this volume is converted to the oven dry weight of the standing aboveground biomass (Smith 1985).

Root biomass was determined by collecting 24 soil cores each measuring 6.3 cm in diameter and 30 cm in length from each stand. All fine root sampling was conducted in August and September. Previous studies utilizing soil pits at each of these sites demonstrated that the vast majority of fine and coarser roots were restricted to the upper soil horizons, therefore, root sampling was restricted to the upper 30 cm of the soil profile. Roots were separated from the soil in the cores by wet seiving on a 120 micron sieve. The roots were then separated into three size classes: fine (0-3 mm diameter), medium (3-10 mm diameter), and coarse (greater than 10 mm diameter). Root samples were then oven dried at a temperature of 65° C for 72 hours to obtain oven-dried weights. Values for each size class were converted to a kg/ha basis using appropriate conversion factors.

Biomass, nitrogen (N), and carbon (C) contained in the forest floor organic layers were determined from 12 randomly located cores from each forest stand. Each core measured 1/10 m^2. Each stand was sampled in the autumn after approximately 95% of leaf-fall had occurred. This protocol was followed to assure that accurate assessments of forest floor biomass and to minimize microbial decomposition of the litter material. Sampling began in the northern part of the state and progressed southward to follow natural leaf-fall patterns. Each core was separated to obtain litter, fermentation, humus layers, and A horizon mineral soil samples. Corrections for mineral soil contamination of the forest floor organic layers were made according to the method of Blair (1988) and Rustad (1994). More detail of the procedure used to make these corrections is available in Rodkey and others (1994). Subsamples of each layer were oven-dried at a temperature of 65° C and these values were converted to a dry weight kg/ha basis. Soil samples for soil horizons below the A horizon were obtained by excavating three soil pits at random locations within each site. Soils were fully described and samples collected from each respective horizon. Annual leaf-fall was determined by placing 15, 1/2 m^2 littertraps at each site. During autumnal leaf-fall, the litter from the traps was collected at approximately biweekly intervals, air dried, and subsamples oven-dried at 65°C to a constant weight to convert all biomass figures to an oven-dried basis.

Nutrient Analysis

Carbon and nitrogen concentrations in annual leaf-fall, standing aboveground biomass, root tissues, and soil samples were determined on a Leco 500 CHN elemental analyzer. Due to the large number of samples, several protocols were developed. The most intensive sampling was reserved for pools which were expected to show the greatest variability between sites and represent frequent turnover rates. Fine roots, forest floor organic layers, and annual litterfall were sampled most intensely. Several assumptions were made to determine carbon and nitrogen storage in aboveground biomass. First, it was assumed that the greatest differences in nutrient concentrations would be found in smaller diameter branches and limbs. Samples of these materials were taken from the dominant species at each site. It was further assumed that tree bole carbon and nitrogen concentrations would remain constant across sites. Inherent in this sampling scheme is some degree of uncertainty. Wood N concentrations will vary from species to species. Johnson and others (1982) found that wood bole nitrogen concentrations could vary as much as 20% between species. Therefore, our sampling likely has this degree of uncertainty in its estimate of aboveground N storage. Wood samples were taken from one site and these values used across sites. Sampling of woody boles and limbs was conducted in mid February to early March well before bud expansion began. Medium diameter roots (3-10 mm in diameter) were assumed to have carbon and nitrogen concentrations equal to branch samples, and coarse diameter roots (> 10 mm diameter) were assumed to have carbon and nitrogen concentrations equal to those found in woody boles.

Samples of forest floor organic layers were also analyzed for nitrogen and carbon concentrations using the Leco 500.

Following carbon analysis, these same samples were dry-ashed in a muffle-furnace at a temperature of 450° C for 4.5 hours. From this information (known carbon concentrations from analysis on the CHN analyzer and mass loss from dry ashing), regression equations were developed to accurately predict carbon concentrations by mass loss due to dry-ashing. R^2 values for these regressions ranged from 0.97-0.99. Nitrogen concentrations in these samples were determined by digesting 100 mg of tissue in 5 ml of concentrated H_2SO_4 in 50 ml Folin-Wu tubes. A mixture of K_2SO_4, $CuSO_4$, and Se in the ratio of 100:10:1 served as a catalyst for the digestion (Nelson and Sommers 1973). Following digestion, the samples were steam distilled, the distillate captured in a mixed boric acid indicator solution and back-titrated using 0.01 N HCl to determine the nitrogen concentration in the tissue.

RESULTS

Basal areas (m^2/ha) increased with increasing productivity and ranged from 21 to 36 m^2/ha (Table 1). There was also a slight change in species composition with changes in site characteristics. On the two most xeric sites, JP and Clark, the oak-hickory group made up at least 95% of the total stand basal area. The more mesic sites, while still dominated by the oak-hickory group, had a significant component of more mesophytic species such as sugar maple, tulip poplar, and beech.

The standing aboveground biomass varied widely among the five sites examined and was directly related to the site index. Standing aboveground biomass ranged from 124,000 kg/ha on low quality sites to over 245,000 kg/ha on highly productive sites (Table 2). This two-fold difference in standing biomass was due to greater stem density, greater average diameter, and greater average height on the more productive sites. The exclusion of small diameter (< 5 cm) stems from the sampling will have a minor effect on total aboveground biomass. Across the five sites, the number of 5.0-6.4 cm diameter stems ranged from 79-225/ha. These stems would contribute from 1300 to 3800 kg/ha and these stems are accounted for in our sampling methods. Stems less than 5 cm in diameter contribute little to aboveground biomass. Even if there are 1000 stems/ha less than 5 cm in diameter present in these stands, stems of this diameter class would contribute 1000-2000 kg/ha or at most 2% of the total aboveground biomass. These small diameter stems measuring less than 5 cm would however contribute to other processes occurring on the site such as root growth, turnover, and annual litter production and these processes are considered in our other measurements.

Root biomass followed a distinctly different trend from standing aboveground biomass. In contrast to aboveground biomass which increased as site quality increased, fine root biomass (0-3 mm) actually decreased as site quality increased. Fine root biomass demonstrated an approximate two fold difference ranging from 2100 kg/ha on higher quality sites to 3800 kg/ha on lower quality sites. These results would indicate that for these oak-hickory dominated forest stands, proportionally greater amounts of fine roots are needed to maintain the trees in the stand as site quality decreases. Accurate estimates of medium and coarse root biomass is more difficult due to their heterogeneous distributions throughout the stand and medium and coarse rooting in the deeper soil horizons. Biomass on a kg/ha basis for medium roots ranged from under 2000 kg/ha to almost 5000 kg/ha. Similarly, estimates of coarse roots (>10 mm diameter) were also highly variable with estimates ranging from 400 to 5,000 kg/ha (Table 2) indicating the difficulty in accurately assessing this component. Total root biomass, the sum of fine, medium, and coarse roots ranged from 6,300 to 12,400 kg/ha.

Annual inputs to the sites via annual litterfall increased as site productivity increased. Lower quality sites, Jasper-Pulaski and Clark, produced substantially lower autumnal leaf-fall than the more mesic, nutrient rich sites. Other litterfall in the form of seeds, twigs, and branches was similar across the five sites examined and contributed lower total biomass inputs than leaf-fall. These differences in annual leaf-fall are due to differences in stand basal areas which should result in significant differences in leaf area indices between the stands examined (Chapman and Gower 1991).

Forest floor biomass storage showed a distinct trend for greater total biomass on more xeric, nutrient poor sites (Table 2). Total forest floor biomass ranged from approximately 18,000 kg/ha at the mesic, nutrient rich sites represented by Nelson-Stokes and Feldun to over 35,000 kg/ha at the xeric Jasper-Pulaski site. Broken down into its individual

Table 2. Biomass of each respective component at the Nelson-Stokes, Feldun, SIPAC, Clark, and Jasper-Pulaski sites. Values are given in kg/ha. Standard deviations are in parentheses under each respective value.

	Site				
Component	Nelson-Stokes 85-90	Feldun 75-80	SIPAC 65-70	Clark 55-60	Jasper-Pulaski 55-60
Living Tree Wood	246,358 (57,620)	221,830 (40,215)	210,319 (61,951)	124,260 (17,849)	124,765 (27,126)
Fine Roots (<3 mm)	2,133 (1,348)	2,742 (1,475)	3,597 (1,331)	3,365 (1,828)	3,759 (1,662)
Medium Roots (3-10 mm)	2,930 (1,394)	4,784 (4,255)	3,782 (2,453)	3,434 (4,992)	1,877 (1,863)
Coarse Roots (>10 mm)	1,218 (4,244)	4,980 (9,585)	381 (1,180)	1,411 (3,801)	2,552 (8,331)
Root Total	6,281 (5,248)	12,391 (11,875)	7,610 (3,043)	8,146 (5,751)	8,032 (8,863)
Annual Leaf-fall	4,890 (519)	4,836 (589)	5,145 (423)	4,317 (493)	3,947 (721)
Non leaf Litterfall	1,003 (687)	967 (606)	1,207 (613)	1,312 (1,254)	972 (584)
Total Litterfall	5,893 (791)	5,803 (1,138)	6,352 (826)	5,628 (1,584)	4,918 (1,026)
Litter Layer	5,394 (879)	5,152 (948)	4,507 (923)	5,646 (828)	3,887 (731)
Fermentation Layer	8,660 (1,885)	10,661 (4,711)	16,171 (6,038)	11,068 (2,011)	16,826 (4,036)
Humus Layer	4,443 (1,486)	3,651 (1,308)	6,265 (2,946)	10,419 (8,024)	14,718 (3,800)
Forest Floor Total	18,496 (1,614)	19,464 (5,461)	26,943 (8,361)	27,133 (8,887)	35,431 (6,687)

components, the litter layer biomass comprised from 29 and 26 percent of the total forest floor biomass at the mesic Nelson-Stokes and Feldun sites respectively, but comprised only 11% of the total forest floor biomass at the Jasper-Pulaski site. The fermentation layer represented a large total proportion of the forest floor ranging from 41-60% of the total forest floor biomass. The humus layer biomass showed a wide disparity between mesic sites and xeric sites. In the more mesic sites, the humus layer represented a relatively small proportion of the total forest floor biomass (19 and 24% at the Feldun and Nelson-Stokes sites respectively) while the more xeric sites, the humus layer contained a relatively large proportion of the total forest floor biomass (38 and 42% of the forest floor biomass at the Clark and Jasper-Pulaski sites respectively).

Total carbon and nitrogen storage in each particular pool (i.e., living trees or annual leaf-fall) is a function of each individual pool biomass multiplied by the carbon or nitrogen concentration of that pool. Therefore, both the size of the pool and the nutrient concentrations of the pool become important in determining the carbon or nitrogen storage potential. It is possible for a particular component to have a high value for nitrogen high concentration, but if the biomass is low, the overall contribution may be small. Likewise, a particular component with a low nitrogen concentration may be highly significant if the biomass is large. In addition to determining the overall carbon or nitrogen storage and the potential for nutrient cycling at a site, the temporal aspects of changes in a specific pool are also important. Pools that turn over on a relatively frequent basis, i.e, a short term period, become more important to the overall cycling of nutrients than a much larger pool that turns over on a less frequent basis, i.e. over longtime period.

When static pool sizes are determined, it is clear that the aboveground biomass and mineral soil are the most significant storage components for carbon in these systems (Table 3). Carbon storage in the aboveground biomass in these systems is a function of the relatively large amount of biomass and high carbon concentrations (approximately 45%). The soil has a low carbon concentration, but its extremely high mass on a kg/ha basis makes this component a highly significant carbon storage pool. The forest floor follows these components as the third primary carbon pool in these systems. In comparison to the aboveground biomass and soil components, the forest floor contains smaller amounts of carbon. Still, this component can be extremely important in these sites. It represents a transition zone between the relatively decomposable organic carbon sources contained in the vegetation to the much more recalcitrant soil organic matter.

Inputs of carbon in these systems come primarily from aboveground litterfall and fine root turnover. While these represent a relatively small percentage of system carbon, the yearly additions of these sources are extremely important. Mortality of mature trees is relatively low and carbon returns from this source is low at the present stage of stand development. Therefore, carbon inputs to the systems come primarily from fixed atmospheric carbon that is returned to the site via leaf-fall, litterfall, and root turnover.

Nitrogen pools in these systems show somewhat different trends. The mineral soil contains the vast majority of the total nitrogen found in each stand. The aboveground biomass becomes the second most important nitrogen pool, with the forest floor organic layers ranking third in importance (Table 4). During decomposition of leaf and litter fall, nitrogen concentrations increase. The nitrogen concentration of the fermentation layer is higher than either the litter or humus layers. The high nitrogen concentration coupled with a relatively high biomass leads to the high total nitrogen storage in this pool. As site quality decreases, the nitrogen contained in the forest floor organic layers becomes increasingly more important. The nitrogen contained in the forest floor organic layers at the Jasper-Pulaski site is approximately 70% of the nitrogen contained in the aboveground biomass but the organic layers at Nelson-Stokes and Feldun contain only 15% and 13%, respectively. These results indicate that the forest floor becomes increasingly more important as site quality decreases. In the more productive stands examined, the total amount of nitrogen contained in the forest floor is relatively insignificant compared to that contained in the aboveground biomass. In the more xeric, less productive stands sampled, the nitrogen contained in the forest floor approaches the level of nitrogen contained in the aboveground biomass. This high nitrogen retention in the forest floor organic layers may be a significant nitrogen conservation mechanism in these stands and may become a critical issue in proper management of these stands.

Table 3. Carbon content of each component at the Nelson-Stokes, Feldun, SIPAC, Clark, and Jasper-Pulaski sites. Values given are in kg/ha. Standard deviations are in parentheses under each respective value.

	Site				
Component	Nelson-Stokes 85-90	Feldun 75-80	SIPAC 65-70	Clark 55-60	Jasper-Pulaski 55-60
Living Tree	114,265 (32,576)	98,892 (17,602)	99,692 (29,627)	61,496 (8,711)	57,180 (12,432)
Fine Roots (<3 mm)	988 (624)	1,202 (674)	1,582 (611)	1,504 (904)	1,726 (796)
Medium Roots (3-10 mm)	1,363 (1,394)	2,132 (1,896)	1,793 (1,163)	1,403 (2,172)	860 (854)
Coarse Roots (>10 mm)	515 (1,794)	2,106 (4,054)	160 (494)	547 (1,548)	1,080 (3,526)
Root Total	2,866 (2,298)	5,440 (5,104)	3,534 (1,425)	3,605 (2,529)	3,666 (3,785)
Annual Leaf-fall	2,258 (240)	2,214 (270)	2,370 (195)	1,989 (227)	1,891 (345)
Non leaf Litterfall	463 (296)	443 (271)	556 (281)	604 (568)	466 (329)
Total Litterfall	2,721 (355)	2,657 (503)	2,926 (380)	2,593 (707)	2.357 (477)
Litter Layer	2,490 (409)	2,339 (453)	2,076 (447)	2,601 (372)	1,862 (352)
Fermentation Layer	3,915 (884)	4,791 (2,181)	7,297 (2,780)	4,932 (906)	7,933 (1,871)
Humus Layer	1,148 (544)	1,031 (517)	2,035 (1,465)	1,803 (396)	4,312 (1,509)
Forest Floor Total	7,553 (966)	8,161 (2,493)	11,408 (3,983)	9,336 (1,167)	14,107 (2,305)
Mineral Soil	45,082	77,156	45,836	45,755	35,543

Table 4. Nitrogen content of each component at the Nelson-Stokes, Feldun, SIPAC, Clark, and Jasper-Pulaski sites. Values are given in kg/ha. Standard deviations are in parentheses under each respective value.

	Site				
Component	Nelson-Stokes 85-90	Feldun 75-80	SIPAC 65-70	Clark 55-60	Jasper-Pulaski 55-60
Living Tree Wood	1,947 (555)	2,024 (360)	1,416 (421)	882 (125)	867 (189)
Fine Roots (<3 mm)	21 (13.3)	24 (14)	29 (11)	25 (15)	43 (20)
Medium Roots (3-10 mm)	19 (19)	41 (36)	33 (21)	25 (39)	23 (22)
Coarse Roots (>10 mm)	6 (21)	26 (51)	2 (6)	7 (19)	14 (44)
Root Total	46 (32)	91 (75)	64 (26)	59 (42)	79 (55)
Annual Leaf-fall	43 (4)	47 (6)	50 (4)	35 (4)	37 (7)
Non leaf Litterfall	9 (6)	9 (6)	12 (6)	11 (10)	9 (7)
Total Litterfall	51 (7)	57 (11)	62 (8)	46 (12)	46 (9)
Litter Layer	47 (8)	51 (13)	44 (9)	46 (11)	36 (9)
Fermentation Layer	154 (35)	146 (64)	215 (55)	154 (29)	316 (83)
Humus Layer	67 (33)	57 (32)	89 (31)	104 (27)	257 (80)
Forest Floor Total	268 (43)	254 (85)	348 (77)	305 (50)	609 (127)
Mineral Soil	12,288	18,005	10,423	23,286	7,561

Nitrogen contained in the annual litterfall and fine root biomass is the primary pool which is recycled in these system. These pools turn over at relatively frequent intervals and while their pool size at any given point is relatively small, the turnover of these pools at approximately yearly intervals is highly significant. Assuming equal longevity and turnover rates of fine roots on all sites, the role of fine roots in the nitrogen cycle in these stands is most significant on the more xeric, nutrient poor sites. On the more mesic sites, the role of leaf-fall and litterfall potentially outweighs the role of root turnover. Current research is underway to determine the rates of fine root turnover in these forest stands and the environmental factors controlling fine root production and turnover. Once this research is completed, a more complete understanding of the role of C and N inputs via above and belowground production will be achieved.

DISCUSSION

The results of this study support previous studies which indicate that standing aboveground biomass, forest floor organic layers, and the mineral soil itself are the primary storage pools for carbon and nitrogen in forest ecosystems. Carbon and nitrogen storage in the five oak-hickory stands examined is a function of both the relative biomass of the pool and the nutrient concentration of the components in the pool. The selection of oak-hickory sites located along a productivity gradient demonstrates the wide differences that may exist within this forest type. In addition, it demonstrates that the relative importance of various pools may change as site quality changes.

Both biomass and nutrient concentration must be determined in order to determine the size of the carbon or nitrogen pool within a system. Biomass determinations of the various components reveal the wide differences that may be encountered in oak-hickory forests. This may be expected due to the broad ecological amplitude of these species, i.e., ranging from xeric, nutrient poor sites to mesic, nutrient rich sites. This wide variation in site quality is readily reflected in standing aboveground biomass of the five stands. The estimates of standing aboveground biomass ranging from 124,000 kg/ha to 246,000 kg/ha is within the range generally encountered in moderately low to high productivity hardwood forests. There are a wide variety of aboveground biomass estimates for hardwood forests ranging from a low of approximately 68,000 kg/ha for northern pin oak/bur oak forest on outwash sands in northwest Wisconsin (Bockheim and Leide 1991) to over 220,000 kg/ha for sugar maple/northern red oak stands in Michigan (Zak and others 1989). Aboveground biomass estimates for oak dominated stands in the same study range from 83,000 to 207,000 kg/ha (Zak and others 1989). Aboveground biomass estimates for mixed hardwood forests in eastern Tennessee range from 188,000-207,000 kg/ha (Johnson and others 1982). The estimates of aboveground biomass are within expected ranges for this forest type, and indicate its broad ecological adaptation.

Fine root biomass is the second component that is estimated in these stands. Our fine root biomass estimates are at the lower range of expected values for hardwood ecosystems. Comparisons between different studies are complicated by several factors. Fine root biomass often undergoes fairly wide seasonal variation (Edwards and Harris 1977, Hendrick and Pregitzer 1993). In addition, many researchers sample differing depths in the soil profile and have different definitions of fine roots (i.e., < 2 mm vs. < 3 mm). These confounding factors make direct comparisons difficult. At the higher range of fine root biomass estimates, Hendrick and Pregitzer (1993) estimate sugar maple stands in Michigan contain 7,900 kg/ha and 9,500 kg/ha of fine roots (<2.0 mm) in April and October respectively. Lower estimates of fine root biomass are also reported. Joslin and Henderson (1987) determine that white oak stands in Missouri contained 5,500 kg/ha of fine roots (<2 mm) in the upper 44 cm of the soil profile. Aber and others (1985) estimate fine root biomass measures ranging from 2,700 to 5,200 kg/ha for oak stands in the state of Wisconsin. Mixed hardwood stands in Massachusetts contain an average fine root (<3 mm) biomass of 5,100 kg/ha (McClaugherty and others 1982). All sampling in our stands is carried out during late summer to early fall during a time at which fine root biomass should reach peak levels (Edwards and Harris 1977). This sampling protocol should assure that the relative differences between fine root biomass in the five stands is accurately reflected in our measurements.

Determinations of medium and coarse root biomass become more problematic. Roots of these diameters exhibit a great deal of spatial heterogeneity. Sampling to determine these components must be intense to accurately access these components. This problem is magnified for coarse roots (> 10 mm). Few of the sample cores collected

contained roots of this diameter. Estimates of these components is also complicated by the fact that roots of these diameters may have a distribution through the lower soil horizons. While examinations of medium and coarse root distribution in soil pits reveal that the majority of these roots are confined to the upper 30 cm of soil, there are medium and coarse roots in lower soil horizons which are not sampled. Therefore, our estimates likely underestimate this component due to their horizontal and vertical distributions in the soil profile. The exact magnitude of this underestimation is difficult to determine, but several other studies have examined root distribution in similar forest stands. Johnson and Risser (1974) report that mixed post oak/blackjack oak stands contained 39,000 kg/ha of total root biomass. Kelly and Joslin (1989) indicate that mixed oak stands in Tennessee contained from 30,000-35,000 kg/ha of total root biomass. Of this total, from 70-80% is contained in the upper 30 cm of the soil profile. In northern hardwood forests in New Hampshire, total root biomass is estimated at 27,000 kg/ha with over 80% of this total contained in the upper 20 cm of soil (Fahey and others 1988). Estimates of the standing crop of fine roots in the stands we examined is much less problematic. Excavations reveal that these roots are distributed much more evenly throughout the stand. In addition, the vast majority of these roots are contained in the upper 30 cm of the soil profile. Therefore, our estimates of this pool, which is the most active belowground in terms of turnover and its significant impacts, are less subject to error due to vertical and horizontal variability.

The trend for fine root biomass to increase as site quality decreases has been observed by other researchers. Binkley and others (1986) found that sites with high levels of nitrate had lower fine root biomass than lower quality sites where lower levels of ammonium were present. In addition, Harris and others (1977) found that belowground biomass increased relative to aboveground biomass as site quality of tulip poplar and loblolly pine stands decreased. The trend observed in our study supported this observation. On the more xeric, nutrient poor sites, fine root biomass was much higher than on the higher quality sites. When the standing aboveground biomass differences between the low and high quality sites was considered, the preferential allocation of fixed carbon belowground on the lower quality sites became more striking.

Inputs to the system via annual leaf and total litterfall were within the commonly reported figures for hardwood ecosystems. Zak and others (1986) reported that annual leaf-fall in Michigan ranged from 1,600 kg/ha for xeric oak ecosystems to 3,100 kg/ha for mesic sugar maple/northern red oak ecosystems. Total litterfall in Missouri oak ecosystems was 5,200 kg/ha (Joslin and Henderson 1987) while figures for oak forests in Wisconsin ranged from 3,000 kg/ha (McClaugherty and others 1985) to 5,900 kg/ha (Nadelhoffer and others 1985). The results of our study indicated that litter inputs in oak-hickory forests varied widely and were dependent upon the productivity of the site. The more productive sites had much greater annual leaf-fall than the lower productivity sites. Other litterfall (seeds, twigs, bark, etc.) was approximately equal across sites.

Differences in the forest floor organic layers were also apparent across this site productivity gradient. Our results indicated that the biomass of the forest floor increases as site quality decreases. This was due to factors other than strictly the input of detritus to the forest floor as inputs were greater on higher productivity sites than on lower productivity sites. Rather, it appeared that either site conditions or the chemical composition of the litter inputs themselves affected forest floor development.

It has been long-established that the rate of organic matter decomposition is moisture and temperature dependent. The highest rates of organic matter decomposition are observed under warm, moist conditions. Either drought, cold conditions, or oxygen limitations can reduce the rate of organic matter decomposition (Meentenmeyer 1978, White and others 1988). Conditions which limit organic matter decomposition are expected to increase forest floor biomass if other factors are held constant. Periodic monitoring of the five stands examined demonstrates little difference in soil temperature, but relatively large differences in soil moisture availability. The Clark and Jasper-Pulaski sites have limited plant available water holding capacity. Monitoring of the soil moisture status of these soils shows that water is rapidly depleted from these sites during relatively short-term midsummer drought events. Therefore, on the lower productivity sites, it is likely that decomposition rates during midsummer are lowered due to water limitations. This in turn may increase the amount of forest floor organic matter compared to that found on more mesic sites.

The second factor that may be significant in altering forest floor developmental patterns is the chemical characteristics of the litter inputs. Litter of different chemical characteristics is found to decompose at vastly different rates. In general, nitrogen and lignin concentrations are the prime chemical constituents controlling litter decomposition rates (Aber and others 1990, Berendse and others 1987, Berg and others 1993, Melillo and others 1989). The early stages of decomposition are characterized by the microbial utilization of cellulose (Melillo and others 1989). Late stages of decomposition are dependent upon microbial degradation of lignin and this process usually proceeds at much slower rates (Aber and others 1990). Further, lignin is often viewed as a precursor to stable soil organic matter formation (Paul and Clark 1989). The differing chemical characteristics of the litter inputs themselves may affect the forest floor development. The litter inputs at the Clark and Jasper-Pulaski sites are composed almost entirely of oak. Previous studies have demonstrated that oak species typically have high lignin concentrations of approximately 20-25% and typically exhibit slow decomposition rates (Aber and others 1990). Other species such as sugar maple (Aber and others 1990) or tulip poplar (White and others 1988) have much lower lignin concentrations and exhibit much more rapid decomposition rates. The leaf-fall at the more mesic sites is composed partly of oak litter, but also of a variety of other species which, based on previous studies, should have lower lignin concentrations and decompose at more rapid rates. Under these conditions, we would expect lower biomass in the forest floor in mesic stands where species other than oak contribute significant amounts of annual leaf-fall.

The total carbon and nitrogen pool size is a function of the biomass of that pool multiplied by its nutrient concentration. Carbon pool sizes follow slightly different trends than nitrogen. While the aboveground biomass, soil, and forest floor organic layers contain the majority of carbon and nitrogen in the oak-hickory forests examined, the frequency of inputs must also be considered to properly assess the true significance of nutrient inputs at any one point. In these second growth forest stands, tree mortality is at a relatively low level. Nutrients that are contained in trees that do die become very slowly available due to the very slow decomposition rates of tree boles and the strong immobilization of nutrients in decaying boles (Alban and Pastor 1993). On a short term basis in these stands, this pool of nutrients is cycled at a very slow rate.

The soil nitrogen and carbon pools are also very large in these systems, but there is a problem in defining the relative importance of this pool. While it cannot be argued that there is a huge reserve of carbon and nitrogen in this pool, the stable soil organic matter of this fraction appears to be relatively recalcitrant and undergoes relatively slow decomposition and nutrient release. The relative rates of turnover from these pools have been determined. Readily decomposable plant material has a half-life of 0.17 years. Plant material that is highly resistant to decay has a half-life of 23 years. In contrast, physically and chemically stabilized soil organic matter have half-lives of 50 and 2300 years respectively (Paul and Clark 1989). Figures such as these indicate that while the soil pools are very large, the turnover rates of the pools must also be considered to assess their true impacts.

Annual litterfall and fine root production and turnover are two pools that are constantly cycling C and N in these ecosystems. Although their pool sizes at any given point are small relative to the other pool sizes, their inputs are fairly steady and over time, can become highly significant. Clearly, the production rates of aboveground litter and fine roots become critical issues in determining the importance of each pool. We currently have good estimates of aboveground leaf and litter production, but standing fine root biomass is less desirable than root production rates. The second critical factor becomes the residence time of fine roots. Fine root longevity becomes a critical issue in these ecosystems. Estimates of fine root longevity can vary widely with estimates ranging from 0.6 years (Harris and others 1977) to 4.8 years (Joslin and Henderson 1987). Most estimates of the residence time of fine roots fall in the range of 0.5 to 2.0 years (Nadelhoffer and others 1985). If these estimates of root longevity hold true for our forest stands and root longevity is the same for the five stands examined, this would indicate that root turnover could be as important or more important than aboveground litterfall in supplying organic matter and nitrogen to these systems. The significance of belowground turnover may be greatest on the more xeric, nutrient poor sites. These stands had higher standing fine root biomass. In addition, on these sites, leaf and litter fall was lower than on the more fertile sites. These observations would indicate that fine roots become increasingly more important as site quality decreases if, in fact, production and turnover is equal across sites. Additional research is currently in place that will determine fine root production, turnover, and decomposition rates in each of the five stands examined. When this is completed, the relative importance of above and below ground pools in the nutrient cycles of these stands will become clear.

CONCLUSIONS

The results of this research support previous research that identified the soil, forest floor organic layers, and the aboveground biomass as the most significant pools of carbon and nitrogen in forest ecosystems. The relative importance of each pool can change as site characteristics change. The nitrogen contained in the forest floor pool becomes increasingly more important as site quality decreases. This suggests that these forest floor organic layers may serve as important nutrient conserving mechanisms in these less productive ecosystems. The research also suggests that while the absolute pool size may be important, the temporal aspects of the turnover of the pool must also be considered. In the forests we examined, most inputs to the systems come from annual litterfall and root turnover. Estimates of annual litterfall is easily obtained, but root production, turnover, and decomposition is much more difficult to estimate. Current research is focusing on quantifying these factors in the five stands described. In addition, more stands have been sampled to develop a greater understanding of the processes taking place in oak-hickory dominated stands found on a wide variety of site types.

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THE DISTRIBUTION OF NITROGEN AND PHOSPHORUS IN FOREST FLOOR LAYERS OF OAK-HICKORY FORESTS OF VARYING PRODUCTIVITY

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Abstract: The forest floor plays a major role in the storage and recycling of nutrients which, in turn, are important in maintaining the growth and productivity of forest ecosystems. The development of forest floor organic layers as influenced by litter quality and site quality is unclear. Previous studies in this lab have shown that the size and distribution of available nutrient pools in the forest floor fractions are related to site productivity.

The objectives of this project are to: (1) determine the biomass distributions of forest floor organic fractions across oak-hickory forest sites along a productivity gradient, (2) determine the distribution of nitrogen and phosphorus in these fractions, and (3) determine the relative contribution of nutrients from each forest floor fraction to the available nutrient pool as site productivity changes. Each forest floor-soil core was grossly separated into O_i, O_e, O_a, and mineral soil (A) layers in the field. In the laboratory, the O_a and A layers were further separated into size fractions via wet sieving and flotation. Biomass, total N, and total P were determined for each fraction.

INTRODUCTION

The sustained growth and long-term productivity of forest ecosystems are dependent, in large part, on nutrient availability and uptake. Nutrient inputs from the atmosphere and rock weathering are important to the long-term development of soils and ecosystems, but on an annual basis nutrient recycling within ecosystems forms the major source of nutrients for plant use. Although the forest floor contribution to the total nutrient pool within a site may be relatively small, it is the location where, by weathering and microbial action, nutrients are eventually released from decomposing organic matter and made available for plant uptake and utilization.

Often nutrient dynamics in the forest floor have been studied by following the decomposition dynamics via litterbag studies. Most of these studies are short term (1-3yrs.), and do not take organic matter beyond the "recognizable" stage (Berendse and others 1987, McClaugherty and others 1985, Melillo and others 1982). On the other hand, there are numerous studies that fractionate soil to particle size to determine the presence of soil organic matter (SOM) and its nutrient contents. With most studies, fractionation techniques combine sieving (dry and wet) with sedimentation by gravity and centrifugation in water (Cameron and Posner 1979, Satoh 1976, Suzuki and others 1975, Suzuki and Kumada 1976), or sieving in combination with density separation with liquids of varying specific gravities (Ladd and Amato 1980, Ladd and others 1977a & b, Sollins and others 1984, Sollins and others 1983, Spycher and others 1983, Spycher and Young 1977, Turchenek and Oades 1979, Young and Spycher 1979). These techniques have all been applied to organic material that is smaller than the sand sized fraction, hence "unrecognizable". Few studies have taken undisturbed core samples and analyzed the organic matter that is found in the larger than sand fraction (Ohta and Kumada 1978), or followed nutrients throughout the profile from freshly fallen litter to soil organic matter (Rustad 1994).

This paper is part of a comprehensive, long-term study examining nutrient cycling processes in oak-hickory dominated forest stands throughout the state of Indiana. The study reported in this paper will examine the forest floor physical and chemical characteristics of a range of forest systems, and will quantify the biomass and nutrients

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contained in forest floor organic fractions in oak-hickory sites found along a productivity gradient. The organic material was purified into fractions of differing size from freshly fallen litter to the upper few centimeters of mineral soil. The objectives were to: (1) determine the biomass distributions of forest floor organic fractions across oak-hickory forest sites along a productivity gradient , (2) determine the distribution of nitrogen and phosphorus in these fractions, and (3) determine the relative contribution of nutrients from each forest floor fraction to the available nutrient pool as site productivity changes.

METHODS

Study Site Descriptions

In the preliminary stages of this study, a large number of sites throughout the state of Indiana were surveyed. To be considered, all sites had to be dominated by oak-hickory species with these species comprising at least 50% of the total stand basal area. The initial list of stands was comprised of a gradient of potential site productivities; i.e. site indices ranging from approximately 55 to 95 feet (base-age of 50 for oak species). The relative productivity of these sites was subjectively judged using a combination of data for soil moisture supplying capacity, soil nitrogen supplying capacity, and indicator plant communities (Van Kley and others 1994). The parent material of the soils ranged from thick loess deposits over glacial till with a high moisture holding capacity and high plant available water to outwash sand deposits with very low plant available water holding capacity. Nitrogen supplying capacity in the upper 30 cm of the mineral soil was determined in all stands in the initial pool of sites using 14-day anaerobic incubations (Powers 1980).

From this initial list of stands, five stands best representing a site productivity gradient were chosen throughout the state of Indiana. The selected stands were second-growth, oak-hickory forests with dominant overstory trees in each stand ranging in age from 80-120 years with no signs of disturbance for at least 20 years. The site of highest productivity, identified as Nelson-Stokes (NS), is located in Putnam County in the Entrenched Valley Section of the Central Till Plain Natural Region (Homoya and others 1985). The Russell silt-loam soil (Fine-silty, mixed, mesic Typic Hapludalf) on this site developed from moderately thick loess deposits overlying glacial till. The overstory vegetation is dominated by white oak (*Quercus alba,* L.), northern red oak (*Quercus rubra,* L.), and assorted hickory species (*Carya* spp., Nutt), with a subcanopy dominated by sugar maple (*Acer saccharum,* Marsh.). The site index for oak on this site is approximately 85-90 ft. at a base age of 50 years.

The second site, Feldun Purdue Agricultural Center (Feldun), is located in Lawrence County in the Mitchell Karst Plain Section of the Highland Rim Natural Region (Homoya and others 1985). The soil is a Caneyville silt-loam (Fine, mixed, mesic Typic Hapludalf) developed from a thin layer of loess covering an older paleosol. The site is underlain by limestone bedrock within 60 cm of the surface of the soil. The vegetation is dominated by black oak (*Quercus velutina,* Lam.) , northern red oak, tulip poplar (*Liriodendron tulipifera,* L.), and beech (*Fagus grandifolia,* Ehrh.). The site index for oak is approximately 75-80, and for tulip poplar approximately 95.

The third site is located at the Southern Indiana Purdue Agricultural Center (SIPAC) in DuBois County. This site is located in the Crawford Upland Section of the Shawnee Hills Natural Region (Homoya and others 1985). The soil is a Wellston silt-loam (Fine-silty, mixed, mesic Ultic Hapludalf) derived from moderately thick loess deposits over paleosols derived from weathered sandstone and siltstone. The overstory vegetation is dominated by white oak, hickory, and northern red oak with a subcanopy dominated by sugar maple. The site index for oak is approximately 65-70.

The fourth site is located in Clark State Forest in Clark County in the Knobstone Escarpment Section of the Highland Rim Natural Region (Homoya and others 1985). The soil is Rarden silty-clay loam (Fine, mixed, mesic Aquultic Hapludalf) derived from a very thin layer of loess over soil derived from weathered shale. The overstory vegetation is comprised primarily of white and black oak. The site index for oak is approximately 55-60.

The fifth site is located at the Jasper-Pulaski Fish and Wildlife Area (JP) in Jasper County. This area is in the Kankakee Sand Section of the Grand Prairie Natural Region (Homoya and others 1985). The soil is a Plainfield fine sand (Mixed, mesic Typic Udipsamment). The site is located in an area of outwash sand reworked into dunes. The vegetation is composed almost entirely of white oak and black oak. The site index for oak is approximately 55-60.

In each of the five stands, three plots each measuring approximately 0.75 ha in size were established at random locations. Table 1 describes the basal area and relative dominance of the major species in each of the five sites. Data summarizing indices of potential site productivity: standing above-ground biomass, annual leaf-fall inputs, N-mineralization potential of the soil to 30 cm, soil moisture-holding capacity, and annual precipitation are presented in Table 2.

Table 1: Species composition of the five study stands. The first row under each species gives its basal area (m^2/ha), and the second row under each species gives its relative dominance in each respective stand (%). SI= site index for oak at a base age of 50 yrs.

Species	Nelson-Stokes (NS) (SI= 85-90)	Feldun (Feld) (SI= 75-80)	SIPAC (SI) (SI= 65-70)	Clark (Cl) (SI= 55-60)	Jasper-Pulaski (JP) (SI= 55-60)
White oak	20.6	1.7	22.0	16.0	15.2
	56.7	4.8	68.0	77.5	67.1
Black oak	0.0	11.9	0.2	2.7	6.4
	0.0	33.9	0.6	12.9	28.4
Northern red oak	3.5	3.1	2.4	0.0	0.0
	9.6	8.9	7.4	0.0	0.0
Hickory spp.	4.3	3.5	3.4	1.1	0.0
	11.9	10.1	10.6	5.2	0.0
Sugar maple	4.6	2.3	4.0	0.1	0.0
	12.7	6.6	12.3	0.4	0.0
Beech	0.3	3.7	0.0	0.0	0.0
	0.9	10.4	0.1	0.0	0.0
Tulip poplar	0.5	7.5	0.0	0.0	0.0
	1.3	21.4	0.0	0.0	0.0
Misc.	2.5	1.3	0.3	0.7	1.0
	6.9	3.9	1.0	4.0	4.5
Total Basal Area	36.3	35.0	32.3	20.6	22.6

Table 2. Study site characteristics that represent indices for potential site productivity.

Site	Above-ground Biomass (kg/ha)	Annual leaf-fall (kg/ha)	Annual N via leaf-fall (kg/ha)	N Min. Potential (*ug*/g)	Avail. Water Capacity (cm/cm)	Annual ppt. (cm)
Nelson-Stokes	246,358	4890	43	58.6	0.23	107
Feldun	221,830	4836	47	52.3	0.18	114
SIPAC	210,319	5145	50	37.2	0.19	107
Clark	124,260	4317	35	36.0	0.19	102
Jasper-Pulaski	124,765	3947	37	13.0	0.05	94

Forest Floor Fractionation

Immediately following leaf fall, twelve 1/10 m^2 forest floor/soil cores from each site (4 per plot) were separated into four principle layers corresponding to the natural horizons: O_i, O_e, O_a, and A mineral soil. The procedure involved placing the sampler on the forest floor, cutting along the edges down through the profile, and hand collecting each horizon. The O_i was all freshly fallen material, the O_e was decomposing but still recognizable material, and the O_a was highly organic but unrecognizable material. The A was sampled to 5 cm below the bottom of the forest floor or O_a horizon. The forest floor in these stands could be best described as mulls to moders. As such, clear delineations are lacking between forest floor organic layers. There is a great deal of mixing of the various forest floor organic layers. In addition, in some of the more mesic stands, litter decomposition proceeds very rapidly. In these stands, much of the litter mass from the previous year decomposes before the current year's leaves senesce, and mixing of various forest floor components can be extensive. In concept, a forest floor horizon is a zone of certain generic characteristics. The nature of the forest floor of the temperate deciduous forests represented in this study is characterized by extensive mixing of the various organic layers. The mixing of horizon components require further separation under laboratory conditions to isolate relatively homogenous components.

Following in-the-field separation, the individual forest floor horizons were hand sorted on a 2 mm sieve to remove any contaminating material from adjacent horizons (Figure 1). Any material falling through the sieve from the O_i, O_e, and O_a layers was included in the O_a horizon. Once contaminating material (i.e. roots, coarse organic material, gravel) was removed from the O_a and the A, these layers were sieved on the 2 mm sieve and the 2 mm fraction was further fractionated by flotation in water (Cameron and Posner 1979) and wet sieving. The entire horizon was placed in 2.5 L of deionized/distilled H_2O, stirred, and allowed to settle for 10 minutes. The floating fraction was removed, placed on a nest of two sieves (250 and 75 *u*m), rinsed until the water cleared, and air dried. This procedure was repeated for the sinking fraction. The floating fractions from the O_a and the A were combined because this (< 2 mm sized organic material) was considered a separate component from the mineral A horizon. Sieve sizes were selected to correspond with studies that fractionated soils into particle sizes (i.e. Sollins and others 1983, Spycher and others 1983, Suzuki and others 1975). In most studies, the 250 *u*m sieve was the largest used, and was used to break up aggregations to obtain the smaller particles. This study is not intended to reach specific particle size, the intent is to capture representative stages of decomposition based on size (aggregations included) as they are found "in nature". Therefore, the 250 *u*m sieve captures all components of "coarse sand" sized (2 mm-250 *u*m), and the 75 *u*m sieve captures all components of "medium sand" sized (250-75 *u*m). In review, for each sample point, four horizons were collected in the field, and eight size fractions were obtained after the final wet sieving step. From this point, the fractions will be

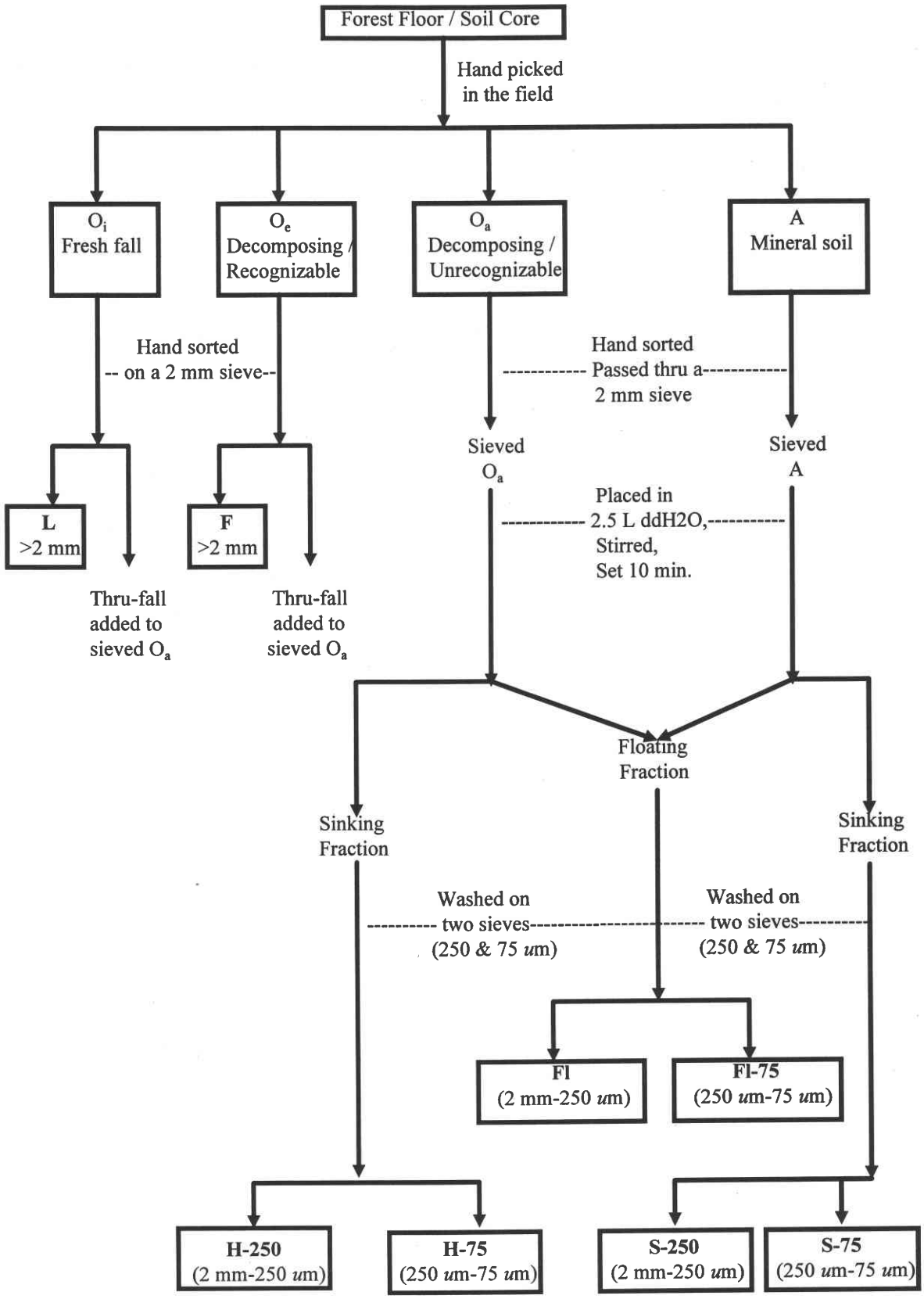

Figure 1. Forest floor fractionation.

called: **L**, corresponding with fresh fall O_i; **F**, O_e material > 2 mm; **Fl**, floating material 2 mm-250 *u*m; **Fl-75**, floating material 250 *u*m-75 *u*m; **H-250**, O_a material 2 mm-250 *u*m; **H-75**, O_a material 250 *u*m-75 *u*m; **S-250**, A horizon material 2 mm-250 *u*m; and **S-75**, A material 250 *u*m-75 *u*m. The fractions could just as easily been numbered 1-8; however, these descriptive labels were chosen to aid the reader in identifying the original source of the material and the minimum size.

Analyses

Biomass was obtained for all eight fractions by placing a 500 mg subsample of each horizon in a 65°C oven until a constant weight was obtained, and correcting the wet weight for moisture. In this study, A horizon mineral soil was considered an organic matter contaminant. The freshly fallen leaves (O_i) were considered free of mineral soil contamination. Visual inspection would indicate that this assumption was true since sample collections were made immediately following leaf-fall and mixing of this layer (O_i) with other layers was minimal at this time of the year. The amount of mineral soil in each forest floor layer increased with increasing proximity to the mineral soil. To account for these differences among components, the mineral soil correction factors described by Blair (1988) were used to determine how much of each respective horizon was actually organic matter. While Blair developed this method to determine mineral soil contamination of litter in litter decomposition studies, this methodology is also applicable to our study. Over extended periods of time, as a single cohort of litter is produced, and the decomposition process proceeds, a portion of this litter gradually becomes stabilized soil organic matter. Thus, in our study the mineral soil was merely considered a contaminating component of the organic fraction. Subsamples of each respective fraction were dry ashed at 450°C for 4.5 hours to obtain % ash-free dry matter (AFDM) and the following equation was used to correct for mineral soil contamination:

$$CF = (SaAFDM - A_tAFDM) - (LAFDM - A_tAFDM)$$

where:
CF is the fraction corrected for mineral soil contamination
SaAFDM is %AFDM of each respective fraction
A_tAFDM is %AFDM of the A horizon
LAFDM is %AFDM of the litter.

This equation assumes that litter inputs are constant over time. That is, that the litter currently being produced on the site is similar in amount and composition to previously produced litter that contributed to the coarse organic fractions in the forest floor and mineral soil. Inherent in this correction factor is the implication that each respective layer is merely a more decomposed representation of the current litter inputs. Through the decomposition process mineral soil contamination will increase due to incorporation of the organic materials into the mineral soil. In addition, the equation also assumes that the A horizon mineral soil is the primary contaminant in each fraction. Based on visual inspection of the forest floor organic layers under natural field conditions and microscopic inspections of the fractionated components, these assumptions are valid.

Total nitrogen (N) was obtained by digesting a 100 mg or 500 mg oven dried subsample by the micro-kjeldahl technique (Nelson and Sommers 1973). One-hundred mg samples were used for highly organic horizons (L thru Fl-75) while 500 mg samples were used for samples that had lower organic matter contents (H-250 thru S-75). The nitrogen contents reported are for the corrected biomass values. Nitrogen concentrations of these fractions were corrected for mineral soil contamination again following the methods of Blair (1988) and Rustad (1994):

$$CFNt = [SaNt - (FSl \times A_tNt)] / CF$$

where:
CFNt is the nutrient concentration for the fraction corrected for mineral soil contamination
SaNt is the nutrient concentration for each respective fraction
FSl is the fraction of sample that is soil
A_tNt is the average conc. of nutrients in the A horizon
CF is the fraction of the sample that is organic.

Total phosphorus (P) was obtained by digesting a 100 mg subsample in perchloric acid and determining P by the phospho-molybdate blue method (Olsen and Sommers 1982). Results for P reported here are based on the fractions before the corrections for soil contamination were made. From these data, N and P concentrations and mass of each respective component on an per hectare a basis, total N and P contents were calculated on a per hectare basis.

Statistical analyses were performed with the general linear model (GLM) procedure contained in the SAS system (SAS institute 1985). In the analysis, plots were nested within sites. Two primary comparisons were made. Within each individual site, comparisons were made between all forest floor components. In addition, comparisons were made between individual forest floor components across sites (i.e. L layer at NS vs. SI vs. JP). Detection of significant differences were made with Duncan's Multiple Range Test at a 5% probability level.

RESULTS

Biomass

Across the site productivity gradient, biomass for the forest floor fractions shows two different trends (Table 3). As productivity decreases, the biomass of the L fraction decreases, with significant differences between the most productive site (NS) and the lowest productive site (JP). JP has consistently and significantly more biomass than NS for all organic fractions other than L. NS contains significantly low amounts of biomass in the F, Fl, Fl-75 fractions, however in the last four fractions (H-250, H-75, S-250, S-75) there are no significant differences between biomass and site.

Table 3: Biomass of forest floor fractions corrected for mineral soil contamination (kg/ha).

Forest Floor Fraction	Nelson-Stokes (NS)	Feldun (Feld)	SIPAC (SI)	Clark (Cl)	Jasper-Pulaski (JP)
L	5394 a* : A**	5152 b : AB	4507 b : BC	5646 bc : A	3887 e : C
F	6210 a : C	8063 a : ABC	10455 a : A	7013 ab : BC	9049 bc : AB
Fl	2309 bc : C	2412 c : C	5281 b : AB	3661 cde : BC	6966 cd : A
Fl-75	141 e : C	186 e : BC	435 d : B	394 f : BC	811 f : A
H-250	3263 b : B	2706 c : B	4749 b : B	9060 a : A	11822 ab : A
H-75	1180 d : B	945 de : B	1516 cd : B	1359 ef : B	2896 ef : A
S-250	2395 bc : B	2231 cd : B	3221 bc : B	4578 bcd : B	11976 a : A
S-75	1726 cd : B	1349 cde : B	2023 cd : B	2632 def : B	5274 de : A
Fraction Totals	22,618	22,044	32,187	34,343	52,681

* Column means with the same lower-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.
** Row means with the same upper-case letters are not significantly different at the 5% level using Duncan's Multiple Range Test.

The importance of each fraction within a site was calculated: i.e. the mass of a specific fraction / mass of the total of the eight fractions X 100. Organic material contained in the smaller size fractions tends to become increasingly important as productivity at the site decreases. Similarly, the percentage of the sum of the mass of the first four fractions (L - Fl-75) to the total of all fractions declines as site productivity declines, i.e. NS- 62%, Feld- 70%, SI- 64%, Cl- 49%, and JP- 40%.

Nitrogen Concentration

Nitrogen concentration of a particular forest floor fraction was not affected by site and showed no consistent trend with the site productivity gradient (Table 4). At each site, N concentration tended to be greatest in the Fl and Fl-75 fractions, followed by the H-250 and H-75 fractions, the L and F fractions, and the S-250 and S-75 fractions. The N concentrations for the A fractions represent total N in the A horizon (mineral soil and organic matter < 2 mm). Nelson Stokes, on the high end of the productivity gradient, and Jasper-Pulaski, on the low end of the productivity gradient were significantly different, and each was different from the sites representing the moderately high to moderately low productivity sites.

Table 4. Nitrogen concentrations (%) in forest floor layers corrected for mineral soil contamination.

Forest Floor Fraction	Nelson-Stokes (NS)	Feldun (Feld)	SIPAC (SI)	Clark (Cl)	Jasper-Pulaski (JP)
L	0.88 f* : BC**	0.98 ef :A	0.97 ef : A	0.82 c : C	0.93 d : AB
F	1.53 cd : A	1.23 de :B	1.24 cd : B	1.19 b : B	1.60 bc : A
Fl	2.25 b : A	1.85 b : B	1.75 b : B	1.71 a : B	2.22 a : A
Fl-75	2.60 a : A	2.44 a : AB	2.00 a : C	1.87 a : B	2.26 a : B
H-250	1.45 de : AB	1.54 c : AB	1.41 c : AB	1.24 b : B	1.80 b : A
H-75	1.75 c : A	1.40 cd : A	1.68 b : A	1.75 a : A	1.47 c : A
S-250	1.19 e : A	1.00 ef : ABC	1.06 de : AB	0.71 c : C	0.80 d : BC
S-75	1.81 c : A	0.89 f : B	0.80 f : B	0.63 c : B	0.77 d : B
A	0.24 g : A	0.19 g : B	0.19 g :B	0.21 c : AB	0.14 e : C

* Column means with the same lower-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.
** Row means with the same upper-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.

Phosphorus Concentration

At all sites, phosphorus concentrations in the F and Fl forest floor fractions were significantly greater than the other fractions (Table 5). For a specific site, the P concentration was greatest for the Fl fraction followed by the F fraction and tended to be followed by the L, H-75, H-250, Fl-75, and the S-250 and S-75 fractions.

Table 5. Phosphorus concentrations (%) in forest floor fractions not corrected for mineral soil contamination.

Forest Floor Fraction	Nelson-Stokes (NS)	Feldun (Feld)	SIPAC (SI)	Clark (Cl)	Jasper-Pulaski (JP)
L	0.077 c* : B**	0.069 c : C	0.062 c : C	0.050 cd : D	0.129 b : A
F	0.119 b : A	0.101 b : B	0.096 b : B	0.096 b : B	0.121 b : A
Fl	0.146 a : A	0.125 a : B	0.113 a : B	0.123 a : B	0.152 a : A
Fl-75	0.058 d : AB	0.040 e : BC	0.046 de : BC	0.036 de : A	0.077 c : A
H-250	0.055 d : A	0.055 cd : A	0.055 cd : A	0.044 de : A	0.060 c : A
H-75	0.090 c : A	0.065 c : B	0.062 c : B	0.062 c : B	0.064 c : B
S-250	0.043 d : A	0.044 de : A	0.020 f : B	0.037 de : AB	0.019 d : B
S-75	0.023 e : AB	0.031 e : A	0.032 ef : A	0.028 e : A	0.011 d : B

* Column means with the same lower-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.
** Row means with the same upper-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.

Nitrogen Content

Nitrogen contents of all of the forest floor fractions (Table 6) tended to follow a pattern similar to forest floor biomass, i.e. except for the L fraction, the JP site had significantly more total N in each fraction than the other sites including NS the highest productivity site. When the importance of each fraction is calculated against the total of all fractions, the F fraction at all sites except JP has significantly more N than the other fractions. Sites at the higher end of the productivity gradient have 65% of the N content in the first four fractions (L - Fl-75) whereas the lower productive sites have their N more evenly distributed through the profile.

Phosphorus Content

The P content of all forest floor fractions at the JP site was significantly greater than the P content of the same fractions at the other sites (Table 7). The P content in all fractions tended to decrease as site productivity increased. Similar to N content, within each profile, the F fraction has the highest P content for the more productive sites (NS, Feld, and SI), and at the lowest productivity site (JP) the P content of the O_a fractions (H-250, H-75) tended to be greatest. Although P is distributed through the profile, there is still a shift from the upper fractions (L - Fl-75) of 55-60% of the P at the higher productivity sites, to 40-45% in these same layers at the lower end of the productivity gradient.

Table 6. Nitrogen content (kg/ha) in forest floor fractions corrected for mineral soil contamination.

Forest Floor Fraction	Nelson-Stokes (NS)	Feldun (Feld)	SIPAC (SI)	Clark (Cl)	Jasper-Pulaski (JP)
L	47.3 b* : A**	51.0 b : A	43.9 cd : AB	46.7 c : A	36.3 d : B
F	98.5 a : B	94.9 a : B	125.6 a : A	84.0 a : B	141.0 b : A
Fl	51.7 b : B	45.9 b : B	81.2 b : B	62.5 b : B	156.1 b : A
Fl-75	3.7 d : B	4.7 c : B	7.7 f : B	7.4 f : B	18.7 d : A
H-250	47.7 b : B	43.3 b : B	64.9 bc : B	80.1 a : B	212.8 a : A
H-75	19.0 c : B	13.5 c : B	24.4 def : B	23.8 de : B	43.7 d : A
S-250	22.9 c : B	23.4 c : B	36.4 de : B	37.9 cd : B	88.5 c : A
S-75	23.3 c : AB	12.5 c : B	15.5 ef : B	14.5 ef : B	33.0 d : A
Fraction Totals	314.1	289.2	399.6	356.9	730.1

* Column means with the same lower-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.
** Row means with the same upper-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.

Table 7. Phosphorus content (kg/ha) in forest floor fractions not corrected for mineral soil contamination.

Forest Floor Fraction	Nelson-Stokes (NS)	Feldun (Feld)	SIPAC (SI)	Clark (Cl)	Jasper-Pulaski (JP)
L	4.15 c* : B**	3.56 bc : B	2.79 c : C	2.86 b : C	5.02 de : A
F	8.44 a : B	8.67 a : B	11.36 a : A	7.58 a : B	11.95 bc : A
Fl	4.12 c : B	3.57 bc : B	6.46 b : B	6.07 a : B	13.54 bc : A
Fl-75	0.13 d : B	0.15 e : B	0.28 d : B	0.25 b : B	1.14 e : A
H-250	5.80 b : B	4.66 b : B	6.89 b : B	7.18 a : B	22.11 a : A
H-75	3.23 c : B	2.12 cd : B	2.68 c : B	2.51 b : B	7.64 cd : A
S-250	3.54 c : B	4.37 b : B	2.61 c : B	8.49 a : B	14.93 b : A
S-75	1.18 d : B	1.93 d : AB	1.71 cd : AB	2.08 b : AB	2.50 de : A
Fraction Totals	30.59	29.03	34.75	37.02	78.83

* Column means with the same lower-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.
** Row means with the same upper-case letter are not significantly different at the 5% level using Duncan's Multiple Range Test.

DISCUSSION

The forest floor plays a critical role in nutrient cycling in forest ecosystems. It is the zone where plant organic materials are converted to stable soil organic matter. Understanding forest floor developmental processes is a major step toward understanding nutrient cycling in forest ecosystems. Forest floor development appears to be controlled by several prime factors. Among these are macroclimate variables such as temperature and seasonal moisture availability (Meentenmeyer 1978), microclimate site characteristics (Berg and others 1993, Paul and Clark 1989), and the chemical composition of the detrital inputs to the system (Aber and Melillo 1982, Blair 1988, Melillo and others 1982). The forest floor represents an integration of all these factors. The forest floor characteristics measured in our study represent many years of development. Federer (1984) found that it took approximately 50 years of development for forest floor organic layers to reach a steady-state following disturbance events in northern hardwood forest ecosystems. Thus, it would appear that the forest floor differences observed in these stands are the result of long-term differences in developmental processes observed at the various sites. Likewise, current developmental processes in these stands will be reflected in future forest floor characteristics. The interacting influences of these factors control decomposition rates of litter inputs to forest systems. The rate of decomposition in turn will affect nutrient mineralization/ immobilization patterns (Melillo and others 1989) and may directly affect the development of the forest floor.

The primary purpose of our study was to quantify differences in biomass and nutrients in forest floor organic layers in oak-hickory dominated stands distributed along a productivity gradient. The stands selected differed in soil water availability, and nutrient availability (Table 2). These factors subsequently interact to control the species composition of the stands. While the oak-hickory species group dominated the basal area and litter inputs in each stand, the importance of mesic site species such as sugar maple and yellow poplar increased as soil moisture holding capacity and nitrogen availability increased. Based on the results of previous studies, (Aber and Melillo 1982, White and others 1988) these differences in species composition between the five stands led to litter inputs of differing initial chemical characteristics. This study was designed to determine if the integration of these differing controlling factors subsequently affected forest floor development and N and P storage patterns.

Forest Floor Biomass

This study, and supporting data from previous work (Table 2), found that on an annual basis, the litter inputs into the systems were directly related to potential site productivity. As site quality decreased, annual leaf-fall also decreased in mass and returned a lower total quantity of nitrogen to the site. Below freshly fallen litter, however, biomass of the various layers of the forest floor have an inverse relationship to site productivity. These data are consistent with the conclusion that decomposition is slowed by conditions that do not promote microbial growth or activity (Swift and others 1979). As a result, organic materials tend to accumulate as decomposition rates are reduced.

Several factors may influence forest floor developmental processes. The inherently low soil water-holding capacity of the lower productivity sites would, over time, tend to limit the rate of litter decomposition and favor and accumulation of forest floor biomass (Meentenmeyer 1978). The lower productivity sites are dominated by oak species which produce litter characterized by high lignin/nitrogen ratios (Aber and Melillo 1982, White and others, 1988). The higher productivity sites, while still dominated by oak leaf-fall, have more of a mix of species which Seastedt (1984) hypothesized would stimulate the decomposition of adjacent recalcitrant litter types. On the xeric, nutrient poor sites, annual leaf-fall was composed almost entirely of oak foliage. At the Jasper-Pulaski and Clark sites, oak leaves accounted for 95% and 85% of the annual leaf-fall respectively. Litterfall on the more mesic sites i.e. Nelson-Stokes and Feldun , while still dominated by oaks, this component of the annual litterfall is lower and ranges from 52 - 60%. The differing chemical characteristics of the various species found in the stands may account for the differences in biomass accumulation observed in the forest floor at each site.

Studies conducted throughout the country have demonstrated the recalcitrant nature of oak litter. Oak litter is typically characterized by a chemical composition which leads to subsequent slow decomposition rates. Lignin concentrations in oak litter are usually between 20-25% of the dry weight (Aber and others 1990) while other

associated species such as yellow poplar (15%) (White and others 1988), sugar maple (12%) (Aber and others 1990), red maple (10%), or dogwood (5%) (Blair 1988) have much lower lignin concentrations. In all these studies, oak litter was found to decompose at much slower rates than the associated species with lower lignin concentrations. Although lignin can be degraded by soil microorganisms, the rate of lignin degradation is 1/10th to 1/20th the rate of cellulose degradation (Berendse and others 1987). The early stages of litter decomposition are characterized by rapid weight loss as cellulose is degraded by microorganisms. The late stage of litter decomposition is characterized by very slow rates of decomposition as microbial populations utilize the recalcitrant lignin fraction of the litter. The lignin in the decomposing litter may also serve as one of the primary building blocks of stabilized soil organic matter (Paul and Clark 1989). The combined effects of site environmental conditions that retard the decomposition process and litter inputs that are inherently more resistant to decay may be responsible for the increased forest floor biomass observed on the lower productivity sites.

The relative importance of the various forest floor layers changed with changes in site productivity. On the more productive sites, the litter and fermentation layers contained a higher percentage of the forest floor biomass than on the less productive sites. On the low productivity sites, the biomass of the H-250 and the S-250 were higher. These lower forest floor and mineral soil horizons are composed of organic materials that have progressed to a late stage of decomposition. The greater biomass in these layers may again be related to the chemical composition of the litter. Melillo and others (1982) found that the amount of humus formed from a litter material was directly related to the lignin concentration of the litter. This reinforces the observation that although inputs to the system are lower on the lower productivity sites, the slower decomposition rates on the more xeric sites offsets the lower inputs, and over time, leads to greater biomass accumulation in the lower soil horizons.

Nutrient Concentrations

The combination of the four lower fractions (H-250, H-75, S-250, and S-75), which represent latter stages of decomposition of the forest floor, had relatively more biomass than the combination of the upper forest floor fractions (L, F, Fl, and Fl-75)at the lower productivity sites, but the nitrogen concentrations of these fractions were not different. This result is consistent with Melillo and others (1989) who hypothesized that litter materials of differing initial chemical composition would be converted to a relatively homogenous material during the late stages of decomposition . The differing litter inputs did appear to be converted to a common substrate with a relatively constant nitrogen concentration. These data supported the idea that the fractionation technique presented in this paper was isolating fractions of similar "quality" or organic content across sites.

Nitrogen concentrations in the forest floor profile showed similar trends across all sites. Nitrogen concentrations increased through the first 4 forest floor organic layers (L, F, Fl, and FL-75). Initial litter inputs in these systems were characterized by relatively low nitrogen concentrations and high carbon concentrations (Kaczmarek and others 1994). Litter with high C/N ratios such as these may initially immobilize nitrogen. It is only after decomposition proceeds that a net mineralization of nitrogen occurs (McClaugherty and others 1985). Fractions representing later stages of decomposition (i.e. H-250 - S-75) , had decreased nitrogen concentrations. *In-situ* litter decomposition studies are currently in place in all five of these stands. In these studies mass loss and nutrient concentrations will be determined and these data used to determine the stage of decomposition at which nutrient immobilization/mineralization takes place in the litter. Explanation of the slight "peaks" in N concentration observed at the H-75 layer (at NS, SI, CL), and the S-75 layer (at NS) may come from studies in which particles of smaller sizes where shown to have higher nitrogen concentrations (Cameron and Posner 1979, Young and Spycher 1979). In these studies, all organo-mineral aggregates were dispersed and as particle size decreased from sand to clay, nitrogen concentration increased. Our study did not try to disperse microaggregates, and it is possible that the material that stayed on the 75 um sieve is made up of silt and clay sized microaggregates. These microaggregates would include soil organic matter (SOM), and/or microorganisms and microbial byproducts occluded on or trapped in silts and clays as described by Cameron and Posner (1979), and therefore, would show higher concentrations of nitrogen than the layer above.

Phosphorus showed no strong trends along the productivity gradient. Phosphorus concentrations increased at all sites through the first three fractions (L, F, and Fl) except at the JP site at which there was a slight drop from the L to the F

with further increases in phosphorus concentrations after that fraction. In short-term decomposition studies such as Blair (1988) and Rustad (1994), P was shown to initially decrease in litter. After approximately six months, phosphorus concentrations began to increase as phosphorus was immobilized. These data can be compared to our study in that the first three layers obtained through our separation technique are similar to the stage of decomposition achieved in a 24 month decomposition study . Our results are similar to the results by Blair (1988) and Rustad (1994) except that we did not detect an initial decrease in P concentrations at the higher productivity sites. The JP site is somewhat of an anomaly in that soil N availability is low, but soil P availability is high. Phosphorus concentrations in all plant tissues sampled at this site have always been very high (Kaczmarek unpublished data). In addition, foliage samples collected during the growing season and at autumnal leaf-fall have demonstrated very low P retranslocation at this site. Whereas white oak on the other sites sampled retranslocated 60-70% of their tissue P concentrations, white oak at the JP site retranslocated less than 20% of foliar P (Kaczmarek unpublished data). Dalla-Tea and Jokela (1994) found that P retranslocation in southern pines was inversely correlated with soil P availability. In addition, P dynamics during the decomposition process depended upon soil P availability upon which the trees grew that produced the leaf-fall. High soil P availability increased the inorganic P concentrations in the leaf-fall. This inorganic P was subject to rapid leaching losses early in the decomposition process (Polglase and others 1992). The high P availability at the JP site may be creating similar P dynamics in the litter produced at this site.

Nutrient Contents

Nitrogen and phosphorus storage in the various forest floor layers was dependent upon the biomass and nutrient concentration of the layer. Total nitrogen storage ranged from approximately 300 kg/ ha on the high productivity sites to over 730 kg/ ha at the low productivity sites. As site quality decreased, increasing percentages of the nitrogen in the forest floor are found in the lower fractions (H-250 - S-75), whereas on higher quality sites the trend is reversed. In addition, as site quality decreased, the relative percentage of nitrogen contained in the forest floor humus (represented by H-250) layer increases. This again reinforces the observation that the biomass and hence nitrogen storage capacity of this layer is enhanced under low productivity conditions.

Phosphorus content of all of the forest floor organic fractions at the Jasper-Pulaski site was 79 kg/ha. Phosphorus contents for all other sites were 29 to 37 kg/ ha and did not show any strong trends by site productivity. The high phosphorus content at Jasper-Pulaski was a function of the high biomass and elevated phosphorus concentrations in the organic material sampled. The majority of the phosphorus at these sites was contained in the fermentation, floating, and humus fractions.

SUMMARY

The results of this study indicate that forest floor organic layers in these stands contain appreciable pools of nitrogen and phosphorus. The results also indicate that forest floor development follows different patterns in sites of differing productivity. In mesic, nutrient rich stands, leaf and litterfall occur at high levels, but the stand environmental conditions that favor this high litter production also favor relatively rapid decomposition rates. As a result, relatively low quantities of biomass remain in the O_e and O_a layers. On the less productive sites, water and nutrient limitations favor almost complete stand domination by oak species. Annual litter inputs are low, but subsequent decomposition of these resistant materials is slow especially under the xeric, nutrient poor conditions characterized by our study sites. These conditions result in relatively high biomass in the O_e and O_a layers. Nitrogen and phosphorus concentrations also change throughout an individual forest floor profile. Beginning with freshly fallen litter material, nitrogen and phosphorus concentrations increase in the decaying tissues. It is only in the latter stages of decomposition that nutrient concentrations decrease. The potential impact of the forest floor organic layers should be considered when management decisions concerning these study areas are made. On the more productive sites, the forest floor layers contain relatively small amounts of nutrients. On the less productive sites however, the amount of these nutrients is relatively large. These results may indicate that the forest floor organic layers serve as a nutrient conserving mechanism and any catastrophic event that would destroy the forest floor organic layers could drastically alter nutrient recycling characteristics of these systems. It also appears that the rate of nutrient turnover in these systems

will outweigh the absolute pool size of nutrients in determining nutrient availability. While the sites with a lower potential site productivity have a greater pool of N in the forest floor, the site conditions may slow the turnover of this pool. *In-situ* N mineralization studies are currently in place in each of these stands to assess N mineralization and nitrification patterns across the productivity gradient. Future research on the forest floor layers will focus on the carbon and lignin they contain as well as assessing nutrient availabilities and microbial biomass contributions to nutrient cycling in these stands.

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PLANT AND SOIL NUTRIENTS IN YOUNG VERSUS MATURE CENTRAL APPALACHIAN HARDWOOD STANDS

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Abstract: Most models of forest succession and forest recovery following disturbance predict changes in nutrient availability. The purpose of this study was to compare soil and herbaceous layer plant nutrients between two young (~20 yr) and two mature (~80 yr) forest stands on Fernow Experimental Forest, Parsons, West Virginia. All sampling was carried out within 15 circular 0.04-ha sample plots per watershed, for a total of 30 plots for each age category. All vascular plants ≤1 m in height (herbaceous layer) were identified and estimated visually for cover in each of 10 circular 1-m^2 sub-plots. All above-ground portions of plants in the two sub-plots in each sample plot with the greatest herb layer cover were harvested and analyzed for nutrient content. In addition, a soil sample was taken to a 10-cm depth from each harvest sub-plot. Soils were analyzed for organic matter, texture, pH, and extractable nutrients. There were no significant differences related to stand age for soil pH, organic matter, cation exchange capacity, or any of the extractable nutrients. With the exception of significantly higher herb layer N for the young stands, there were few differences in herb layer tissue nutrients between young and mature stands. There were, however, differences in correlations among soil nutrients and other soil variables (e.g., texture and organic matter) which varied with stand age, with extractable nutrients more highly correlated with organic matter and texture in the young stands than in the mature stands. Furthermore, herb layer tissue nutrient concentrations were correlated significantly with soil organic matter for virtually all nutrients in young stands; in contrast, none were correlated with soil organic matter in mature stands. These results suggest minimal change in nutrient availability between 20 and 80 years of forest recovery from disturbance. Data in this study further emphasize the importance of soil organic matter decomposition as a source of available nutrients in these hardwood forests following disturbance and suggest that this importance might decrease with stand age. Finally, these results are consistent with the hypothesis that herb layer dynamics are controlled by nutrient availability early in succession, but that other factors, such as light, become more important later in succession.

INTRODUCTION

Most current models of secondary forest succession and forest response to disturbance predict measurable changes in nutrient availability over time (Bormann and Likens 1979, Vitousek and Reiners 1975). Although these patterns of change will vary among ecosystems and degrees of disturbance, it generally is accepted that nutrients will increase in availability and mobility immediately following a disturbance, such as forest harvesting, which usually results in ching increases and nutrient loss via streamflow. As regrowth occurs, however, plant root uptake increases rapidly, resulting in decreased nutrient leaching and increased nutrient storage in plant biomass. Thus, it is reasonable to generalize that nutrient availability increases for a short period following forest disturbance and that it decreases subsequently through time. Studies of stream chemistry response to forest harvests on montane watersheds of the eastern U.S. largely have supported these generalizations (Aubertin and Patric 1974, Bormann and Likens 1979, Kochenderfer and Wendel 1983, Swift and Swank 1981).

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Other research has focused on forest floor organic matter and its response to forest harvests (e.g., Covington 1981); fewer studies have looked directly at soil nutrients. Although forest successional changes in N are fairly well understood (Vitousek and others 1989), less is known about P and the base cations Ca, Mg, and K. These nutrients are not generally considered as frequently growth-limiting as is N, but limitations can occur, especially in forest soils developed from parent materials low in these nutrients (Jenny 1981), such as the highly weathered soils of the pine forests of the southeastern Coastal Plain (Gilliam 1991, Gilliam and Richter 1991), or the deep sands of the Adirondacks in New York (Stone and Kszystyniak 1977). In contrast to N change through time, which is tied into somewhat predictable changes in organic matter decomposition, it may be much more difficult to generalize about ecosystem changes in P and nutrient cations through succession because they are influenced both by decomposition rates of organic materials and by weathering rates of primary and secondary soil minerals.

Although difficult to assess empirically (Chapin and Van Cleve 1991), nutrient availability generally is defined in terms of amounts of nutrients present in the soil in chemical forms which allow for plant root uptake. Although N availability is often measured via incubation techniques, cation availability usually is measured as exchangeable amounts in soil extract solutions (Binkley and Vitousek 1991). Measurements of plant tissue concentrations also have been used to assess soil fertility and nutrient availability (Binkley and Vitousek 1991). This paper examines nutrient availability as both amounts of extractable nutrients from the mineral soil and tissue concentrations in plant material.

The herbaceous layer, defined here as vascular plants $\leq$1 m in height, is an important stratum of forest ecosystems, playing a significant role in initial competition among juveniles (seedlings and sprouts) of potential overstory canopy tree species (Gilliam and others 1994). Furthermore, the herb layer has been shown to be quite sensitive to a variety of forest disturbances, including gap openings (Moore and Vankat 1986) and forest management practices (Duffy and Meier 1992, Gilliam and Christensen 1986, Gilliam and Turrill 1993). Perhaps more important in the context of changes in nutrient availability, the herb layer also has been used as an indicator of edaphic factors and forest site quality (Cserep and others 1991, Pregitzer and Barnes 1982, Strong and others 1991). Reiners (1992) reported on 20 yr of vegetation recovery following deforestation and herbiciding on Watershed 2 at Hubbard Brook and found 6-8 fold increases in herb layer content of N, Ca, Mg, and K. In contrast, the shrub layer increased only 3-5 fold for these nutrients over the same 20-yr period (Reiners 1992).

The purpose of this study was to compare herbaceous layer and extractable soil nutrients (N, P, Ca, Mg, and K) between watersheds supporting young, clearcut (~20-yr old) vs. mature ~80-yr old) stands of central Appalachian mixed hardwood forests. We were also interested in examining soil-plant nutrient interactions and how these interactions might vary with stand age.

METHODS

Study Site

Research was done on four contiguous watersheds of the Fernow Experimental Forest (FEF), a 1900-ha area of montane hardwood forests within the unglaciated Allegheny Plateau, located in Tucker County, West Virginia (39°03'N, 79°49'W). WS7 and WS3 support stands which were approximately 20 yr old and even-aged (following clearcutting) at the time of sampling and will be referred to as "young." These stands are dominated primarily by black cherry (*Prunus serotina* Ehrh.) and yellow poplar (*Liriodendron tulipifera* L.). WS13 and WS4 support stands which were older than 75 yr at the time of sampling and will be referred to as "mature." They are mixed-age, being dominated by sugar maple (*Acer saccharum* Marsh.) and northern red oak (*Quercus rubra* L.), and originated from natural regeneration from the heavy cutting which characterized the period of railroad logging from ~1880 to 1930 (Marquis and Johnson 1989). Herbaceous layer species have been described in previous papers (Gilliam and Turrill 1993, Gilliam and others 1994).

Field Sampling

Fifteen semi-permanent circular plots were established within each of the study watersheds, for a total of 60 sample plots for the study. Plots were established by locating plot centers throughout each watershed to include the extremes of elevation and aspect. A radius of 11.3 m was measured from each plot center for a total area of 0.04 ha per plot.

Methods of herbaceous layer and soil sampling also has been described in those papers, but is summarized briefly here. Within each sample plot all vascular plants within 10 1-m^2 circular sub-plots were identified to species and estimated visually for cover (total of 600 sub-plots for the study) using the technique of Gilliam and Christensen (1986). This method has been shown to be both reproducible and capable of accurate prediction of herb layer biomass (Gilliam and Turrill 1993). The two sub-plots with the greatest percentage of herb layer cover in each were chosen as "harvest" sub-plots, within which all above-ground herb layer material was collected. Harvesting was confined to only the two high-cover sub-plots in each plot to ensure enough herb tissue biomass for nutrient analysis of individual species. Although this criterion potentially confounds cover-nutrient relationships among watersheds, it allowed for a full range of cover values (from 1 to ~80%) for harvested species (Gilliam and Turrill 1993). Furthermore, there was a good correlation between the mean cover of the two sub-plots per plot and mean cover of the entire plot (based on all 10 plots). In other words, relatively high-cover sample plots had relatively high-cover harvest sub-plots, and visa versa. Therefore, using high cover as a selection criterion for herb layer harvests does not appear to create problems for comparison among watersheds. Finally, a sample of mineral soil was taken to a 10-cm depth from each of the two harvest sub-plots following removal of organic forest floor material.

Sample Analysis

Herb layer samples from each of the 120 harvest sub-plots were separated into species, oven-dried at 50°C, weighed, and ground in a Wiley mill to pass a 1-mm screen. Plant tissue (and soil--see below) samples were shipped to the University of Maine (U. M.) Soil Testing Service and Analytical Laboratory for macronutrient analysis (N, P, Ca, Mg, and K). Total plant N was measured as Kjeldahl N. All other nutrients were measured with plasma emission spectrophotometry after dry-ashing plant material and extraction with HCl and HNO_3.

Mineral soil samples were sieved to pass a 2-mm screen, air-dried, and shipped to the U. M. laboratory for analysis, including pH (1:1 w/v, soil/H2O), 1N KCl-extractable Ca, K, Mg, and P (plasma emission), 1N KCl-extractable NO_3 and NH_4 (flow-injection colorimetry), and soil organic matter (loss-on-ignition). Particle-size (texture) was determined on each soil sample at the laboratory at Marshall University using the hydrometer method of Bouyoucos (1951).

Data Analysis

Data analysis for this study was based on mean plot values (i.e., not sub-plot values) of all soil and plant variables. Stand age class means of all variables were calculated as the average of 30 plots (15 plots in each of two watersheds per stand age class) and were compared (young vs. mature) using Student's t-test (SAS 1982, Zar 1984). Within each age class, relationships among soil and plant variables (soil vs. soil, plant vs. plant, and plant vs. soil) were determined with Pearson product-moment correlation analysis (SAS 1982).

RESULTS

There were no significant differences between stand age classes for any of the measured soil variables, including organic matter, cation exchange capacity, pH, and extractable nutrients (Table 1). There were also no differences between age classes for P, Ca, Mg, and K concentrations in the herbaceous layer (Table 2). However, the herb layer of the young stands was significantly higher ($P<0.01$) in N than that of mature stands (Table 2).

Table 1. T-test comparisons of soil variables of young vs. mature stands of the Fernow Experimental Forest, WV. OM=organic matter; CEC=cation exchange capacity. NS indicates no significant differences between stand age classes at P<0.05. Values in parentheses are one standard error of the mean.

Stand Age	OM (%)	CEC (meq/kg)	pH	NO_3	NH_4	P	Ca	Mg	K
				mg/kg					
Young	13.8	45.5	4.39	1.3	2.1	0.8	12.6	2.1	2.3
	(0.7)	(4.6)	(0.06)	(0.2)	(0.2)	(0.4)	(5.0)	(0.4)	(0.2)
	NS	NS	NS	NS	NS	NS	NS	NS	NS
Mature	12.6	40.1	4.32	0.9	1.9	0.4	6.4	2.4	2.2
	(0.4)	(1.2)	(0.05)	(0.1)	(0.1)	(0.0)	(1.1)	(0.7)	(0.1)

Table 2. T-test comparisons of herbaceous layer tissue nutrients of young vs. mature stands of the Fernow Experimental Forest, WV. Significant differences (P<0.01) between stand age classes indicated by *. NS indicates no significant difference between age classes at P<0.05. Values in parentheses are one standard error of the mean.

Stand Age	N	P	Ca	Mg	K
	mg/kg				
Young	2.35	0.16	0.62	0.22	2.68
	(0.07)	(0.01)	(0.08)	(0.01)	(0.26)
	*	NS	NS	NS	NS
Mature	1.99	0.15	0.75	0.24	2.21
	(0.09)	(0.01)	(0.06)	(0.02)	(0.19)

Numerous significant correlations (nearly 60% of total possible combinations) existed among soil variables for the young stands (Table 3). Notable variables with which there were several significant correlations were texture classes (sand and silt) and organic matter. In contrast, neither extractable P nor extractable NH_4 were correlated with any other soil variables (Table 3). Soils of mature stands exhibited far fewer significant correlations (<20% of the total) compared to young stand soils (Table 4).

In contrast to differences between stand age classes in patterns of soil variable correlations, correlations among herb layer nutrient concentration were quite similar between young and mature stands. All plant nutrients were correlated significantly with each other for mature stands, and all but Ca:K were significant for young stands (Table 5).

In comparing plant vs. soil nutrients between stand ages, the young stand exhibited significant correlations for the nutrient cations, but neither N nor P; the mature stand had significant correlations for P, Ca, K, and plant N vs. soil NO_3 (Table 6). Finally, all plant nutrients except N were correlated significantly with soil organic matter in the young stands, whereas no plant nutrients were correlated with organic matter in mature stands.

Table 3. Correlations among soil variables of young stand plots. Values shown are Pearson product-moment correlation coefficients (r) significant at P<0.05.

	Sand	Silt	Clay	OM	Ca	K	Mg	P	NO_3	NH_4	pH
Silt	-0.89**										
Clay	-0.70**	---									
OM	0.56*	-0.41	-0.53*								
Ca	0.46	-0.42	---	0.66**							
K	0.56*	-0.42	-0.52*	0.77**	0.72**						
Mg	0.47*	-0.39	---	0.73**	0.97**	0.76**					
P	---	---	---	---	---	---	---				
NO3	0.50*	-0.44	---	0.78**	0.68**	0.76**	0.73**	---			
NH4	---	---	---	---	---	---	---	---	---		
pH	0.55*	-0.49*	-0.39	0.51*	0.75**	---	---	---	0.40	---	
CEC	0.44	-0.37	---	0.68**	0.98**	0.69**	0.98**	---	0.72**	---	0.61**

*indicates correlation coefficient significant at P<0.01.
**indicates correlation coefficient significant at P<0.001.

Table 4. Correlations among soil variables of mature stand plots. Values shown are Pearson product-moment correlation coefficients (r) significant at P<0.05.

	Sand	Silt	Clay	OM	Ca	K	Mg	P	NO_3	NH_4	pH
Silt	-0.82**										
Clay	-0.47*	---									
OM	0.38	-0.42	---								
Ca	---	---	---	---							
K	---	---	---	---	0.59**						
Mg	---	---	---	---	0.61**	---					
P	---	---	---	0.44	---	---	---				
NO3	---	---	---	---	---	---	---	0.67**			
NH4	---	---	---	---	---	---	---	---	---		
pH	---	---	---	---	0.42	0.48*	---	---	---	---	
CEC	---	---	---	---	0.74**	---	0.46*	---	---	---	---

*indicates correlation coefficient significant at P<0.01.
**indicates correlation coefficient significant at P<0.001.

Table 5. Correlations among herbaceous layer nutrient concentrations of young and mature stand plots. Values shown are Pearson product-moment correlation coefficients (r) significant at P<0.05.

	N	P	Ca	Mg
Young stands				
P	0.63**			
Ca	0.49*	0.62**		
Mg	0.71**	0.38	0.58**	
K	0.56*	0.38	---	0.67**
Mature stands				
P	0.83**			
Ca	0.53*	0.48*		
Mg	0.88**	0.64**	0.54*	
K	0.88**	0.67**	0.58**	0.89**

*indicates correlation coefficient significant at P<0.01.
**indicates correlation coefficient significant at P<0.001.

Table 6. Plant-soil nutrient interactions of young and mature stands of Fernow Experimental Forest, WV. Values shown are Pearson product-moment correlation coefficients (r) for indicated pairs significant at P<0.05.

Variables			Stand Age	
Plant	vs.	Soil	Young	Mature
N	vs.	NO_3	---	0.63**
N	vs.	NH_4	---	---
P	vs.	P	---	0.63**
Ca	vs.	Ca	0.78**	0.43
Mg	vs.	Mg	0.57*	---
K	vs.	K	0.44	0.42
N	vs.	OM	---	---
P	vs.	OM	0.48*	---
Ca	vs.	OM	0.75**	---
Mg	vs.	OM	0.50*	---
K	vs.	OM	0.37	---

*indicates correlation coefficient significant at P<0.01.
**indicates correlation coefficient significant at P<0.001.

DISCUSSION

Based on soil and plant tissue nutrient comparisons alone, there appears to have been little long-term change in nutrient availability over time following forest disturbance at this site. Soil variables were virtually identical between young and mature stands. In general these forest soils were acidic, with moderate levels of organic matter (~13%) and a relatively low cation exchange capacity (~42 meq/kg). It should be emphasized that the extractable N data represent pools of available N and not fluxes. Current research using in situ incubations ("buried bag" technique) on many of these same plots indicate that fluxes (e.g., uptake) are >10-fold higher than available N pools (Gilliam and Adams, submitted manuscript).

Herb layer tissue nutrient data are indicative of small decreases in N availability with stand development over time. Because these data are for the herb layer as an entire stratum (i.e., all species combined), the significant difference in plant N may have resulted from stand age-related differences in herb layer species. However, these stands are remarkably similar with respect to herb layer species composition, regardless of stand age (Gilliam and Turrill 1993, Gilliam and others, submitted manuscript), with the possible exception of higher fern cover on WS7 (Gilliam and others 1994). Herb layer nutrient concentrations for FEF were similar to other hardwood forests for N and P, but lower for Ca, Mg, and K (Grigal and Ohmann 1980, Peterson and Rolfe 1982, Siccama and others 1970). These differences may be related to the sandstone parent materials of the FEF watersheds. Concentrations of herb layer nutrients at FEF are well within the range of values reported as adequate for crop plants (Chapin and Van Cleve 1991).

Although correlation analysis is more appropriate for generating hypotheses than for testing them, such analyses can be useful in describing general patterns of relationships of soil and plant nutrients. For example, the numerous significant correlations among soil variables of the young stand plots suggest that there may a single factor determining, or controlling, overall soil fertility of these young stands. Furthermore, the general lack of correlations for the mature stand plots indicate that this factor is not as prevalent for the mature stands as it is for the young stands.

Soil organic matter is the variable which is consistently the most highly correlated with extractable soil nutrients in the young stands (Table 3), suggesting that organic matter is an important factor determining overall fertility of these soils. Indeed, it was significantly correlated ($P<0.01$) with all other young stand soil variables except extractable P and NH_4 (Table 3). Significant correlation ($P<0.001$) between organic matter and cation exchange capacity further support this contention, and indicate that exchange sites of these soils are dominated by organic colloids (Tate 1987). In sharp contrast in mature stand soils, among the nutrients organic matter was correlated only with extractable P, and it was not correlated with cation exchange capacity.

Surprisingly, there were numerous significant correlations among plant nutrients for both stand age classes. Other studies have reported fewer significant correlations among plant nutrients, but ones which suggest either biochemical or functional groups. For example, using nutrient data from 110 plant species, Garten (1978) found significant correlations for the following plant nutrient pairs: N:P, Mg:K, and Ca:Mg. He concluded that these pairings were related to biochemical functions within the plant, e.g., nucleic acids-proteins for N:P. Gilliam (1988) found significant correlations for most of these same nutrient pairs and for K:P. He concluded that the latter was the result of limitation of herb layer plant growth from both nutrients in the highly weathered soils of a southeastern U.S. pine forest. Thus, the numerous correlations found in the present study are certainly notable, though difficult to interpret.

Young and mature stands differed slightly from each other with respect to correlations between plant tissue nutrient concentrations and extractable forms of the same nutrients in mineral soil (Table 6). However, correlations between plant nutrients and soil organic matter varied greatly with stand age. Virtually all nutrients were correlated in young stands, whereas none were correlated in mature stands (Table 6). The numerous significant correlations in young stands further illustrates the importance of soil organic matter in mediating nutrient availability following forest disturbance. Organic matter serves both as a nutrient source via decomposition and a cation exchange source.

CONCLUSIONS

Although differences between stand age classes in soil organic matter/plant nutrient correlations might suggest that this importance decreases with stand age, this explanation is probably unlikely considering the nature of the soils of these watersheds (i.e., that they are derived from low-nutrient sandstone parent materials). Because soil organic matter plays such an important role in determining fertility of these sandy soils, stand age-related differences in correlations of herb layer nutrient concentration with organic matter may arise from successional changes in factors limiting herb layer dynamics. Significant correlations would suggest soil-mediated limitations (e.g., nutrient availability), whereas lack of correlation would suggest limitations from other sources. Gilliam and Turrill (1993) hypothesized that the herb layer of these forests are nutrient-limited early in succession when light availability is relatively high and uniform in the stand, and that the herb layer becomes more limited by light availability later in succession following canopy closure and stratification. Although certainly not conclusive, data presented in this paper are consistent with this hypothesis.

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NUTRIENT BUDGETS OF TWO WATERSHEDS ON THE FERNOW EXPERIMENTAL FOREST

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Abstract: Acidic deposition is an important non-point source pollutant in the Central Appalachian region that is responsible for elevated nitrogen (N) and sulfur (S) inputs to forest ecosystems. Nitrogen and calcium (Ca) budgets and plant tissue concentrations were compared for two watersheds, one that received three years of an artificial acidification treatment and an adjacent reference watershed, both located on the Fernow Experimental Forest, West Virginia. Treatments consisted of ammonium sulfate fertilizer applied aerially three times per year at an annual rate of 61 kg S ha^{-1} and 54 kg N ha^{-1}. Trees of four species (*Betula lenta* L., *Prunus serotina* Ehrh., *Acer rubrum* L. and *Liriodendron tulipifera* L.) were harvested for biomass and nutrient determinations. Some tree species on the treated watershed showed elevated N and decreased Ca levels in some tissues, particularly foliage, but no consistent pattern for any species or tissue component was found. Compared to other watersheds in the central and southern Appalachians, Fernow watersheds are losing more N in streamflow. This loss is almost solely in the form of NO_3, which appears to bring about increased leaching of Ca from watersheds.

INTRODUCTION

During its initial 10-year existence, the National Acid Precipitation Assessment Program (NAPAP) supported a great deal of air pollution research, primarily related to acidic deposition, and its effects on structures, human health, and ecosystems. Although much was learned through NAPAP, many unanswered questions remain, and acidic deposition continues. Nitrogen (N) deposition is expected to increase, despite recent Clean Air legislation (Aber and others 1993), and although reduced sulfur (S) deposition has been reported in some parts of the U.S. (Baier and Cohn 1993), in other regions no change has been detected. The effects of acidic deposition on central Appalachian hardwood forests are not well-understood. Hypothesized effects include nitrogen saturation, altered susceptibility to pests and pathogens, increased tree mortality, and increased growth. None of these hypotheses have been adequately tested, however. Ecosystem level studies can improve our ability to make predictions regarding the effects of acidic deposition on the health and long-term sustainability of central Appalachian forests.

One method to evaluate acidic deposition effects on forest ecosystems is whole-watershed manipulation, i.e. applying a potentially acidifying agent to one watershed and comparing the results with an untreated reference watershed. Because of their expense, whole-watershed experiments are not often replicated. Comparisons between watersheds are therefore based on pseudoreplicated measurements (Hargrove and Pickering 1992). Nonetheless, much can be learned with careful interpretation, and numerous paired watershed studies have been the basis for meaningful ecosystem research (Likens and others 1977, Swank and Crossley 1988). The objective of this paper is to describe and compare nutrient and biomass budgets on two experimental watersheds on the Fernow Experimental Forest, and to evaluate the early effects of an artificial acidification treatment on nutrient cycling. This paper will focus on N and calcium (Ca) because of the recent concerns about N saturation of forest ecosystems (Aber and others 1993) and potential Ca deficiencies, and implications for forest health.

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METHODS

Description of Watersheds

Two experimental watersheds were used for this study. Watershed 3 (WS3) is the treatment watershed, and watershed 7 (WS7) serves as a control or reference watershed. The watersheds are located on the Fernow Experimental Forest near Parsons, West Virginia, USA (39° 3' 15" N, 79°49' 15" W). The Fernow is located on the unglaciated Allegheny plateau of the Appalachian mountains and is characterized by steep slopes and shallow soils (<1m). Precipitation is distributed evenly between dormant and growing seasons and is among the most acidic in the United States. Average annual pH is 4.20, but pH values below 4.0 are common in summer (Edwards and Helvey 1991). The predominant soil type on the experimental watersheds is Calvin channery silt loam (loamy-skeletal, mixed, mesic Typic Dystrochrept) underlain with fractured sandstone and shale of the Hampshire formation (Losche and Beverage 1967). The experimental watersheds are drained by intermittent, second-order streams. Stands on both watersheds have similar species composition and originated at the same time (1969). Prior to 1969, WS7 was maintained barren with herbicides (Patric and Reinhart 1971), while WS3 was clearcut and permitted to regrow without interference. Other watershed characteristics are given in Table 1.

Table 1. Characteristics of two experimental watersheds on the Fernow Experimental Forest, West Virginia.

	Watershed	
	3	7
Age (yr)	24	24
Stand density (trees ha $^{-1}$)	537.5	376.8
Basal area (m^2 ha^{-1})	3.87	3.27
Area (ha)	34.3	24.2
Aspect	S	ENE
Minimum elev. (m)	735	725
Maximum elev. (m)	860	855
Average annual precipitation (mm)	1480	1417
Average annual streamflow (mm)	666	882
Dominant tree species	Black cherry	Black birch
	Red maple	Red maple
	Black birch	Sugar maple
	American beech	Black cherry

Methods and Measurements

Ammonium sulfate fertilizer was applied to WS3 three times per year by helicopter beginning in 1989. March and November applications consisted of 33.6 kg of fertilizer per hectare, which corresponds to 8.1 and 7.1 kg ha^{-1} of S and N, respectively. Each July we applied 100.8 kg fertilizer per hectare, or 24.4 and 21.2 kg ha^{-1} S and N, respectively. Multiple applications per year were used to more closely mimic seasonal variations in chemical inputs. These application rates were approximately double the amount of S and N deposited on the watersheds in throughfall, which we believe to be a good estimate of bulk deposition. Consequently, the total amount of S and N deposited annually on the treatment watershed was 60.5 kg S (ambient S + 40.6 kg from treatment) and 53.8 kg N (ambient N + 35.4 kg from treatment) per hectare, or approximately three times that received by the reference watershed.

In July 1991, total aboveground portions of five trees from each of four species (black cherry, red maple, black birch and yellow-poplar: *Prunus serotina* Ehrh., *Acer rubrum* L., *Betula lenta* L., and *Liriodendron tulipifera* L., respectively) were sampled from both WS3 and WS7. Only dominant and codominant trees were selected for nutrient

determinations. (For biomass determinations, an additional six trees per watershed per species were selected to insure representative sampling of smaller trees.) All leaves were removed from the felled trees, weighed, then subsampled for nutrient analysis and moisture content determination. Each tree was limbed and divided into stemwood (bole to bottom of live crown), top-wood (remainder of bole), and small and large branches (<1 cm diameter and >1 cm diameter, respectively); all constituent parts were weighed in the field. Subsamples were collected for nutrient analysis and moisture content determinations. Each subsample was weighed to the nearest 0.1 g, dried at 70° C and reweighed. Dead wood biomass was determined on 100 square-milacre plots. On each plot, standing dead and down dead biomass were weighed separately and subsamples collected for nutrient and moisture content determinations. Soil nutrients and herbaceous layer biomass and nutrient methods were described by Gilliam and others (1994).

Aboveground biomass was measured on 0.004 ha (.01 A) plots and stand biomass for each watershed was calculated using biomass equations of Brenneman and others (1978). For those species for which Brenneman and others had no equation, an equation for a species of similar specific gravity was used. Watershed foliar biomass was estimated from litterfall collected from 25 1-m^2 litterfall traps per watershed. Root biomass was estimated from four root cores (45 cm depth) collected from each of 25 plots per watershed during May and September 1991. Roots were washed from the cores, separated into fine and coarse roots, dried (70°C) and weighed. Samples were composited by watershed, horizon and size class to provide sufficient tissue for nutrient analyses. Roots were not separated by species.

Tissue samples were analyzed at the Plant and Soil Analysis Laboratory at the University of Maine. Total N was determined by block digestion using a sulfuric acid-hydrogen peroxide solution, and analyzed using a Wescan 360 ammonia analyzer (Wescan Instruments, Santa Clara, CA)[2]. To determine total Ca, ground tissue samples were ashed at 550°C for 5-6 hrs, dissolved in 50% HCl, and analyzed using inductively coupled plasma emission spectrometry.

Nutrient budgets were constructed using mean watershed nutrient concentrations, multiplied by estimated watershed biomass of each component. Soil nutrient pools were calculated for a 60 cm soil depth. Average root biomass (fine + coarse roots) was calculated for each watershed. Herb layer values were calculated from data of Gilliam and Turrill (1993) and Gilliam and others (1994).

Annual (water year, May 1 - April 30) nutrient exports in stream water were calculated from streamflow volume and weekly ionic concentrations of grab samples to relate to watershed nutrient pools. Streamflow measurement, sampling techniques and analytical procedures were described by Edwards and Kochenderfer (1993) and Edwards and Wood (1993). Effects of the acidification treatment on stream and soil water chemistry were reported by Edwards and Wood (1992) and Adams and others (1993).

Watershed nutrient budgets were calculated for treatment comparisons. Mean mass, concentration and content were also calculated for each of the 4 tree species on each watershed and compared using Student's t tests.

RESULTS AND DISCUSSION

Biomass and Nutrient Budgets

Tree biomass varied between the two watersheds (Figure 1). The treatment watershed, WS3, had greater aboveground woody biomass (stems, tops and large branches) than the control, WS7. Because of different pre-treatment histories, the difference in total tree aboveground biomass between WS3 and WS7 can not be attributed to the acidification treatment. The vegetation on WS7 originated mostly from seed, going through the grass - herbaceous - semi-woody - woody plant successional phases (Kochenderfer and Wendel 1983), while the vegetation that developed on WS3

[2]The use of trade, firm, or corporation names in this publication is for the information and convenience of the reader. Such use does not constitute an official endorsement or approval by the U.S. Department of Agriculture or the Forest Service of any product or service to the exclusion of others that may be suitable.

originated mainly from stump sprouts, getting a "head start" on biomass accumulation. Moreover, it is unlikely that such a large treatment effect on woody biomass (approximately 12 of the 17 mt ha^{-1} difference between the two stands is in stemwood) would be evident after only three years of fertilization treatment. Changes in production of foliage or small branches have been reported as a short-term growth response to N fertilization (Auchmoody and Smith 1977, Brix and Ebell 1969, Carlson and Preisig 1981). However, foliar biomass did not differ between the two watersheds, and small branch biomass was slightly lower on the treated watershed. The total root biomass was less on WS3 than on WS7.

Total biomass of dead wood was greater on WS3 (Figure 1). Because WS7 was cut earlier (1964-67) than WS3 (1969) and then maintained barren for 5 years, surface temperatures were higher, resulting in greater rates of decomposition (Mattson and others 1989). Also, WS7 was cut several years prior to WS3, allowing more time for woody slash to decompose. Therefore biomass differences between the watersheds are not attributed to fertilization.

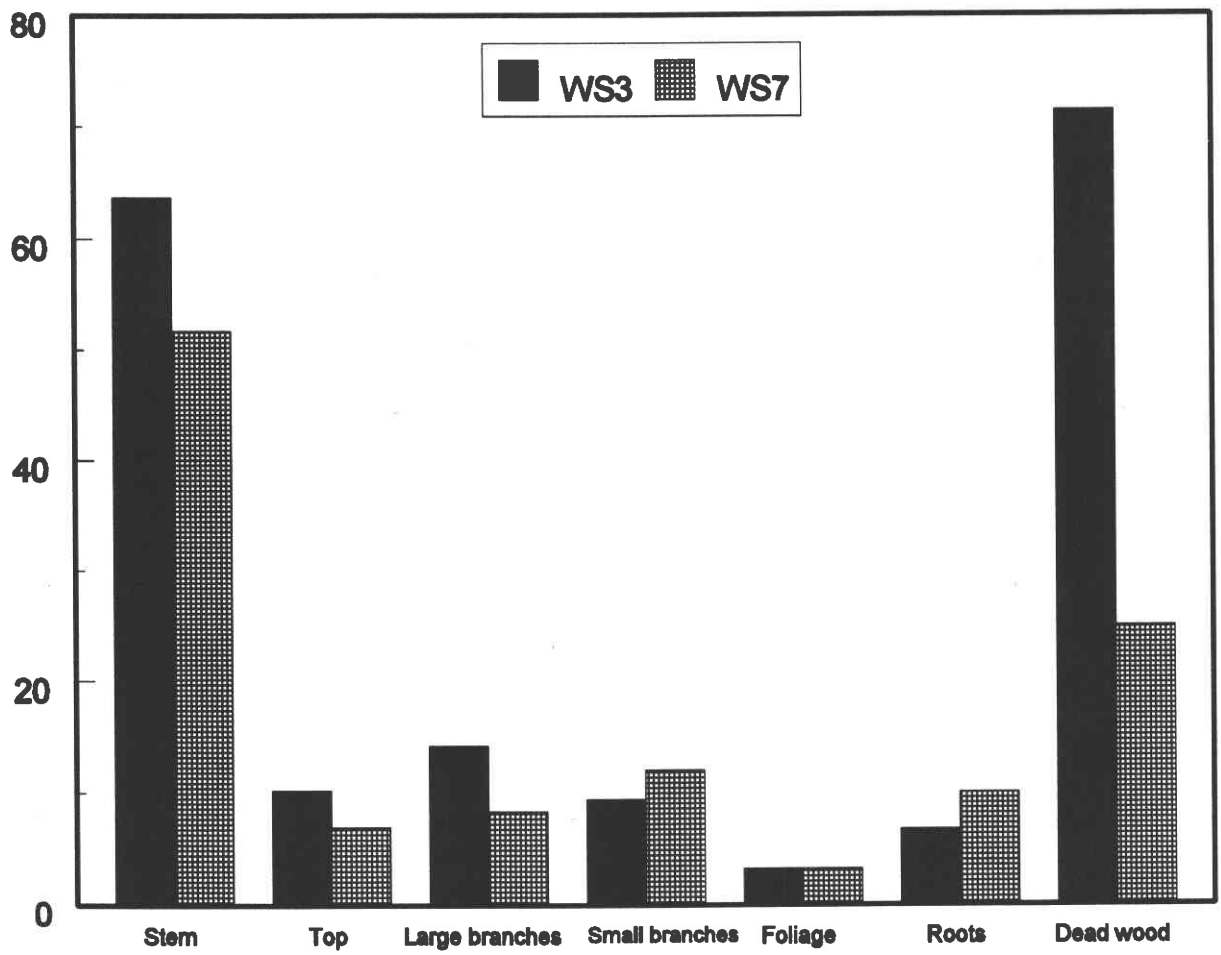

Figure 1. Biomass (mt ha^{-1}) by tree component for 2 experimental watersheds on the Fernow Experimental Forest, West Virginia.

Despite these confounding factors, some observations about the acidification treatment can be made. Aboveground N content was greater for WS3, except in the herb layer (Figure 2). The large herb layer N content in WS7 is due to a large fern community on WS7 which results in twice the herb biomass of WS3 (Gilliam and others 1994). Tissue N concentrations for the herb layer did not differ significantly between the two watersheds, nor did soil extractable N pools or pH differ significantly (Gilliam and others, 1994). Aspect did not significantly affect the response of small stands/watersheds to fertilization (Edwards and others 1991), therefore this is not an alternative explanation. Overall, it did not appear that the treatment had much effect on soil N and vegetation N levels at the watershed scale. Three years after the initiation of treatment, streamwater export of N from WS3 was not significantly different from WS7, although nitrate export from WS3 was significantly greater than that of another reference watershed (Adams and others 1993). This can be explained by examining the differences in pre-treatment N export (Figure 3). For six years prior to treatment, WS3 export of N was consistently less than that from WS7. Within 3 years of initiation of treatment, N export from WS3 increased to a level equal to that of WS7. Because of high variability, this change in trend is not statistically significant, but does agree with results of Adams and others (1993) and Edwards and others (1991) that report significant changes in soil water N and stream water N.

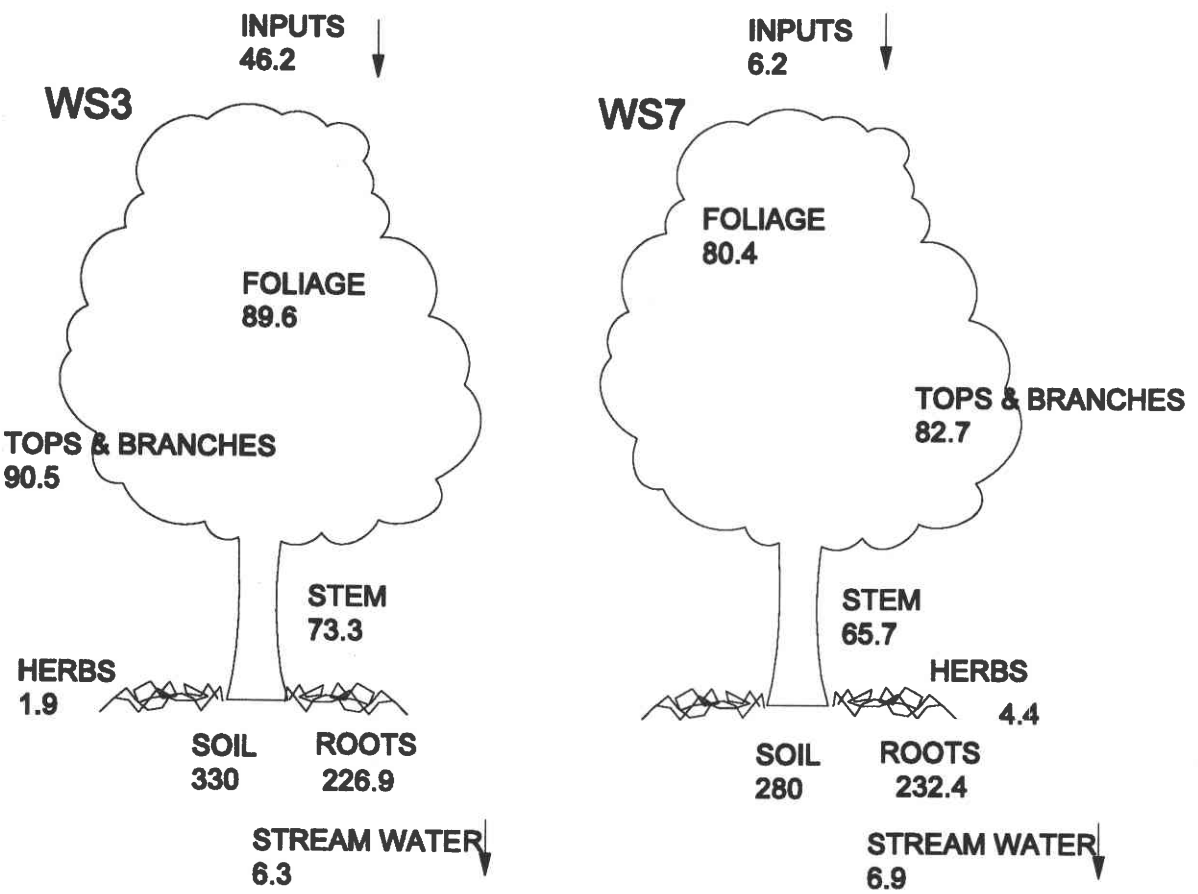

Figure 2. July nitrogen budgets for WS3 (treatment watershed) and WS7 (control watershed), Fernow Experimental Forest. Values are kg ha^{-1}.

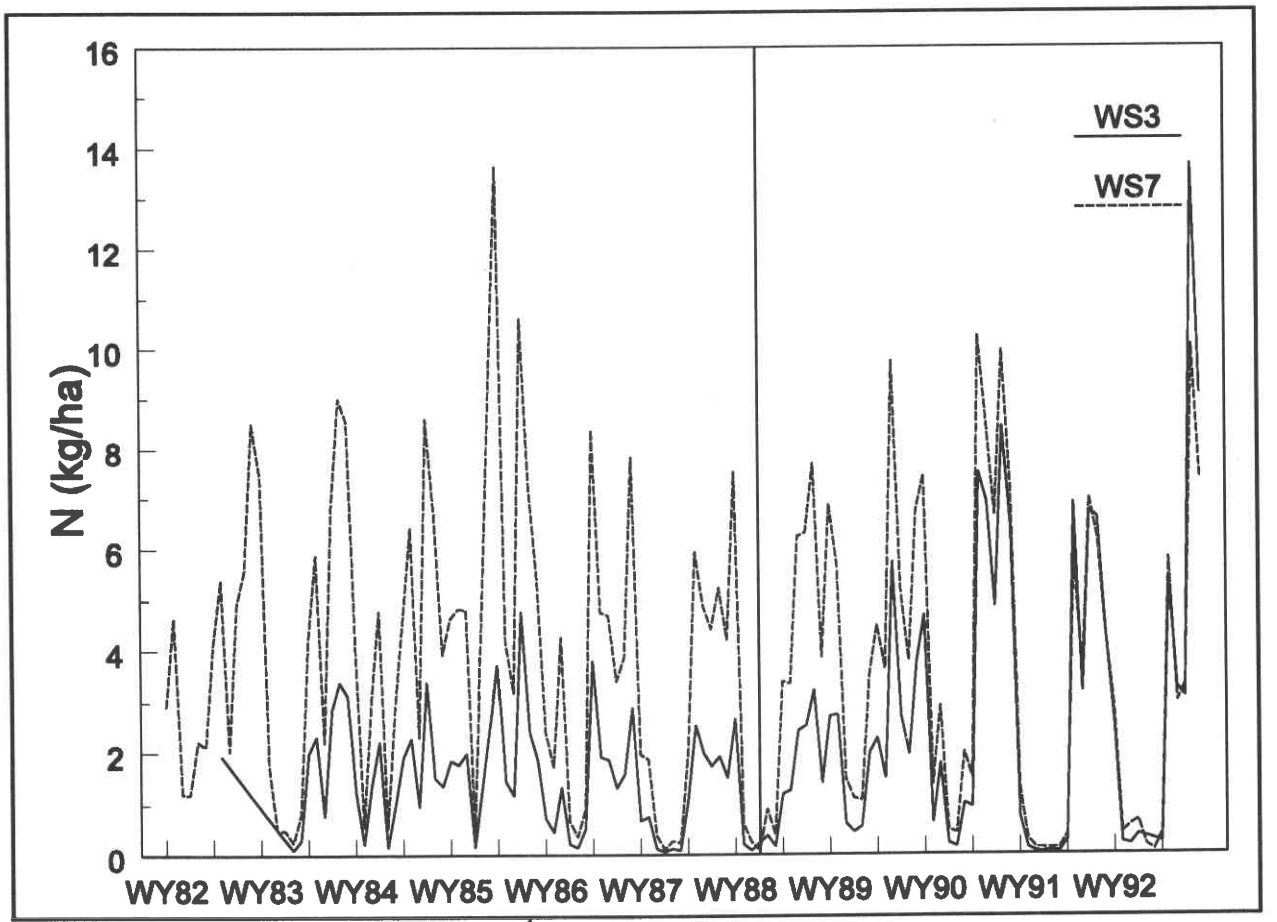

Figure 3. Mean monthly nitrogen export (kg ha^{-1}) from WS3 (treatment watershed) and WS7 (control watershed) on the Fernow Experimental Forest during water years 1982-1992. Vertical line indicates initiation of treatment.

Foliar Ca levels were lower on WS3 than WS7 (Figure 4), reflecting significant differences in foliar Ca concentrations for most species (Tables 3,4). Stemwood Ca did not differ between the two watersheds. Herb layer Ca concentrations for WS3 were twice as high as for WS7, a significant difference (Gilliam and others 1994). However, the difference is masked in the whole watershed budget because of differences in herb layer biomass described previously. Edwards and Kochenderfer (1993) reported ion pairing of Ca and NO_3 in soil and stream water export from WS3. This observation suggests increased availability of Ca on WS3, as Ca is removed from soil exchange sites during export of the mobile NO_3. Although soil Ca levels appear lower on WS3 than WS7, Gilliam and others (1994) found no statistically significant differences in Ca concentrations of soil between the two watersheds, due to high variability. Annual streamwater Ca export from WS3 and WS7 did not differ significantly (Figure 5).

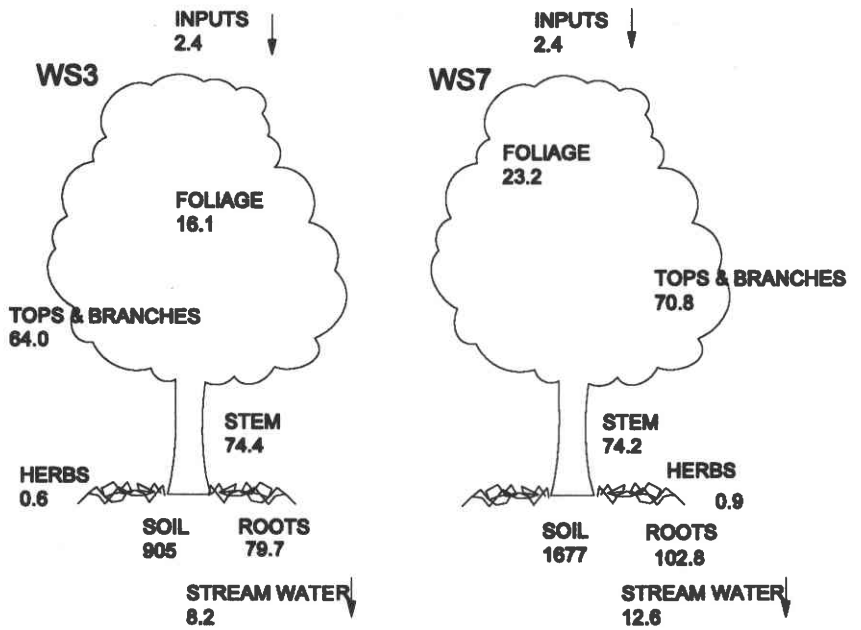

Figure 4. July calcium budgets for WS3 (treatment watershed) and WS7 (control watershed), Fernow Experimental Forest. Values are kg ha^{-1}.

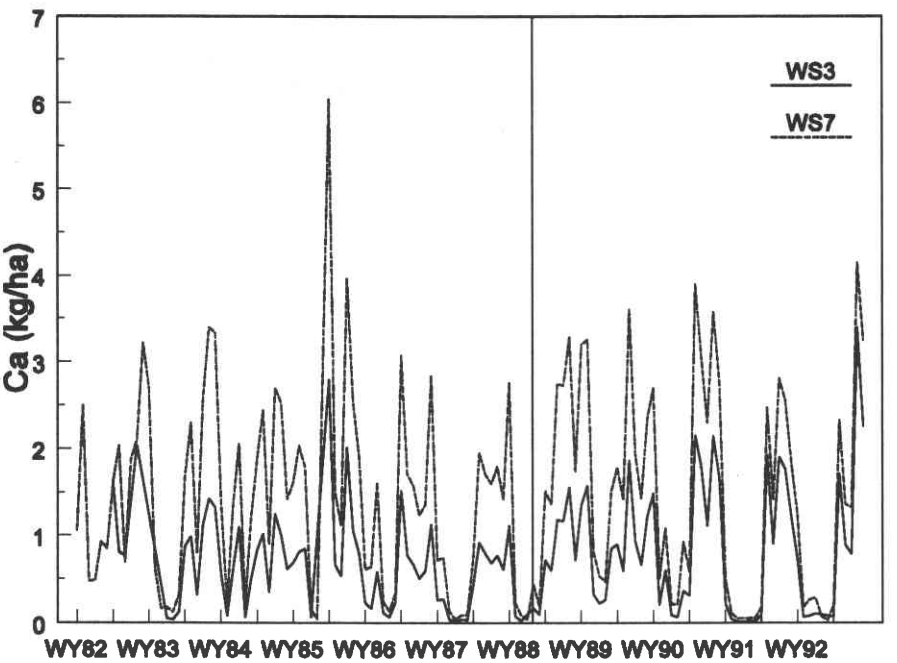

Figure 5. Mean monthly calcium export (kg ha^{-1}) from WS3 (treatment watershed) and WS7 (Control watershed) on the Fernow Experimental Forest during water years 1982-1992. Vertical line indicates initiation of treatment.

Nitrogen and Ca budgets for WS7 (reference watershed) were compared with those of Coweeta's Watershed 18 (Swank and Crossley 1988) and the Walker Branch Watershed in Tennessee (Johnson et al. 1988), both untreated reference watersheds (Table 2) to provide context for the observations from the acidification treatment. It was reasoned that if the three untreated controls showed similar nutrient levels, then the effects of the acidification experiment might be more obvious. The Fernow aboveground vegetation values appear low relative to the other two watersheds. This can be attributed mostly to differences in stand age -- stands on both the Coweeta Watershed 18 and Walker Branch Watershed were approximately 60 years old, while stands on WS3 and WS7 were 24 years old and had not accumulated biomass or nutrients for as long. Nutrient inputs in precipitation are similar among the three sites. However, it should be noted that the 6.2 kg N ha^{-1} reported here for the Fernow for 1991 is below the average value of 13.7 kg N ha^{-1} reported by Helvey and Kunkle (1986), who also reported average Ca inputs of 7.9 kg Ca ha^{1}. This is attributed to lower precipitation in 1991 (125 cm compared with the long-term average of 380 cm). Even during this year of relatively low deposition, N exports were much greater from WS7 than from the Coweeta Watershed 18 or Walker Branch Watershed. Younger, more vigorous stands are normally more conservative of N than older stands, thus this seems a contradiction. For this same year, the export from an 85 year old reference watershed on the Fernow for the same year was 3.45 kg N ha^{-1}. Therefore, N export from the Fernow watersheds does appear elevated, relative to Walker Branch and Coweeta.

Table 2. Comparison of N, P, and Ca budgets for three reference watersheds in the central and southern Appalachian Mountains. (NM=not measured).

	Fernow WS7	Coweeta[1]	Walker Branch[2]	
	kg ha^{-1}			
			YP[3]	OH[4]
Nitrogen				
Input	6.2	8.8	9.8	9.8
Aboveground	233.2	563.2	450	353
Soil	467[5]	6803	5730	2873
Output	6.9	0.09	0.056[6]	0.20[6]
Calcium				
Input	2.4	4.0	8.6	8.6
Aboveground	168.2	550.8	922	1283
Soil	2796	514	1445	668
Output	12.58	7.1	28.1[6]	4.81[6]

[1] Monk and Day (1988)
[2] Johnson and others (1988)
[3] Yellow-poplar stand, mean of 2 values
[4] Oak-hickory stand, mean of 2 values
[5] Extractable soil N, Coweeta and Walker Branch are total soil N
[6] Calculated from soil flux values of Johnson and others (1985)

Nutrient concentrations by species

Nutrient concentrations varied among the four species sampled, and among tree components. Foliar nutrient concentrations appeared the most responsive to treatment (Tables 3, 4). Foliar N concentrations were higher on WS3 for all four species studied, but were only significantly higher ($P \leq 0.10$) for black cherry and red maple. No significant differences in foliar mass were observed for these two species between the two watersheds. Foliar Ca was higher for all species on WS7 and was significantly different ($P \leq 0.10$) between the two watersheds for black birch, yellow-poplar and red maple. Some differences in foliar micronutrient concentrations also were detected, most notably for boron (data not shown). Overall, yellow-poplar showed the most differences, with significant differences in macro- or micronutrient concentrations for all components except small branches, suggesting that this fast-growing species may be more responsive to changes in nutrient supply than the other species examined.

Table 3. N and Ca concentrations by species, watershed and tissue type, from trees from two experimental watersheds on the Fernow Experimental Forest, WV. N is expressed in % and Ca as mg kg^{-1}. Watershed 3 = treatment, watershed 7 = control.

Species	Watershed	Tissue Type	N	Ca
Black birch	3	Foliage	2.720	3985
		Lg.branches	.198	1994
		Sm.branches	.462	2634
		Top	.180	1123
		Stem	.104	1000
	7	Foliage	2.540	5580
		Lg.branches	.170	2480
		Sm.branches	.448	3809
		Top	.130	1118
		Stem	.154	1195
Yellow-poplar	3	Foliage	3.002	6996
		Lg.branches	.212	2405
		Sm.branches	.475	3725
		Top	.193	1579
		Stem	.125	1294
	7	Foliage	2.750	11457
		Lg.branches	.222	2717
		S.branches	.435	4221
		Top	.230	1973
		Stem	.134	1915
Black cherry	3	Foliage	2.990	4354
		Lg.branches	.192	930
		Sm.branches	.508	1723
		Top	.180	776
		Stem	.126	1066
	7	Foliage	2.682	5899
		Lg.branches	.185	1042
		Sm.branches	.510	2413
		Top	.185	840
		Stem	.120	1071
Red maple	3	Foliage	2.310	4536
		Lg.branches	.172	1588
		Sm.branches	.448	3710
		Top	.183	1202
		Stem	.107	1306
	7	Foliage	2.070	6102
		Lg.branches	.178	2151
		Sm.branches	.446	4244
		Tops	.153	1486
		Stem	.101	1555

Table 4. Probability of >/t/, WS3 compared to WS7 for tissue biomass and nutrient concentrations, by tree component and species. Statistically significant values (P ≤ .10) are underlined.

	Mass	N	Ca
Stem			
Black birch	.029	.390	.297
Yellow-poplar	.035	.750	.003
Black Cherry	.848	.780	.974
Red maple	.102	.800	.124
Top			
Black birch	.340	.049	.982
Yellow-poplar	.455	.038	.063
Black cherry	.162	.746	.520
Red maple	.878	.264	.067
Small branches			
Black birch	.099	.796	.027
Yellow-poplar	.184	.163	.301
Black cherry	.499	.962	.195
Red maple	.984	.970	.132
Large branches			
Black birch	.278	.280	.200
Yellow-poplar	.052	.732	.282
Black cherry	.395	.720	.578
Red maple	.237	.806	.089
Foliage			
Black birch	.165	.502	.040
Yellow-poplar	.090	.298	.082
Black cherry	.285	.030	.136
Red maple	.300	.011	.013

Differences in nutrient concentrations were detected in the overstory trees, but not in the herb layer (Gilliam and others 1994). We predicted treatment effects first in the herb layer, particularly in response to N. It may be that the herb layer was limited by factors other than nutrition (e.g., light) (Gilliam and others 1994), and was unable to take advantage of the elevated N inputs. Gilliam and others (1994) reported significant changes in Fe and Al concentrations in herbaceous vegetation which they attributed to increased mobility of these ions in the soil due to the acidification treatment. Differences between watersheds in Al concentrations were significant only for yellow poplar and black cherry stemwood, but no significant differences in Fe concentrations were detected. Thus, the short-term response of these and other ecosystem components is variable and our ability to detect effects of acidic inputs also varies.

CONCLUSIONS

Whether observed differences in plant nutrient content reflect pretreatment differences between the watersheds is unknown. However, the lack of a consistent pattern of N enrichment for any tree species or tree tissue component suggest that plant nutrient status has been only minimally affected by three years of treatment. Soil acidification was not observed, but changes in stream water export of N were measured. The increased export of N from the treatment watershed suggests changes in N cycling at the watershed level. Nitrogen export may be elevated on the control watershed as well, suggesting ambient deposition may be causing effects, though more subtle, to those observed on the treated watershed. This research is ongoing, and will continue to evaluate the changes in nutrient cycling and other processes that result from acidic inputs, and consider implications for long-term sustainability of these important ecosystems.

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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

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Abstract: Atmospheric deposition has been recognized as a significant environmental problem for several decades, but its impact on forest ecosystems in North America remains controversial. In an effort to further elucidate the impacts of atmospheric deposition to forested watersheds in the Mid-Appalachian region, several related watershed studies have been initiated by the U.S. Forest Service in Parsons, WV over the past decade. One of the experiments involved doubling the annual ambient rate of N and S deposition by aerial application of ammonium sulfate to Watershed 9 which had previously been converted from low grade hardwoods to Japanese larch. We report here on the impacts of this treatment on soil and foliar chemistry, and height and diameter growth of the Japanese larch. Foliar samples were collected from treated and control trees and analyzed for Al, Ca, K, Mg, Mn, N, P and S. Soil sample analysis included Al, C, Ca, K, Mg, Mn, N and S. Treated soil had significantly greater exchangeable Al and Mn and significantly lower exchangeable Ca and Mg ($\alpha \leq 0.05$) compared to control soil. Treated larch had significantly higher ($\alpha \leq 0.05$) foliar N, Mn and Al concentrations and significantly lower ($\alpha \leq 0.05$) P and Mg concentrations. Height and diameter growth of treated larch were significantly ($\alpha \leq 0.05$) less. The reasons for these growth differences are as yet unclear; however, preliminary analysis indicated significant correlations with selected soil chemical variables, some of which have changed in response to the treatment.

INTRODUCTION

The debate continues over the effects of anthropogenic nitrogen and sulfur deposition on forest health, resulting in a shift in research emphasis away from direct impacts and towards predisposing and/or inciting stresses. The forest health perspective has broadened on the issue of acid deposition to include effects on forest soils, tree nutrition, accelerated soil base cation leaching and metals toxicity (Tomlinson and Tomlinson 1990). Nutrient budgets for forests, such as those on Walker Branch Watershed, Tennessee, and Bear Brook Watershed, Maine (Johnson and others 1985; Kahl and others 1993) have been studied carefully and the impacts of atmospheric deposition on the cycling of nutrients in these forest ecosystems documented.

Ulrich and Matzner (1986) characterized soil acidification by base cation loss from the soil and acidic cation accumulation. Selective cation exchange leads to accelerated leaching of exchangeable cations (VanMiegroet and Cole 1984). Federer and others (1989) reported that calcium was being leached from some eastern forest soils. For forest soils already low in base cations, a further decrease in these nutrients associated with acidic deposition leaching may be critical to plant growth.

Documented changes in soil characteristics due to acidic inputs under field conditions are few (David and others 1990). Elevated acidic loadings have been shown to increase cation leaching (including Al^{3+}), decrease soil pH and adversely affect biological processes (Fernandez and Rustad 1990). The loss of cations is dictated by the amounts of

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anions in solution, and concentrations of strong acids, such as H_2SO_4 and HNO_3. Increased soil acidity also increases Al, Mn and Fe solubility (Johnson and Cole 1980). Soil solution electroneutrality is maintained, following base cation losses, by these potentially-phytotoxic elements. Thus, soil chemistry changes may influence vegetation adversely.

The objective of this study was to quantify soil and foliar chemical changes and the response of Japanese larch to inputs of strong acids as a part of a catchment-scale acidification project. The results reported in this paper relate the growth and nutritional status of larch trees that have received increased deposition of N and S to the changes in soil chemistry resulting from this treatment.

METHODS

Study Site

Watershed 9 is an 11.6-ha catchment located in the unglaciated Allegheny Plateau of north central West Virginia. Kochenderfer and Helvey (1989) provided a general description of the watershed and a history of past land use. Minimum and maximum elevations on the watershed are 744 and 878 m, respectively. The watershed has a southerly aspect, with a 25% average slope. The predominant soil is Calvin channery silt loam (loamy-skeletal, mixed, mesic Typic Dystrochrept) weathered from the Hampshire formation. Agriculture was the primary land use until the early 1930s, when the watershed was abandoned and allowed to naturally revegetate with low grade hardwoods. The U.S. Forest Service obtained the land in 1939. A mechanical site preparation study was conducted on the watershed beginning in 1983, in which all vegetation was removed except for an undisturbed 1.42-ha buffer zone along the stream. Site preparation included root raking and windrowing brush. In the spring of 1984, the watershed (excluding the buffer zone) was planted with 2-0 stock Japanese larch at 1.8 by 1.8 m spacing.

Experimental Treatments

Prior to treatment, six 30.5 by 30.5 m control plots were randomly located in the watershed and permanently marked. To simulate inputs of double the ambient rates of N and S deposition, granular ammonium sulfate fertilizer (21-0-0-24, proportions of N, P, K and S) was applied to watershed 9 by helicopter beginning in April 1987. The entire watershed including the buffer zone but excluding the control plots received three ammonium sulfate treatment applications per year through August 1993 (for a total of 18 applications prior to the August 1992 sampling and 20 applications prior to the August 1993 sampling) in an attempt to mimic seasonal deposition patterns. Determination of treatment amounts were based on historical throughfall deposition in the area, accounting for seasonal variation. Double the deposition rate for the January-April period was applied in March, double the rate for the May-August period was applied in July, and double the September-December rate was applied in November. The rates corresponded to 8.1 kg S ha^{-1} and 7.1 kg N ha^{-1} for applications in March and November, and 24.4 kg S ha^{-1} and 21.3 kg N ha^{-1} for the July applications (Adams and others 1993).

Study Design

Access to the control plots on the watershed was restricted by dense vegetation; consequently, only one control plot was selected for study. Other control plots could not have been studied without severe disturbance to the plots. A northeastern control plot was selected, and a 25 by 25-m plot was surveyed and marked within the 30.5 by 30.5-m control plot to allow a treated buffer zone. One adjacent treated area of the same dimensions was surveyed and marked along the same contour 10 m east of the control plot. Four transect lines along a north-south orientation were selected at 5-m intervals for both the control plot and the treated plot (Figure 1). Starting at the southern boundary line, 6 Japanese larch trees were selected and brass tagged at 5 m intervals in a northern direction along each transect line. Three additional larch trees were tagged between transect lines 1 and 2 and between lines 3 and 4. The total of 30 sample trees per plot represents approximately 45% of the total number of larch trees on each plot.

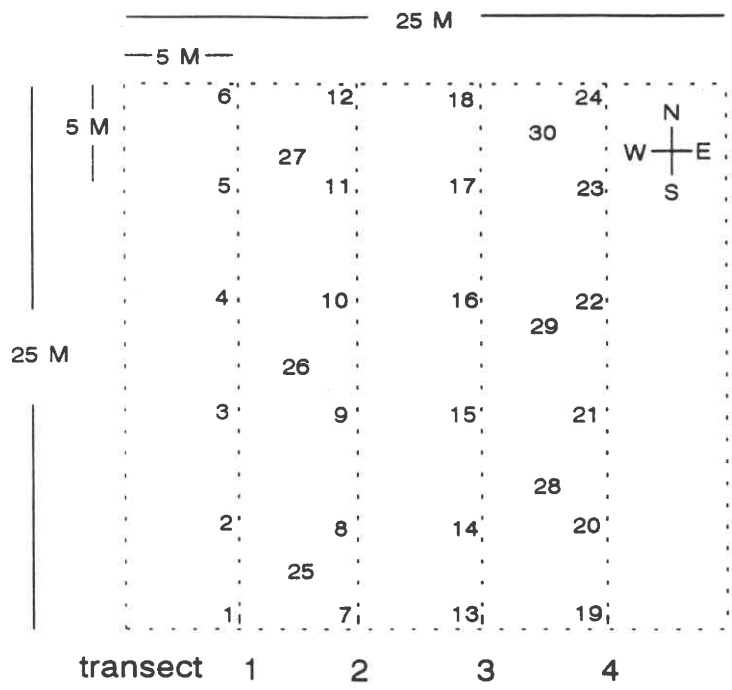

Figure 1. Transect and sampling design.

Sampling and Measurements

In August of 1992 and 1993, a telescoping pole was used to measure tree height to the nearest 0.1 cm and a caliper was used to measure tree diameter to the nearest 0.1 cm on all sample control and treated trees. A foliar sample for chemical analysis also was obtained from each of these trees. All needle whorls from a breast-high lateral shoot immediately below the terminal shoot were removed. Samples were handled with vinyl gloves, placed in paper bags, and kept cool until delivery for analysis (within 24 hours). Foliar analysis was performed by the Agricultural Analytical Services Laboratory, The Pennsylvania State University. Elemental analysis included Al, Ca, K, Mg, Mn, and P by inductively coupled plasma emission spectroscopy (Dahlquist and Knoll 1978), Kjehdahl N (Isaac and Johnson 1976) and sulfur (Huang and Schulte 1985).

In August 1992, 6 control and 6 treated soil pits (approximately 20 by 20 cm) were hand excavated near the sample larch trees along control transect 1 and treated transect 4. In August 1993, small pits were hand excavated by each sample larch tree on transect lines 1 and 4 for both the control and treated plots (12 pits per plot total). Soil samples were collected in the 0-2 cm layer (O horizon) and in the 2-10 cm layer (A horizon). These depths were chosen to represent the effective rooting horizons (Kochenderfer 1973). Samples were placed in paper bags and transported to the Environmental Resources Research Soil Chemistry Laboratory at The Pennsylvania State University. All analysis was conducted on air-dried soil sieved to 2 mm. Soil pH (1993 only) was determined in water with soil-to-solution ratios of 1:1 for both horizons (Black 1964). The base cations Ca, K, and Mg (K 1993 only) and metal cations Al and Mn (Mn 1993 only) were determined by atomic absorption spectrophotometry following 0.01 M $SrCl_2$ extraction and preparation as outlined by Joslin and Wolfe (1989). Soil carbon (Schulte 1991), NO_3-N (Isaac and Johnson 1976) and SO_4-S (Black 1964) 1992 analysis were performed by Agricultural Analytical Laboratory. In 1993, total C and N were analyzed by combustion using an organic elemental analyzer and for total S by combustion using the organic elemental analyzer with an electron capture detector (Baccanti and others 1993; Carlo Erba Instruments 1990) at the Timber and Watershed Laboratory, Parsons, WV.

Data Analysis

Statistical analysis followed a Model I two-treatment ANOVA for a fixed treatment. The SAS statistical package was used for data analysis, incorporating t-tests for mean comparisons (SAS Institute 1985). All significant differences are reported at $\alpha \leq 0.05$. Correlations using soil and tissue variables were performed using means for a particular treatment, soil horizon and chemical element. Quality assurance/control for all analysis included field duplicates, analytical duplicates, standard reference materials and blanks. Generally, duplicate analyses had relative standard deviation values below 10 percent.

RESULTS AND DISCUSSION

Larch Height and Diameter

Larch height and diameter growth were significantly greater for the control trees compared with treated trees for both 1992 and 1993 (Figure 2). Positive growth on both control and treated plots was evident, in that height and diameter measurements were greater in 1993 than 1992. However, control larch exhibited significantly greater height and diameter growth than treated larch in both years. Data for larch height growth on the entire watershed also indicate significantly greater height growth on the control plots compared to larch receiving the acidifying treatment (personal commun., J. Kochenderfer, U.S. Forest Service).

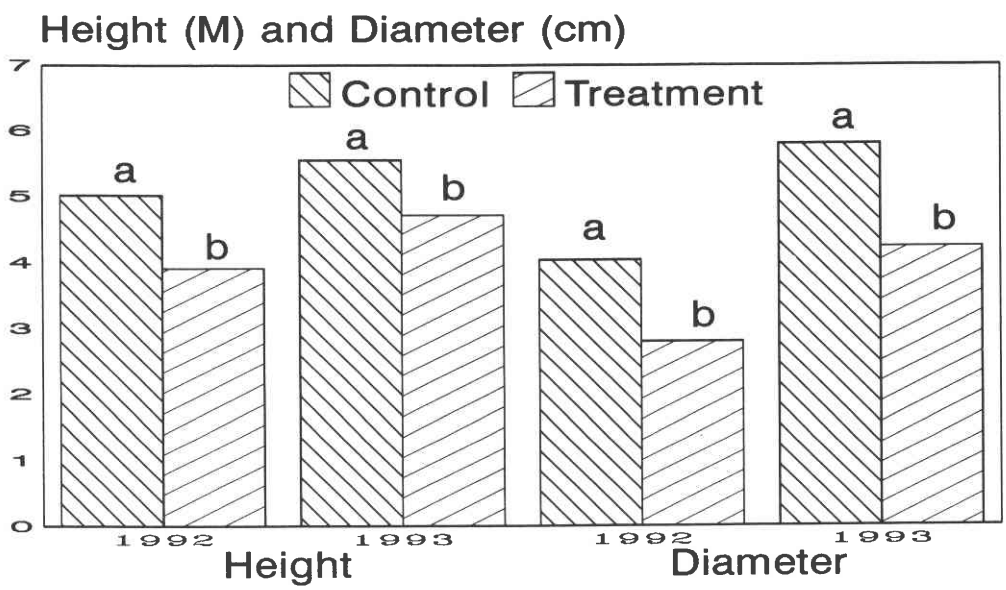

Figure 2. Larch height and diameter comparisons between control and treated plots; ($\alpha \leq 0.05$).

Soil Chemistry: Treatment Response (Within Horizon Comparisons)

Control and treated soil data comparisons were made for 1992 and 1993 by sample depth (O horizon or A horizon). In 1992, treated soil from the O and A horizon had greater values of NO_3-N and C compared to soil from controls. The A horizon differences were statistically significant. Treated soil from the A horizon had significantly greater SO_4-S than the control (Table 1).

Table 1. Soil SO_4-S, NO_3-N and C comparisons between control and treated samples collected in August 1992.

	O HORIZON			A HORIZON		
	SO_4-S (mg/kg)	NO_3-N (%)	C (%)	SO_4-S (mg/kg)	NO_3-N (%)	C (%)
Control	148.83a	0.965a	16.9a	29.50a	0.113a	1.9a
Treatment	131.33a	1.108a	18.6a	121.60b	0.155b	4.1b

different letters within a soil parameter indicate significant difference at $\propto \leq 0.05$ n=6

According to Stevenson (1986), net mineralization leading to an increase in mineral N levels occurs at C:N ratios below 20. For the 1992 soil data, the O-horizon C:N ratio was 17:1 and the A-horizon C:N ratio was 15:1, in both the control and the treated soil. Control O- and A-horizon total C and N were positively correlated with one another (O horizon: $r=+0.89$, $p < 0.05$, A horizon: $r=+0.97$, $p < 0.01$), as was treatment O-horizon C and N ($r=+0.96$, $p < 0.05$). Mineralization of N under these C:N ratios may be enhanced with N addition. Elevated humus layer N and increased organic matter in the Germany's Solling forest has been attributed to litter decomposition retardation, due to declining microbial activity under increasing acidic inputs (Robarge and Johnson 1992). The potential for N saturation is indicated by the low C:N ratios (Kahl and others 1993). The possibility of N saturation and the increase in N and S in treated soils provided the potential for SO_4^{2-} and NO_3^--induced leaching and increased soil acidification. In 1993, soil pH was significantly lower in the treated soil (O-horizon pH 4.47, A-horizon pH 4.91) compared to control (O-horizon pH 4.98, A-horizon pH 5.38), indicating increased soil acidification in the treated soil.

Exchangeable Al values in the treated soils were significantly greater and exchangeable Ca and Mg values significantly lower compared to control, for both years and at both depths (Table 2). In 1993, Mn was found to be significantly greater in the treated soil at both depths. Robarge and Johnson (1992) described increased solubility and mobility of Al as a potential effect of increased N and S inputs to forest soils. Soil acidification also can induce the loss of exchangeable cations such as Ca and Mg and increase exchangeable Al and other metals such as Mn (Schlegel and others 1992). Because the growth of larch is not stimulated by N and the Ca demand of larch foliage is relatively high, N-induced acidification and consequent Ca leaching may be contributing to the observed growth reduction of the treatment trees (Tyrrell and Boerner 1987; VanGoor 1953).

Table 2. Chemical comparisons (0.01 M $SrCl_2$ exchangeable) between control and treated soil sampled in August of 1992 and 1993.

	O HORIZON						A HORIZON					
1992	Al	Ca	K	Mg	Mn	pH	Al	Ca	K	Mg	Mn	pH
	--------	--------	-mg/L	--------	--------		--------	--------	-mg/L	--------	--------	
Control	2.63a	175.5a	NA	22.48a	NA	NA	2.02a	170.7a	NA	12.29a	NA	NA
Treatment	6.21b	104.0b	NA	13.04b	NA	NA	17.23b	30.8b	NA	3.96b	NA	NA
1993												
Control	4.11a	160.8a	13.45a	18.08a	9.07a	4.98a	1.54a	123.5a	5.89a	8.33a	0.87a	5.38a
Treatment	12.95b	96.5b	21.63b	10.18b	21.41b	4.47b	10.66b	65.3b	5.14a	3.64b	1.76b	4.91b

different letters within a soil parameter indicate significant difference at $\propto < 0.05$
1992 n = 6 1993 n = 12 NA = not analyzed

Soil Chemistry: Within Horizon Observations

Control and treated soil N was significantly greater in the O horizon than the A horizon in 1992 and in 1993. Sulfur showed a similar, non-significant trend in 1992. Analysis of S for 1993 is ongoing. For the control soil sampled in 1992 and 1993, Mg was significantly greater in the O horizon than the A horizon. In 1993, O-horizon Al, K, Mn and N were all significantly greater than A-horizon levels, while pH was significantly lower in the O horizon. These differences may be due to greater leaching losses in the A horizon or to greater nutrient inputs (needle-fall) in the O horizon or both.

For the treated plots in 1992 and in 1993, O-horizon Mg was also significantly greater than A-horizon Mg. This may be a reflection of initial, pretreatment soil status, as the control O horizon also had significantly greater Mg than the A horizon. Therefore, although the control and treated soil comparisons showed that the treated soil had lost more Mg in both horizons compared to the control, the losses may be more severe in the A horizon as a result of its inherently lower Mg status compared to the O horizon.

There were no significant differences between control and treated soils for Mg in the O horizon between 1992 values and 1993 values, although both treatments had a trend of decreasing Mg values. However, for the control A horizon, the Mg level in 1993 was significantly less than that for 1992, and a similar trend (non-significant) was observed in the treated plots.

In 1992, treated plot Al and Ca levels were significantly greater in the O compared to the A horizon, but these differences were not significant in 1993. In 1993 the O horizon had significantly greater K and Mn and significantly lower pH compared to the A horizon.

Overall, there are more nutrients found in the O horizon compared to the A horizon on both the control and the treated plots. Carbon and N were also greater in the O horizon. These differences may be due to greater leaching losses in the mineral horizon or to greater nutrient inputs (needle-fall) in the O horizon. Larch allocate a small proportion of total annual C gain below ground (Gower and Richards 1990). Fine roots growing in the nutrient-rich O horizon are also exposed to elevated Al levels in the treated soil. Joslin and Wolfe (1989) showed root branching sensitivity in northern red oak (*Quercus rubra* L.) to Al in acidification treatments. Larch, which has been planted extensively on harsh sites, would not be expected to be very sensitive to Al toxicity; however, it is possible that Al toxicity may have decreased nutrient uptake and contributed to decreased growth.

Foliar Chemistry: Treatment Response

Foliar samples from the larch on the treated plot collected in 1992 and 1993 had significantly greater Mn and Al levels and significantly lower P and Mg levels compared to the control (Figure 3). Foliar K values were not significantly different between control and treatment larch. The treated larch had significantly greater foliar N (23000 mg/kg) compared to the control (21300 mg/kg) in 1993. Also, the 1993 values of treated larch for Mn, N, P and S were significantly greater than its 1992 values (1993 values: Mn=3443.8 mg/kg, N=23030 mg/kg, P=1330 mg/kg , S=2120 mg/kg; 1992 values: Mn=2427.2 mg/kg, N=20960 mg/kg, P=1040 mg/kg, S=1860 mg/kg). Lower foliar P in treated larch may be due to the formation of an Al-PO_4 precipitate, possibly contributing to lower P availability (McCormick and Borden 1974). VanGoor (1953) reported the existence of a N/P antagonism in Japanese larch. The lower the availability of soil P the more quickly this antagonism develops with added N. VanGoor (1953) reported that the optimum range of N/P in needles was 4 to 5. The ratios in this study were 17 and 15 for the treated and control larch, respectively. Both soil and foliar Mn were elevated in the treated larch compared to the control. Studies with agronomic crops have shown that Mn can become elevated in acidic soils and reach toxic levels, impairing Ca and Mg uptake, disturbing physiological processes and reducing growth (Elamin and Wilcox 1986; Smith and others 1983). A greenhouse study designed to evaluate larch growth under elevated soil Mn found no height or diameter depressions in soils with Mn levels ranging from 12.8 to 150 mg/L and foliar Mn from 1655 to 5550 mg/kg (Unpublished data, C. Pickens, The Pennsylvania State University).

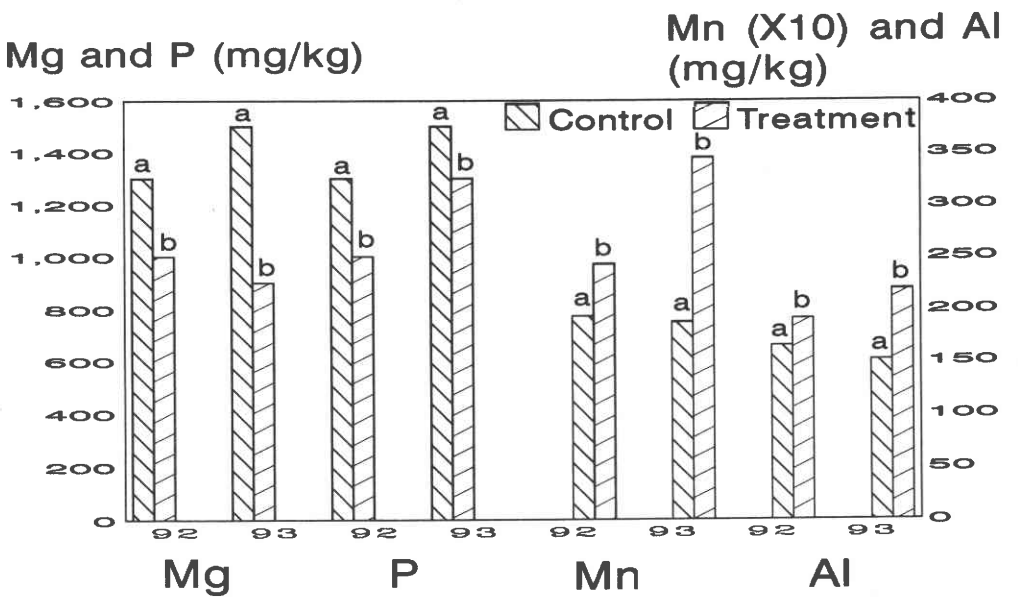

Figure 3. Foliar chemical comparisons between control and treated larch.

The patterns of foliar Al, Mg, Mn, N and S in the treated larch foliage compared to the control were somewhat similar to the patterns observed for soils. A general trend was that the treated plot had lower foliar and soil Mg levels, while Al, Mn, N and S were greater in treated soils and foliage compared to control values. The link between increased deposition of strong acids and an increase in soil Al has been made (Ulrich and others 1980), as has the antagonistic relationship between Al and the uptake and transport of Ca and Mg (Joslin and others 1988). Prior farming on watershed 9 included the addition of burned lime (personal comm., J. Kochenderfer, U.S. Forest Service) One possible explanation for the lack of Ca response (leaching) to acidic inputs may be due to these Ca additions, which still provide adequate Ca. However, Mg levels appear to be affected by the acidification treatment, with Mg levels reduced in the treated soil and foliage, possibly as a consequence of Mg leaching losses with mobile SO_4^{2-} and NO_3^- ions. Magnesium content of sandstone derived soils is low relative to Ca and, thus, it might be expected that the available pool of Mg might be affected before that of Ca. Magnesium deficiencies have been reported in declining Norway spruce (*Picea abies* L.) in Europe (Tomlinson and Tomlinson 1990) and in the eastern US (Ke and Skelly 1994), and Tomlinson and Tomlinson (1990) attributed these deficiencies to excessive leaching as a consequence of acidic deposition. Bergmann (1992) gives an adequate Mg range for European larch (*Larix decidua* Mill.) of 1200-3000 mg/kg. The treated plot larch had less than 1200 mg/kg of Mg in both 1992 and 1993 suggesting a possible deficiency.

Foliar and Soil Chemistry Correlations

Foliar Mg consistently correlated with a number of soil chemistry parameters. In the treated samples, foliar Mg was positively correlated with A-horizon Ca ($r = +0.86$, $p < 0.05$) and Mg ($r = +0.82$, $p < 0.05$) (1992) and O-horizon Ca ($r = +0.58$, $p < 0.05$) (1993). In 1992, soil values for treated A-horizon Ca and Mg were correlated positively with larch height (Ca: $r = +0.83$, $p < 0.05$; Mg: $r = +0.84$, $p < 0.05$) and diameter (Ca: $r = +0.91$, $p = 0.01$; Mg: $r = +0.83$, $p < 0.05$). Negative correlations were observed between O-horizon N and larch height ($r = -0.81$, $p < 0.05$) and diameter ($r = -0.84$, $p < 0.05$), and A-horizon Al and diameter ($r = -0.81$, $p < 0.05$). In 1993, A-horizon N again was correlated negatively with diameter measurements ($r = -0.81$, $p < 0.05$). In 1993, foliar Mg in treated larch were correlated positively to both height ($r = +0.60$, $p < 0.05$) and diameter ($r = +0.67$, $p < 0.05$). Negative correlations existed between foliar Mg and O-horizon C in 1992 ($r = -0.97$, $p < 0.01$) and 1993 ($r = -0.63$, $p < 0.05$), and between foliar Mg

and O-horizon N ($r = -0.97$, $p < 0.01$) and A-horizon S ($r = -0.81$, $p < 0.05$) in 1992, and A-horizon Al ($r = -0.70$, $p < 0.01$) in 1993.

Based on correlations with foliar and soil chemistry, Mg appears to be the most important variable potentially influencing larch growth. The negative correlations between larch growth parameters and soil N support the hypothesis that larch has a low N requirement and responds adversely to increased N (Leyton 1957). Treated soil Ca and Mg were correlated positively with growth; deficiencies of these nutrients under acidification stress may be detrimental to larch growth. Heenan and Campbell (1981) demonstrated that increasing Mn supply reduced Mg uptake in soybean, and that Mn reduced Mg leaf concentrations by reducing the rate of Mg absorption. It also has been reported that a liberal supply of Ca can counteract Mn toxicity (Ouellette and Dessureaux 1985). Plant available Mn was elevated in the treated soil, and such an increase along with a decrease in exchangeable Mg may have interfered with larch growth. The negative correlations between larch growth and soil Al also may implicate Al in the observed growth reduction.

SUMMARY AND CONCLUSIONS

After 18 (1992) -20 (1993) treatments (total applications) with ammonium sulfate fertilizer over 5 yrs/8 mos (1992) and 6 yrs/8 mos (1993), the larch trees growing on watershed 9 have reduced height and diameter growth compared to control trees. Soil chemistry revealed greater levels of Al, C, Mn, N and S and lower levels of Ca, Mg and pH compared to control soils. Significantly greater foliar N in the treated larch indicates greater N availability on the treated areas. Earlier studies have shown high N/P ratios to be detrimental to larch growth, but the N/P ratios for foliage in control and treated larch were high and well above the reported optimums. Treated larch height and diameter were correlated positively with foliar Mg. Foliar Mg was reduced significantly on the treated plot. Reduced Mg availability also may have contributed to reduced larch growth, since foliar Mg concentrations of treated larch were less than adequate. It appears most likely that a change in Mg supply coupled with increases in available Al and Mn have contributed to nutrient deficiencies and imbalances and impaired growth of treated larch.

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TREE SURVIVORSHIP IN AN OAK-HICKORY FOREST IN SOUTHEAST MISSOURI, USA UNDER A LONG-TERM REGIME OF ANNUAL AND PERIODIC CONTROLLED BURNING

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Abstract: Fire significantly altered survivorship in southeastern Missouri forests burned at annual and four-year (periodic) intervals since 1949 and 1951. In the ANOVA model tested, treatment, species, pretreatment diameter and the interaction between species and pretreatment diameter were highly significantly related to survivorship. Overall survivorship and survivorship of hickories (*Carya* spp.) and red oak group species, including scarlet oak (*Quercus coccinea* Muenchh.), black oak (*Quercus velutina* Lam.), southern red oak (*Quercus falcata* Michx.), was greatest in control plots, less in annually burned and least in periodically burned plots. In contrast, the survivorship of post oak (*Quercus stellata*, Wangenh.) did not significantly differ among fire regimes. General fire survivorship rankings were post oak > red oak species > hickory. Logistic regression indicated that survivorship was significantly and positively correlated with diameter at breast height for post oak and red oak group species, but not for hickories. This relationship of DBH with survivorship was apparently attributable more to self-thinning and differential life-span of trees rather than fire treatment effects. However, survivorship of red oak species was more sensitive to burning regime at smaller pretreatment DBH. When using fire to maintain or increase the abundance of red oak species of oak-hickory forests, age, species life-span, stage of stand development and other environmental stresses should be considered.

INTRODUCTION

The role of fire in oak-hickory forests in the Eastern Deciduous Forest of North America varies across regions. These regional differences are due, in part, to differences in fire behavior which are strongly influenced by topography and fuel type (Pyne 1984). In addition, variation in climate, species composition of the forest itself, soil type and composition of the surrounding vegetation also contribute to regional differences in fire response of oak-hickory forests (Pyne 1984).

In the southeastern U.S., frequent low intensity fires favor the dominance of southern pines, such as shortleaf (*Pinus echinata* Mill.), longleaf (*Pinus palustris* Mill.) and slash pine (*Pinus elliottii* Engelm.) over species of oaks (Garren 1943). In this region oak-hickory forests exist in areas protected from fire, such as in ravines (Garren 1943). Oaks are better adapted to fire than associated hickory species (Harlow and others 1991). As a result of these differential responses and the frequency of fire in this region, oak species can co-exist with pine better than hickories, forming oak-pine forests over much of the southeastern United States (Braun 1950).

Fire helps to maintain the composition of oak-hickory forests through selective mortality, or by creating conditions which alter growth rates of surviving trees (Curtis 1959, Bazzaz 1979, Crow 1988). However, fire also may kill and damage oaks (Stickel 1935). Fire may have both short- and long-term effects on tree mortality. The immediate danger fire poses is high temperature. Cells are killed between 50 and 64°C depending on cellular water content and species (Byram 1958, Hare 1961, Levitt 1980, Rouse 1986). Longer term risks posed by fire result from cambial injury and associated increased susceptibility of the stem to insect attack and diseases (Hedgecock 1926, Johnson 1974, McCarthy and Sims 1935, Stickel 1935, Stickel and Marco 1936).

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Fuel loading and moisture content determine both the maximum temperatures reached and duration of elevated temperatures during a fire (Byram 1958, Pyne 1984, Smith 1986, Vines 1968). In controlled burns at University Forest, southeastern Missouri, periodic burns, conducted every four years since about 1950, were more intense and caused greater mortality by 1964 than annual burns because of the accumulation of a deeper and more continuous litter layer in plots burned periodically (Paulsell 1957, Scowcroft 1966).

Many plant characteristics influence survival and growth after fire. Differential development of these characteristics explains much of the variation among species in post-fire survival and growth (Hare 1961, Kimmins 1987). Thick bark can insulate vital tissues, such as the cambium, which would otherwise be killed by high temperatures (Hare 1961, Vines 1968). Sufficient starch reserves and an ability to resprout allow some trees to survive despite experiencing topkill after fire (Johnson 1974, Kozlowski and others 1991, Little 1974, Malanson and Trabaud 1988). Thus, larger diameter trees may survive fire better than small diameter trees because they have thicker bark and greater starch reserves and because there is a lower probability that complete girdling will occur (Greene and Shilling 1987, Maslen 1988, McCarthy and Sims 1935, Paulsell 1957, Stickel 1935).

Previous field studies and observations have shown that species can be ranked according to fire tolerance. Hickory species are noted for their sensitivity to fire due to their thin bark. Fire sensitivity of oak species varies according to bark thickness, susceptibility to decay and ability to sprout. *Quercus coccinea* and *Q. falcata* both have thin bark and low resistance to decay after being wounded, making them quite sensitive to fire (Belanger 1990, Johnson 1990, Harlow and others 1991). Although more fire resistant because they have thicker barker, fire scars create entry points for decay organisms in both *Q. alba* and in *Q. velutina* (Sander 1990, Rogers 1990). Other oak species, such as *Q. stellata* and *Q. marilandica* survive fire better because they possess thick bark and can sprout after top-kill to form scrub thickets on sites subject to frequent fires (Harlow and others 1991, Stransky 1990, Vines 1968).

White oak group species (Subgenus *Leucobalanus*) generally live longer than red oak group species (Subgenus *Erythrobalanus*, Harlow and others 1991). Hence in young or near mature forests, white oak group species should have inherently lower mortality rates than red oak group species and, other factors held constant, should thus have greater survivorship. These differential fire and life history traits suggest that species differences in survivorship should be observed in the oak-hickory forests of southeastern Missouri.

Specific hypotheses addressed in this paper focused on factors affecting fire survivorship in these forests that have been subjected to three different long-term burning regimes: (1) annual and periodic burning decrease survivorship, (2) periodic burns reduce survivorship more than annual burns, (3) survivorship increases with DBH, and (4) fire survivorship differs among species.

METHODS

Experimental Design

In 1949 and 1951 two replicates of burning treatments were established about one mile apart from each other at the University State Forest in Wapapello, Missouri (36° 55' N, 90° 15' W). Replicate 1 had a site index of 65 on a Typic Fragiudalf while Replicate 2 had a site index of 56 on a Typic Fragiudult (Godsey 1988). Prior to establishment of the two study sites, fire had been excluded since 1930 and there had been no grazing permitted for 15 years. Each replicate contained two unburned control plots, two annually burned plots and two periodically burned plots for a total of six plots randomly arranged in a two by three grid. Each plot was 40 m by 40 m with 10.1 m buffer strips between plots. The burning schedule was staggered between replicates. Treatment of burn plots began in 1949 in replicate 1 and in 1951 in Replicate 2. Prescribed fire was conducted every four years after treatments began in periodically burned plots. Burning was conducted in the spring between March to May. The dates of the spring burns were chosen so that conditions in the late afternoon, when the burns were conducted, fell within moderate to high fire danger conditions (Paulsell 1957). The dates of prescribed burning always coincided with the occurrence of local fires (Paulsell 1957).

Stand Characteristics

Experimental plots had initial basal areas prior to treatment ranging from 12.49 to 15.25 m^2/ha in Replicate 1 and from 15.56 to 17.34 m^2/ha in Replicate 2. By 1984 total basal area, including ingrowth, had increased for all plots. Species of oak and hickory were predominant in tree-sized individuals ($\geq$ 4.06 cm DBH). Hickories reported at the sites include shagbark hickory (*Carya ovata* (Mill.) K. Koch), mockernut hickory (*Carya tomentosa* (Poir.) Nutt.) and black hickory (*Carya texana* Buckl.). Hickory basal area ranged from 1.82 to 6.07 m^2/ha. Measurements of all species of hickory were pooled in this study. White oak group species were *Quercus alba*, ranging in basal area from 0.00 to 1.18 m^2/ha before treatments, and *Quercus stellata* ranging in basal area from 2.79 to 10.00 m^2/ha in plots before treatment. Red oak group species were *Quercus falcata*, *Q. coccinea*, *Q. marilandica* and *Q. velutina* ranging in basal area before treatment from 3.59 to 8.15 m^2/ha. Other trees on the plots were flowering dogwood (*Cornus florida* L.), common persimmon (*Diospyros virginiana* L.), ash (*Fraxinus* spp.), plum (*Prunus* spp.), sassafras (*Sassafras albidum* (Nutt.) Nees) and winged elm (*Ulmus alata* Michx.). Data from these last species were eliminated before calculations (see below).

In the original inventory, a j-shaped DBH distribution was observed in all plots, as was shown when plots were grouped by treatment for each replicate (Figure 1). Replicate 1 plots had a greater density of red oak group species, but fewer *Q. alba* or *Q. stellata* trees than Replicate 2. In addition, Replicate 1 plots had fewer hickories than replicate 2.

Statistical Analysis

Only individuals of the 1949 (1951) cohort were included in the analysis. In contrast with Godsey (1988) who analyzed plot frequencies, mean percent survivorship was calculated by plot, in 4 cm pretreatment diameter classes for each species. Separation by size was performed to increase the sensitivity of the statistical analyses (see below). Trees in diameter classes larger than 20 cm DBH were removed before analysis in order to ensure that there were observations for every species and pretreatment diameter class combination in all plots. No hickories larger than 20 cm were observed on the plots before treatments were started. In addition, observations of large oaks were not consistent among large diameter classes and were generally a small proportion of the cohort (Figure 1). Because the number of white oak trees and trees of "other species" were low, plot survivorship means of these species categories also were removed from the analysis.

Mean survivorship of each plot for each species group and diameter class combination was then arcsin transformed before analysis of variance to normalize error as described by Box and Cox (1964). Least square means of transformed 1984 survivorship were calculated by employing a nested ANOVA model with pretreatment diameter as a covariate and using the SAS General Linear Models procedure (Table 1). Least square means were then back-transformed into percentages.

In a second model, logistic regression (Harrell 1986) was used to develop functions describing the probability of an individual of a species or species group (hickory, post oak or red oak) surviving from 1949 (or 1951) until 1984 using the binary data of individual survivorship with DBH treated as a continuous independent variable. Pretreatment DBH was included as both linear and squared terms in logistic regression because a curvilinear relationship between survivorship and DBH^2 has been observed for other species of oak (McCarthy and Sims 1935). The Wald-Chi square values calculated for each logistic curve were used to determine the significance of DBH and DBH^2 coefficients for all combinations of species groups and treatments for which both sufficient numbers of observations were present and convergence was achieved in the logistic regression. Significant differences are reported at $p \leq 0.05$ unless otherwise noted.

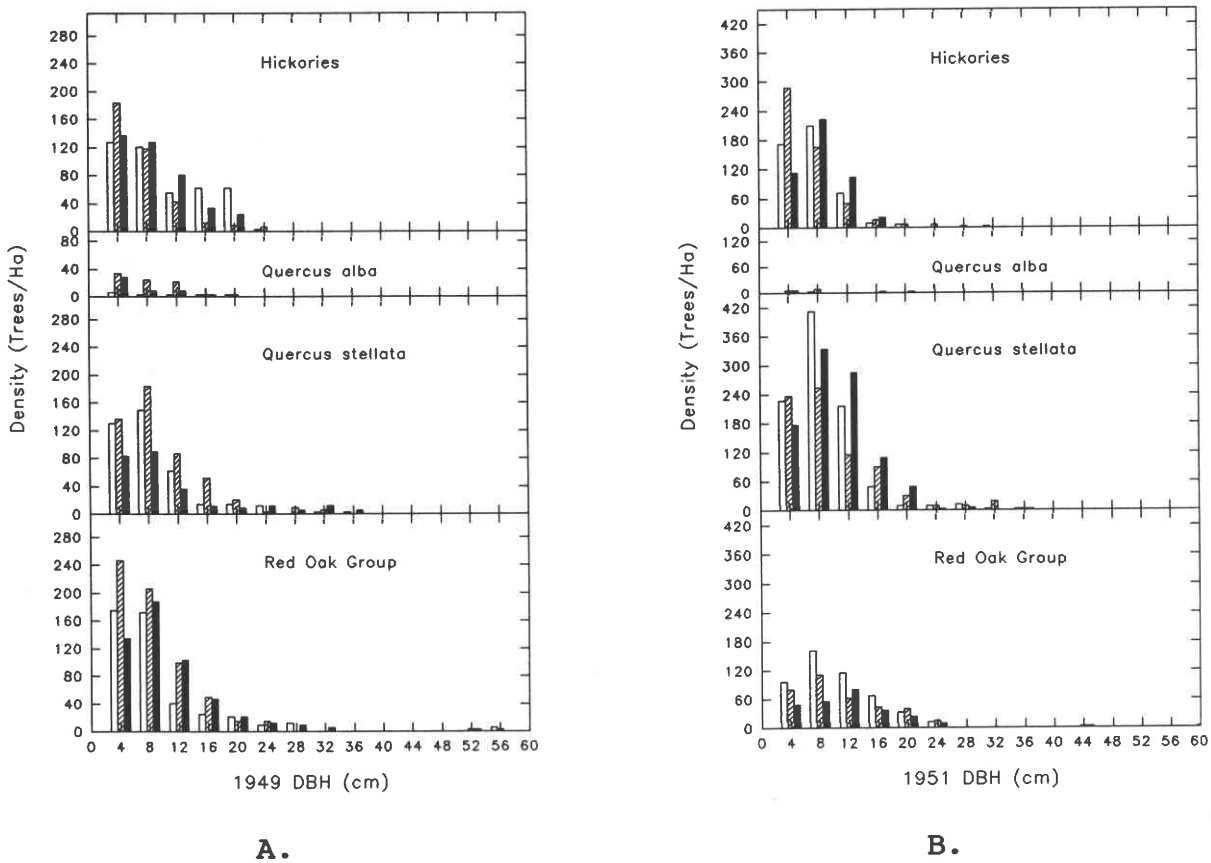

Figure 1: A - Density-diameter relationship in 1949 in Replicate 1 of species of hickory, white oak and red oak. B - Density-diameter relationship in 1951 in Replicate 2 of species of hickory, white oak and red oak. Open bars - control plots; cross hatched bars - annually burned plots; solid bars - in periodically burned plots. In Replicate 1, total densities were 1544 trees/ha in control plots, 1601 trees/ha in annually burned plots, and 1147 trees/ha in periodically burned plots. Density-diameter relationship Open bars - control plots; cross hatched bars - annually burned plots; solid bars - in periodically burned plots. In Replicate 2, total densities were 1872 trees/ha in control plots, 1631 trees/ha in annually burned plots, and 1616 trees/ha in periodically burned plots.

Table 1: Effects and their associated p-values from the ANOVA model used to calculate least square means of transformed 1984 plot survivorship.

Effect	Degrees of freedom	Associated p-value
DBH	4	<0.01
Species	2	<0.01
Species*DBH	8	<0.01
Treatment	2	<0.01
Treatment*DBH	8	0.43
Species*Treatment	4	0.22
Species*Treatment*DBH	16	0.63
Error	44	(Replicate*Species*Treatment*DBH)

RESULTS

Statistical and Logistic Regression Models

In the complete ANOVA model, effects of pretreatment diameter, species, treatment, and the interaction between species and pretreatment diameter all were highly significant ($p \leq 0.001$, Table 1). Neither the interactions between treatment and pretreatment diameter, nor the interaction between species, treatment and pretreatment diameter were significant. Although the F-test of the interaction between species and treatment was not significant ($p = 0.22$, Table 1) statistically significant differences existed between least square means (Table 2).

Except for the DBH^2 term for post oak on control plots, all terms of the logistic regression models (intercept, DBH and DBH^2) were highly significant ($p \leq 0.01$) for post oak and red oak group trees in all treatments. In contrast, only the intercept term was significant in all logistic regression models for hickory species. In periodically burned plots DBH and DBH^2 approached statistical significance in relation to survivorship of species of hickory ($p = 0.071$ and 0.084, respectively).

Species Differences in Survivorship

Both the effects of species and the interaction between species and pretreatment diameter were highly significant in the ANOVA model tested (Table 1). Differences among species in percent survivorship in all treatments were significant. Post oak showed the highest mean survivorship (55.6%), red oak species showed the next highest mean survivorship (26.1%) and the hickories showed the lowest mean survivorship (4.8%). The effect of burning regime was highly significant with mean survivorship in control plots (37.9%) being significantly greater than those in either annually burned plots (24.7%) or periodically burned plots (16.2%) (Table 2).

Although the interaction between species and treatment was not significant ($p = 0.22$, Table 1), there were differences among species/groups in response to fire regime. For example, both hickories and red oak species exhibited significant reductions in survivorship in periodically burned plots compared with control plots (Table 2). In addition, the survivorship of hickories in periodically burned plots was significantly lower than that of either control plots or annually burned plots (Table 2). In contrast, post oak showed no significant decrease in percent survivorship in annually burned or periodically burned plots compared with control plots.

The 1984 diameter distribution illustrated two ongoing population processes in the experimental plots. First, as the cohorts are followed through time, the form of the distribution based on the total number of living individuals changed from a j-shaped into a bell-shaped curve, reflecting stem exclusion resulting from competition (cf. Figures 1 and 2). Second, growth of hickories was slower than that of either post oak or the red oak group species. The largest diameter of hickories was 28 cm in Replicate 1 and 20 cm in Replicate 2 (Figure 2). In contrast, the largest diameter of post oak was 52 cm in Replicate 1 and 44 cm in Replicate 2. Likewise, the largest diameter of the red oak group was 48 cm in Replicate 1 and 44 cm in Replicate 2. Peaks of the diameter distributions appeared to be shifted among treatments. However, it was not possible to ascertain whether this shift in diameter distribution was due to the treatments or to differences in pretreatment size-class distributions among plots (cf. Figures 1 and 2).

Although the probability of hickory survivorship increased slightly with increasing DBH in control and annually burned plots (Figure 3), its significance is questionable because of the small numbers of surviving hickory trees in 1984. In contrast with hickory species, both post oak and the red oak group species had sizable surviving populations that had grown into larger DBH classes (cf. Figures 1 and 2). The estimated probability of survival of trees of post oak increased with DBH (Figure 3) with most trees with a pretreatment DBH 12 cm or larger surviving until 1984. Additionally, burning treatments had little effect on the relationship between DBH and the probability of survivorship in the post oak (Figure 3). Like post oak, pretreatment diameter had a noticeable effect on the probability of survivorship in the red oak group (Figure 3). However, the effect of pretreatment DBH on survivorship of red oaks differed from that of post oak in two respects.

Table 2: Percent survivorship from 1949 (1951) to 1984. Least square means with different letters are significantly different ($p \leq 0.05$).

Treatment	Species Group	% Survival in 1984
CONTROL PLOTS		
	Hickory	17.9 c
	Post Oak	57.3 ab
	Red Oak Species	40.9 b
ANNUALLY BURNED PLOTS		
	Hickory	2.4 d
	Post Oak	62.1 a
	Red Oak Species	23.0 bc
PERIODICALLY BURNED PLOTS		
	Hickory	0.5 d
	Post Oak	47.2 ab
	Red Oak Species	16.2 c

First, survivorship of red oak species showed distinct peaks at about 18 cm in both control and periodically burned plots rather than simply increasing monotonically with DBH as was observed in post oak. Second, the peak in the probability of survivorship changed with burning regime for red oak group species. Below a pretreatment DBH of 20 cm, the probability survivorship of red oaks was greatest in control plots, followed by annually burned plots, and then periodically burned plots (Figure 3). Above a pretreatment DBH of 20 cm the probability of survivorship of red oaks was greatest in periodically burned plots, less in control plots, and least in annually burned plots (Figure 3). In contrast, the relationship between DBH and estimated survivorship probability differed little among treatments in post oak.

DISCUSSION

The relationships of survivorship with burning regime and pretreatment size depended on species. Burning reduced survivorship of hickory and red oak group species in both annually burned plots and periodically burned plots, but had little effect on survivorship of post oak. Reductions in the survivorship of hickory and red oak species were greater in periodically burned plots than in annually burned plots. Overall, the hickories had low survivorship rates which were further reduced with burning; the red oak group had intermediate survivorship rates that were sensitive to both burning regime and initial tree size; the post oaks had the highest survivorship rates that were relatively insensitive to burning regime and increased monotonically with initial tree size.

Because environmental stresses interact synergistically to increase susceptibility of trees to attack by insects and disease (Houston 1987), other stresses such as drought may account for some of the variation in survivorship among species. There are reports of decline symptoms in oaks in the Missouri Ozarks, particularly in the red oak group, that have been attributed to drought stress occurring in the 1980's (Jenkins 1992). The conservative growth strategy of post oak (Harlow and others 1991) and its high drought tolerance (Ni and Pallardy 1991, 1992) may have given post oak an advantage during and/or after this period of drought. However, while the species of hickory found in the study sites are somewhat sensitive to water stress, red oak group species are noted for being drought tolerant (esp. *Q. coccinea* and *Q. falcata*) (Belanger 1990, Johnson 1990, Sander 1990).

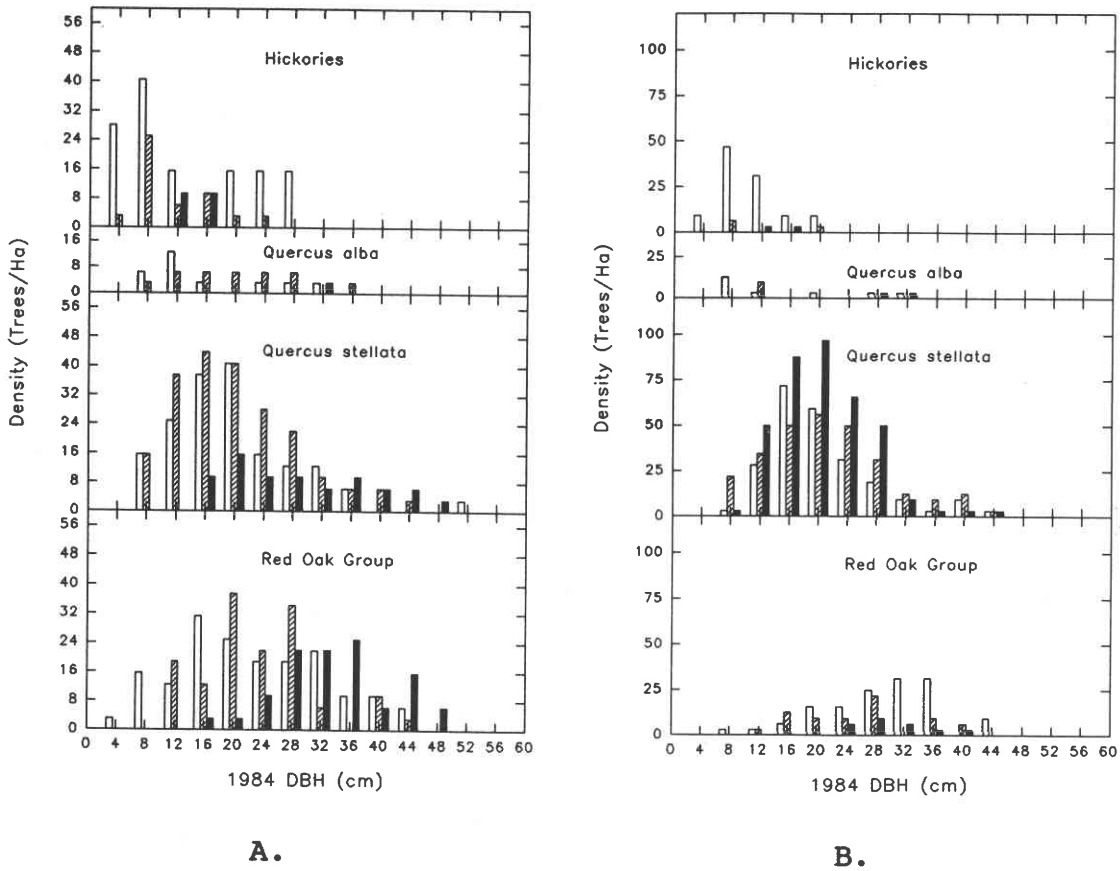

Figure 2: A - Density-diameter relationship in 1984 DBH distribution of species of hickory, white oak and red oak in Replicate 1. B - Density-diameter relationship in 1984 DBH distribution of species of hickory, white oak and red oak in Replicate 2. Open bars - control plots; cross hatched bars - annually burned plots; solid bars - in periodically burned plots.

Because forests at University Forest are between 80 and 90 years old (Jenkins 1992), the effects of differential average life-spans also may have influenced survivorship. White oak group species may live an average of 300-400 years while red oak group species generally live an average of 200 years or less (Harlow and others 1991). Logistic regressions supported the hypothesis that differential life-span accounts for at least some of the higher mortality observed in red oak species compared with post oak. Probability of post oak survivorship increased monotonically with DBH and hence showed no decline in larger, presumably older, trees. In contrast, the red oak group species in two of these burning regimes showed a distinct peak in survivorship probability with respect to DBH indicating mortality of older, larger trees as well as small trees.

As both the DBH*treatment and species*treatment*DBH interaction effects were not significant, while both DBH and species*DBH interaction were, most of the observed species difference in relationships between survivorship and DBH appeared to be the result of self-thinning and senescence-related mortality rather than the result of differential fire response. However, the logistic curves do suggest some species differences in survivorship response to fire regimes. For hickory species, DBH had little effect on survivorship. However, for red oak group species, the relationship between DBH and survivorship predicted by logistic regression differed among burning regimes. The curves suggest that survivorship of red oak group species was sensitive to burning, particularly for smaller diameter

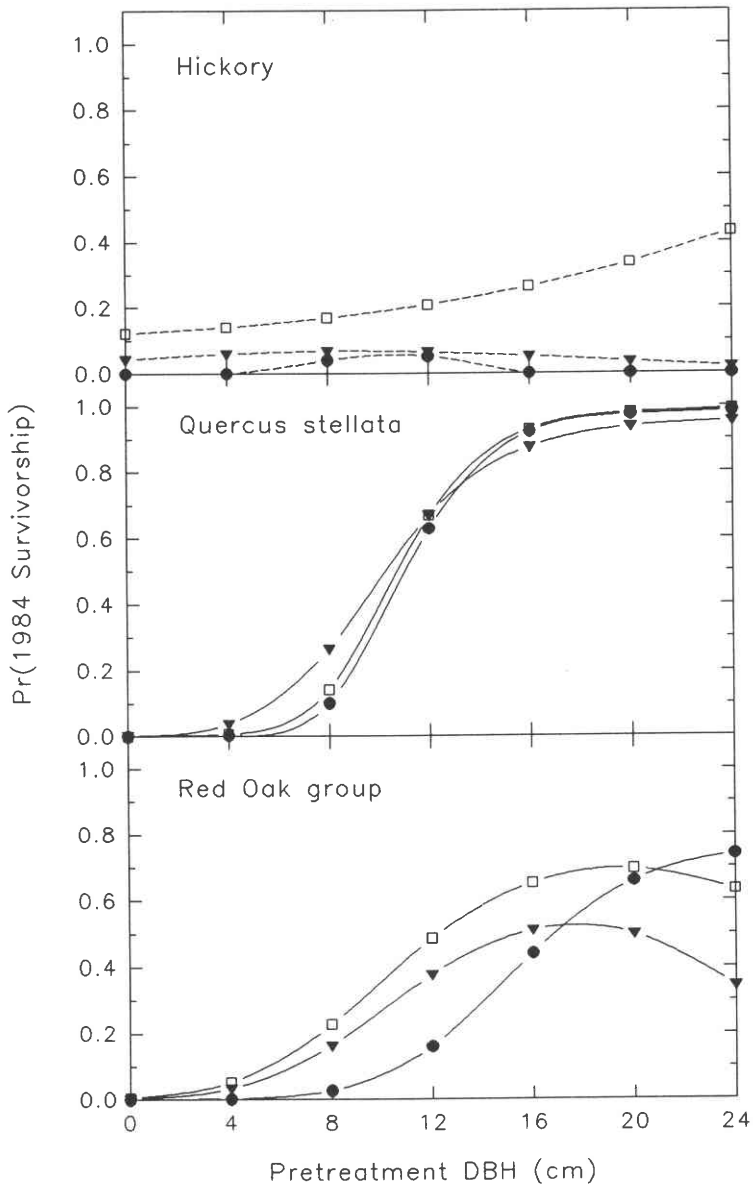

Figure 3: Probabilities of survivorship until 1984 given pretreatment DBH for hickory species, *Q. stellata* and the red oak group. Curves calculated from logistic equations using DBH and DBH^2 terms using all 1984 survivorship data. □ denote survivorship of control plots; ▾ denote survivorship in annually burned plots; ● denote survivorship in periodically burned plots.

trees. The considerable shade tolerance of hickory species (Graney 1990, Harlow and others 1991, Smith 1990) may account for the lack of significance of DBH terms in logistic regression for this species. Burning regime also had little effect on survivorship of post oaks which increased monotonically with DBH. These results suggest that post oak may become more important in oak-hickory forests of the Ozarks because of its longevity and/or superior fire adaptation. Red oak group species, on the other hand, appear to be somewhat more susceptible to fire and inherently shorter-lived. Thus, if a forest is reaching maturity, recruitment and protection of regeneration is important if maintenance of species, particularly short-lived species, is a management objective.

CONCLUSIONS

Annual and periodic burning regimes decreased survivorship of hickory and red oak group species, but had little impact on survivorship of post oak. Reductions in survivorship of hickory and red oak group species were greater in periodically burned plots than in annually burned plots. Survivorship increased with DBH for post oak and peaked at an intermediate DBH for the red oak group species. The relationship between DBH and survivorship appeared more likely associated with self-thinning and life-span of trees rather than with fire treatment effects. However, in red oak species there was some evidence of preferential fire mortality in small diameter trees. Results suggest that when using fire, age, species life-span, stage of stand development and other environmental stresses must be considered if maintaining or increasing the red oak component of an oak-hickory stand is an important management objective.

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CHEMICAL RELEASE OF POLE-SIZED TREES

IN A CENTRAL HARDWOOD CLEARCUT

J. W. Van Sambeek[1], D. Abugarshall Kai[2], and David B. Shenaut[3,4]

Abstract: Our study evaluated the effectiveness of tree injection and full basal bark treatments using three herbicide formulations at reduced or standard practice rates to release crop trees in an overstocked pole-sized Central Hardwood stand. Herbicides tested included glyphosate (Accord), dicamba only (Banvel CST), and dicamba+2,4-D (Banvel 520). The study was conducted in a mixed hardwood stand in southern Illinois that regenerated following a clearcut harvest 18 years earlier. A highly significant interaction occurred among the six herbicide treatments and the method of application. Full basal bark treatment with any of the three herbicide formulations at either rate produced only light (8 to 25%) crown reduction at 45, 90, and 360 days after treatment (DAT). Tree injection with the high and low rate of glyphosate (4 and 8% ai), the ready-to-use dicamba (10.6% ai), and the registered rates of dicamba+2,4-D (1.7 and 2.4% ai, respectively) caused severe (64 to 78%) crown reduction on the treated trees and shrubs. The shrubs as a group were the most susceptible to the herbicide treatments, while sugar maple was the most herbicide-tolerant tree species. The elms, ashes, low-value hardwoods, and high-value hardwood groups were intermediate in response. Non-treated (crop) trees showed no visible crown reductions or reduced diameter growth following chemical release. We conclude tree injection application of glyphosate or dicamba can be an acceptable method for chemical release of crop trees in mixed hardwood stands.

INTRODUCTION

Forest inventories for the Central Hardwoods indicate that the amount of young hardwood growing stock has continued to increase following planting of abandoned agricultural lands and sapling regeneration in clearcuts. Many of the resulting stands are now overstocked with young pole-sized trees of little or no commercial value. Inexpensive, environmentally acceptable methods need to be developed if landowners are going to manage these stands to meet their objectives for wildlife habitat improvement, recreational opportunities, and timber production (Miller and Glover 1991).

Chemical release is a relatively inexpensive method of removing unwanted shrubs and trees when methods are identified that prevent ground water contamination, pose little risk to non-target organisms, and are not labor intensive. In the past, hardwood stands were thinned mechanically by cutting or girdling or chemically released using tree injection and, to a lesser extent, full basal bark treatments. Limited information exists on the effectiveness of several broad-spectrum herbicides developed for agricultural use on hardwood shrubs and trees. This includes using

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both tree injection and basal bark treatments (Griswold et al. 1989). Ideally, these broad-spectrum herbicides should exhibit low toxicity to non-target organisms and would be quickly adsorbed by soil, thus preventing ground water contamination.

Another advantage of chemical release of hardwood stands over mechanical methods would be less drastic changes in stand density as trees slowly die out. The partial crowns of treated trees would provide side shade to the non-target trees reducing chances of epicormic sprouting. In addition, upright decaying stems could provide improved wildlife habitat over a longer time interval.

Label recommendations for chemical release are usually established as the maximum amounts needed to achieve a nearly complete kill of all treated trees and shrubs within a single growing season while still protecting the environment, the user, and the chemical company. Substantial differences in tolerance can exist among different species depending on the herbicide, its formulation, and method of application (Gjerstad and Nelson 1986; Norris 1981). The effectiveness of reducing rates by one-third to one-half are usually not given on the herbicide label. Potentially reduced rates could cause partial crown reduction of treated trees leading to reduced growth and subsequent entrance to a less competitive canopy position.

This study was designed to evaluate the effectiveness of tree injection and basal bark treatments with three herbicide formulations applied at standard practice and reduced rates to chemically thin an 18-year-old mixed hardwood stand. Secondly, we wanted to determine if differences in tolerance exist among different tree species to these herbicides and/or their methods of application. Finally, we wanted to learn if these herbicides affected growth of the adjacent untreated hardwoods.

METHODS

We initiated the study in 1990 on a mixed hardwood stand on a small watershed at the Dixon Springs Agricultural Research Center in southern Illinois. The stand originated following a clearcut harvest 18 years earlier. Stand composition was approximately 35% sugar maple, 10% red bud, 9% slippery elm, 6% sassafras, 6% winged elm, 5% white ash, 4% hickory, 4% black cherry, 4% devil's walking stick, 3% northern red oak, 3% mulberry, 3% yellow poplar, 3% white oak, and 5% other minor trees and shrubs. Soils were primarily Grantsburg silt loams (fine-silty, mixed, mesic Typic Fradiudalfs) with slopes between 3 and 7%, southerly aspect, a pHw of 5.7, and an organic matter content of 5.1%.

In 1989 the stand had been marked to retain approximately 400 non-treatment trees per hectare. These included primarily high-value timber and mast-producing hardwoods (white oak, northern red oak, white ash, hickory, black cherry and tulip poplar). For this study we divided the watershed into forty-eight 8- x 16-m rectangular plots containing 2.8 ± 1.9 ($n = 48$) non-treatment trees each. Twenty-four plots outside the riparian zone were randomly assigned for treatment by the tree injection and 24 additional plots for full basal bark treatment. Plots contained an average of 24.5 ± 13.9 ($n = 48$) treated trees or shrubs and 6.2 ± 1.5 ($n = 48$) different species.

All chemical treatments were applied during the last week of May 1991. For tree injection (hack-and-squirt), we made a series of 4.5-cm-long incisions or frills 3 cm apart into the sapwood conductive tissues with a hand ax approximately 1 to 1.5 m above the ground. Approximately 1 ml of test solution was immediately applied to each incision using a squirt bottle. We tested glyphosate (Accord) at below recommended rate by diluting it with water to give a 10% or 20% (ai) solution. We applied ready-to-use dicamba (Banvel CST) at the recommended rate using an 11.0% solution. We also tested a dicamba+2,4-D mix (Banvel 520) diluted with oil (Androc) containing either 1.3% or 3.9 % dicamba and 3.4% or 10.2% isooctyl ester of 2,4-D, respectively. Trees in the control plots were treated with 1 ml of water per incision. Four replications of each herbicide treatment were randomly assigned to the 24 plots marked for herbicide application by tree injection.

For full basal bark treatments, we wetted the lower 20 to 25 cm of the stem to drip-point using a backpack sprayer equipped with a cone tip. We applied glyphosate at reduced rates as either a 4% or 8% (ai) solution in water without the addition of oil or a surfactant. Dicamba only was applied directly as the ready-to-use aqueous formulation containing 10.6 % ai dicamba. The dicamba+2,4-D mix was applied as a solution containing 0.7% or 1.3% ai dicamba and 1.7% or 3.4% ai 2,4-D diluted with oil. Trees in the control plots were sprayed with water only. Four replications of each herbicide treatment were randomly assigned to the 24 plots marked for basal bark treatment.

We visually estimated average plot crown reduction to within 5% for each shrub or tree species 45, 90, and 360 days after treatment (DAT). A percentage crown reduction of 0% indicated no visible dying leaves or defoliated branches while 100% indicated complete defoliation of the crown with no basal sprouts. To determine treatment responses by species, we divided the treated shrubs and trees into the following classes: maples (essentially all sugar maple), ash (mostly white ash), elms (nearly equal proportions of slippery and winged elms), mixed shrubs (redbud, sassafras, persimmon, sumac, and devil's walking stick), low-value hardwoods (persimmon, mulberry, and various hickories), and the high-value hardwoods (northern red oak, white oak, black cherry, black walnut, and yellow poplar).

The species and diameter at breast height (dbh) of each crop tree were determined in early March 1991 and again in early November 1992 (after two growing seasons). Visual estimations of crown reduction were made also on the non-treated trees 45, 90, and 360 days after application of herbicides to the adjacent hardwoods.

Percent crown reduction was calculated for each plot and subjected to a two-way analysis of variance for a completely random design with factorial arrangement of the five herbicide formulations and water control, the two methods of application, and their interaction (SAS Institute Inc. 1988). We used Fisher's protected LSD (5% t-test value) to determine where differences existed among treatment means for main effects or for their interactions. Terminology for degree of crown reduction (very light to severe) is according to Miller and Glover (1991). We subjected the 2-year dbh growth of non-treated trees to the same two-way ANOVA after excluding the small number of black cherry and tulip poplar crop trees with growth rates nearly double that of the other crop trees.

RESULTS AND DISCUSSION

We found a highly significant interaction among the six herbicide treatments and the method of application for the average percent crown reduction of treated mixed hardwoods (Table 1). This interaction was present at 45, 90, and 360 DAT. Basal bark treatment with glyphosate at both rates, dicamba+2,4-D at both rates, or dicamba ready-to-use resulted in light crown reduction of treated trees. Tree injection was a more effective method for applying glyphosate, dicamba, or dicamba+2,4-D than was full basal bark treatment. The standard practice rates of dicamba and dicamba+2,4-D plus both rates of glyphosate when injected resulted in severe crown reduction of the treated trees. In contrast, the reduced rate of dicamba+2,4-D resulted in light crown reduction compared to the severe crown reduction from the lowest rate of glyphosate. We suspect that severe crown reduction will cause many of these codominant trees to die or become part of the intermediate or suppressed canopy class in the treated plots.

With the full basal bark treatment, only one herbicide (dicamba+2,4-D) at the standard practice rates showed increasing amounts of crown reduction during the second growing season (Table 1). These results suggest that this oil-based herbicide once absorbed into the bark can remain active for more than one growing season. A similar, but less pronounced, pattern existed when the oil-based herbicide was injected. These results suggest expanded studies need to be done comparing injection vs. basal application of other highly effective basally applied herbicides such as triclopyr and picloram with and without 2,4-D. The test herbicides in this study were chosen because of their effectiveness when injected and low risks for environmental damage.

According to Miller and Glover (1991), herbicide treatments resulting in more than a 70% control of the treated trees with acceptable crop tree tolerance should be considered successful. Miller (1990) found that chemically treated hardwoods with over 80% crown reduction eventually died, while trees with less than 40% crown reduction usually recovered. Miller, however, did not indicate if recovered trees now occupied a lower canopy class resulting in less

Table 1.--Crown reduction (%) of treated hardwoods 45, 90, and 360 days after treatment (DAT) with six herbicide formulations using two methods of application.

Method and Herbicide	Conc. Active Ingredient	Percentage Crown Reduction Treated Trees[1]		
		45 DAT	90 DAT	360 DAT
	%	--%--	--%--	--%--
TREE INJECTION:				
Glyphosate	20	74	72	68
Glyphosate	10	63	64	69
Dicamba+2,4-D	3.9+10.2	64	69	74
Dicamba+2,4-D	1.3+ 3.7	23	25	27
Dicamba only	11.0	74	74	78
Water	----	1	1	2
BARK BASAL TREATMENT:				
Glyphosate	8	19	19	18
Glyphosate	4	11	10	5
Dicamba+2,4-D	1.3+3. 4	14	18	31
Dicamba+2,4-D	0.7+1.7	1	1	1
Dicamba only	11.0	25	28	23
Water	----	0	0	0
5% t-test value		19	18	21

[1]Values deviate from those previously reported by Kai et al. (1992) because of method of calculation and correction of values for basal bark treatment with dicamba+2,4-D.

competition to the released trees. Thus, nearly all the herbicide treatments using the injection method would have qualified as being successful assuming no damage to the non-treated trees. Percent crown reduction averaged across all species, however, does not indicate if these treatments resulted in severe to very severe crown reduction of some hardwood species and light to moderate crown reduction for less susceptible species or groups of species.

When percentage crown reductions were analyzed by species groups, we found that the maples were least susceptible to most of the herbicide treatments (Table 2). At the standard practice rate for glyphosate, dicamba, or dicamba+2,4-D, the injection method only resulted in moderate crown reductions on the sugar maples. Basal bark treatment resulted in very light to light crown reductions from which most trees will probably quickly recover. This confirms earlier observations on the high tolerance of pole-sized sugar maples to herbicides (Brinkman 1970, Newton and Knight 1981).

As a group, the shrubs were more susceptible to the tested herbicides than most of the other hardwood species (Table 2). Injections using the reduced rates of glyphosate or dicamba+2,4-D resulted in severe to very severe crown reduction of most shrubs. Although basal bark treatments with reduced rates were ineffective for most of the species groups, they did result in moderate crown reductions for the treated shrubs. Typically, most shrubs have thin bark that may account for their increased susceptibility to these broad spectrum herbicides.

Only minor differences were found in susceptibility among the ashes, elms, and other tree species to the tested herbicide combinations (Table 2). In general, these hardwoods had severe to very severe crown reductions following injections of glyphosate, dicamba, or dicamba+2,4-D. Full basal bark treatments with standard practice rates for glyphosate and dicamba+2,4-D tended to produce light to moderate crown reduction. The response to basal bark application of ready-to-use dicamba gave mixed results. The number of treated trees within any species group never

Table 2.--Crown reduction (%) for each species class 360 days after treatment (DAT) with six herbicide formulations using two methods of application. Changes of more than 10% from percentages after the first growing season are marked with a + or -, respectively[1].

Method and Herbicide	Conc. Active Ingredient	Maple	Ash	Elms	Hardwoods Mixed shrubs	Hardwoods Lesser value	Hardwoods High value
	%	--%--	--%--	--%--	--%--	--%--	--%--
TREE INJECTION:							
Glyphosate	20	41 -	38	100	96	100	100
Glyphosate	10	48	82	69 +	100 +	81 +	87
Dicamba+2,4-D	3.9+10.2	45 +	92 +	100	91	50 +	89 +
Dicamba+2,4-D	1.3+ 3.7	4	0	29	67	2	0
Dicamba only	11.0	38	100	100	100	100 +	77 +
Water	----	0	2	0	0	0	0
BARK BASAL TREATMENT:							
Glyphosate	8	0	17	35	29	26	0
Glyphosate	4	1	0	0	13 -	0	4
Dicamba+2,4-D	1.3+3.4	8	25 +	27 +	50 +	25	33 +
Dicamba+2,4-D	0.7+1.7	0	0	0	0	0	9
Dicamba only	11.0	7	100	15	65	40	19
Water	----	0	0	0	0	0	0
5% t-test value:		19	37	33	39	39	36
Number plots included:		46	28	43	42	39	41

[1]Values for 45 and 90 DAT can be found in Kai (1993).

met the criteria of 30 to 50 test plants as recommended by Miller and Glover (1991) for herbicide efficacy determinations; thus, these observations will require further testing.

Visual inspection of the crop trees did not show any discolored or deformed foliage or deformed shoots. Similarly, we found no differences in stem diameter growth of the crop trees during the first two growing seasons after treatment (Table 3). This suggests that the non-treated trees had absorbed little if any of the herbicides through root grafts or from the soil. In addition, it also indicates the crop trees have not responded to any of the release treatments.

SUMMARY

Chemical release using tree injections of glyphosate, dicamba, or dicamba+2,4-D effectively removed most unwanted hardwoods, except sugar maple, in a mixed hardwood stand. Reduced rates of glyphosate were nearly as effective as the standard practice rates for dicamba-based herbicides. Reduced rates for dicamba+2,4-D were generally unsuccessful except on the thin-barked shrub species. For full basal bark application, ready-to-use dicamba was more effective than either glyphosate or dicamba+2,4-D solutions. Standard practice rates of dicamba+2,4-D in oil using either tree injection or full basal bark application consistently caused more crown reduction during the second growing seasons than during the first growing season. The non-treated or crop trees showed no visible crown reduction or reduced diameter growth in response to any of the herbicide treatments. In conclusion, tree injection application of dicamba or dicamba+2,4-D at standard practice rates or glyphosate at less than standard practice rates can be an acceptable alternative to release crop trees in mixed hardwood stands unless stands are dominated by sugar maple.

Table 3.--Average diameter growth of crop trees two years after chemical release using factorial combinations of six herbicide formulations and two methods of application[1].

Release treatment		Method of Application	
Herbicide	Rate	Tree Injection	Full Basal Bark
		--cm--	--cm--
Glyphosate	High	0.29	0.34
Glyphosate	Low	0.43	0.26
Dicamba+2,4-D	High	0.36	0.31
Dicamba+2,4-D	Low	0.33	0.32
Dicamba only	RTU	0.25	0.34
Water	----	0.37	0.28
5% t-test value		0.16	

[1]Crop trees in order of relative density include white oak, white ash, northern red oak, green ash, sugar maple, and black walnut.

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FOREST VALUES AND HOW TO SUSTAIN THEM

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The forestry profession has the responsibility of managing forests for all of their values. In the past, this has usually not been done. Often, timber has been the only value considered. The emphasis has not been on practices that combine other values with timber production. Such forestry practices must consider forest types and conditions, site quality and characteristics, and the long time needs of the prople. This will require a broad concept of economics and based on all values. These would include watershed protection, erosion control, recreation, aesthetics, wildlife habitat, wilderness management, and the survival of Earth as a home for humanity.

First, we need more foresters in the woods who understand the art and science of ecological management. This usually emulates nature but often on a different time scale. Most values can be obtained together in the same forest at the same time. We must also know the objectives and slant the actions toward them, but not at the expense of the natural ecological requirements of the forest involved. We must use a workable value system, not just "dollar economics".

Except for badly damaged and decadent forests, this means some form of selection harvest and improvement cutting. It requires practices to obtain new regeneration, such as knowledge of opening size to make, tree classes, stand structure and maturity guides are required for optimum residual stands. In the Central Hardwoods, the ecological method for high total values is group selection, not patch cutting or clearcutting.

A total of 42 references based on a 20-year compartment study on Kaskaskia Experimental Forest in Hardin County, Illinois, are given as the chief foundation of this article.

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CHANGES IN A MISSOURI OZARK OAK-HICKORY FOREST DURING 40 YEARS OF UNEVEN-AGED MANAGEMENT

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Abstract: Changes in basal area, density and average diameter were examined on a 156,000-acre privately owned oak-hickory forest in the Missouri Ozarks. The forest has been managed since 1954 using the single-tree selection method. Trees greater than five inches dbh were monitored on 486 one-fifth-acre permanent plots at five year intervals from 1962 through 1992. Seven species/species groups accounted for approximately 90% of both basal area and density throughout this 30-year period. Average basal area increased by 48% and average density by 69% during the study period. Quadratic mean diameter declined by 8.5% over the first 15 years and has since remained relatively stable.

INTRODUCTION

Silviculture is defined as the art and science of manipulating a forest to meet the landowners' objectives (Smith, 1986). Since traditional even-aged management has fallen into disfavor with a large segment of the general population in recent years, there is increased pressure to apply uneven-aged management in systems such as the oak-hickory type (Sander and Graney, 1993). However, little long-term information is available on the growth and compositional changes occurring in oak-hickory forests under uneven-aged management. By applying selection techniques to these systems at this time, the practice of silviculture becomes less of a science and more of an art.

The single-tree selection method has been successfully used to manage shade-tolerant species. Such species are able to develop beneath the continually present canopy of uneven-aged forests. In contrast, shade-intolerant species require a light intensity for their survival and recruitment into the overstory that is often lacking in uneven-aged forests (Oliver and Larson, 1990). It has even been suggested that single-tree selection will not regenerate the oak component of a forest (Sander and Clark, 1971; Sander and Graney, 1983). A major concern in using uneven-aged methods of management then centers on a compositional shift toward more shade-tolerant species, which may be of lesser commercial value (Johnson, 1977; Niese and Strong, 1992). However, most studies that have demonstrated a shift in tolerance were conducted in mesic systems where oak reproduction does not tend to survive and accumulate in the understory. In xeric, oak-dominated systems such as are present in the Missouri Ozarks, oak reproduction does tend to accumulate, so the possibility exists for sustained oak recruitment into the overstory (Johnson, 1992).

To improve our understanding of the long-term effects of uneven-aged management, this paper presents 30 years of data on changes in the composition and basal area of a predominantly oak-hickory forest managed by the single-tree selection method for 40 years. The information represents an initial assessment of uneven-aged management as applied in a xeric to dry-mesic oak-hickory forest in the Missouri Ozarks.

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Management Strategy

The management plan for this forest was developed to meet three objectives: (1) to obtain income from timber sales; (2) to maintain a continuous forest cover; and (3) to refrain from the use of clearcutting in managing the forest. Although maximization of income has not been a primary goal, the forest is required to be self-supporting.

To meet these requirements, the forest plan calls for an approximate 20-year cutting cycle. However, the timing of reentry coincides with the stand reaching 95 - 100 ft^2 of basal area per acre. Upon reentry, the stand is cut to about 65 ft^2 per acre. The operational unit on the forest is the equivalent of one section (640 acres) but topography rather than section lines are used to delimit section units. This reduces adverse site impacts when adjacent sections are harvested.

Only merchantable trees (hardwoods greater than 10 inches dbh and pine greater than 9 inches) are marked for harvest. Culls that suppress potential crop trees are marked and felled as a condition of a timber sale. Lack of markets and the length of time required to grow small diameter stems to merchantable size makes the cost of precommercial thinning prohibitive. Only where a market exists are smaller stems removed.

When a stand is marked for harvesting, the goal is to create or maintain a three-tiered condition comprised of an overstory, a midstory and a sapling/reproduction layer. If the time between entries is too great, small diameter trees are lost and a two-tiered stand results. During marking, each merchantable tree is examined for possible removal. Those deemed likely to die before the next entry are marked for removal. Healthy trees are also considered for harvest. Whenever crop trees are competing, one is removed. Vigor, potential value (e.g. branching patterns limiting merchantable height gain), slope position, aspect and species are used to determine trees to be removed.

Salvage harvests are also made. Marking is accomplished in a manner similar to that of a normal harvest with one exception. If after marking all dead and damaged stems it is determined that the stand will be understocked to the point where the next harvest will be significantly longer than 20 years, all stems of merchantable size are reevaluated as to their ability to survive to the next entry. Any trees deemed likely to survive are left.

METHODS

For forty years this forest has been monitored through the use of well-distributed permanent 0.2 acre continuous forest inventory (CFI) plots. The first inventory was conducted in 1952 with remeasurements made every five-years. In 1957, the current monitoring procedure was established. The number of CFI plots was doubled; one plot was installed for every 320 acres of land on the forest. On each 0.2 acre plot, all stems 5.0 inches dbh and greater, were permanently numbered. Species, dbh, merchantable height to the nearest two-foot class and percent soundness were recorded along with forest-specific product and mortality codes. During subsequent inventories, trees were remeasured, mortality since the last inventory noted and any ingrowth stems were assigned new identification numbers and tallied.

Description of the Forest

The forest covers over 156,000 acres in the Ozark Highlands of southern Missouri. Approximately 90,000 acres were purchased by the current owner in 1954. Additional land was purchased to bring the size of the forest to 134,000 acres by 1962 and to 150,000 acres by 1972. Additional CFI plots were installed as the size of the forest increased. As a provision of the original purchase, the previous owners retained the right to harvest white oak (*Quercus alba* L.) greater than 16 inches dbh. Harvesting of those trees occurred on selected areas of the forest into the early 1960's which left some stands with very low residual stockings. In 1962, 31.6% of the CFI plots supported less than 31 ft^2 of basal area per acre, 86% of the plots had less than 61 ft^2 (Figure 1).

Because of the size of the forest, site quality and vegetation vary widely. However, topography is generally steep with broad flat ridges. The soils are generally rocky and droughty, but range from deep gravel and rock outcrops to areas where the soil is over four feet deep. The soil is derived mainly from dolomitic limestone. Site index (base age 50-years) ranges from 50 to 90 for black oak (*Quercus velutina* Lam.). Although the forest is predominantly oak-hickory, shortleaf pine occurs throughout as scattered trees and as pure stands on upper south-facing slopes. Lower north-facing slopes and deep valleys typically support mixed mesophytic hardwoods.

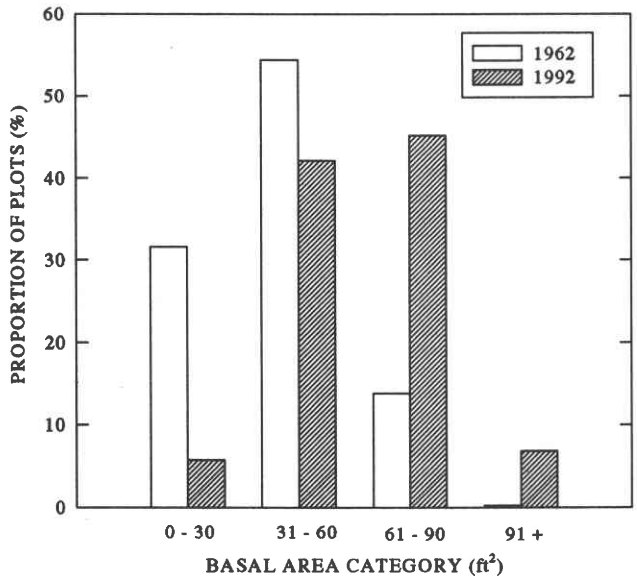

Figure 1. Breakdown of CFI plots, by basal area, at beginning and end of study period.

Analysis

Only the CFI data from 1962 to 1992 are presented in this paper. Basal area (ft^2 per acre), density (trees per acre), and quadratic mean diameter (QMD), were calculated by species for each measurement year. This information reflects a preliminary analysis of the dataset. Data from all available CFI plots have been included in the averages presented. Therefore, because additional property was added to the forest through time, of the landbase included in the 1992 data, 78% had been managed with single-tree selection for 40-years, 8% for 30-years, 10% for 20-years, 3% for 15-years and 1% for 10-years.

RESULTS

Seven principal species/species groups accounted for approximately 90% of both basal area and density throughout the 30-year study period from 1962 to 1992. These species include white oak, scarlet oak, black oak, northern red oak (*Quercus rubra* L.), post oak (*Quercus stellata* Wangenh.), shortleaf pine and hickories (*Carya* spp.). During that time, total basal area on the forest increased by 48% and density by 69% (Table 1). However, of the primary species studied, only scarlet oak maintained the same proportion of basal area and density relative to the entire forest. White oak relative density increased by 12.8% while a similar percentage decline was observed in the remaining species. White oak and shortleaf pine have shown the strongest increase in relative basal area, 8.3 and 5.4% respectively.

Table 1. Average basal area, density and quadratic mean diameter, by year, averaged across species and CFI plots on the forest.

Year	Basal Area (ft^2 / acre)	Density (stems / acre)	QMD (inches)
1962	40.94	86.31	9.33
1967	44.57	97.77	9.14
1972	47.21	107.87	8.96
1977	49.58	124.72	8.54
1982	52.14	131.34	8.53
1987	54.03	134.32	8.59
1992	60.51	145.90	8.72

In absolute terms, five of the seven principal species have increased in density during the study period (Figure 2). Post oak (data not shown) and northern red oak, the two least common of the seven species, maintained relatively stable densities. The hickories, scarlet oak and black oak exhibited similar trends in density change, increasing from 1962 through 1977 and remaining stable thereafter. Density of shortleaf pine has been constant since 1972. Only white oak has continued to show substantial increases throughout the study period and accounts for 87% of the total gain in density occurring on the forest between 1977 and 1992.

The three red oaks (black, scarlet and northern red), exhibited similar trends for both basal area and quadratic mean diameter (QMD) through time (Figure 2). A steady decline was observed in QMD for the first 20 years followed by a recovery. Changes in QMD appear to be closely tied to changes in density. The decline in QMD was accompanied by an increase in density, northern red oak excepted, from 1962 - 1977. Stabilization and recovery of QMD for the red oak group has occurred since the density of black and scarlet oak leveled off. Basal area changes have shown an upward trend with scarlet oak exhibiting the greatest gain followed by black oak and finally northern red oak (Figure 2).

The pattern of change in QMD differed between the white and red oak groups. White and red oaks demonstrated the same initial decline in QMD; however, white oak QMD did not recover while red oak QMD did. Since 1977, white oak QMD has continued to decline, but at a greatly reduced rate. At the same time, white oak density and basal area have increased sharply.

For shortleaf pine, both basal area and QMD have steadily increased through time. The rate of change for these variables appears to have been effected when density peaked in 1977 with QMD increasing and basal area decreasing.

DISCUSSION

Indications are that the composition of this forest is not shifting toward shade tolerant species. The seven species found to be most prominent when single-tree selection was implemented in 1954, still comprise the same relative proportions of the forest today. Density for all species except white oak has been relatively constant since 1977 (Figure 2). White oak, which has high value both for timber and mast production, is increasing in prominence on the forest. Its density has increased over three-fold and its basal area has more than doubled during the 40-year monitoring period. As a proportion of the forest total, in 1962 white oak comprised less than 18% of the total basal area. In 1992, 26% of the basal area was in this valuable species.

Ingrowth and the harvesting of timber both have a suppressing effect on QMD. Since ingrowth is measured and averaged into the plot total when a tree reaches five-inches dbh, if QMD is larger than 5.0 inches any new ingrowth will bring about a reduction in QMD value. Likewise, if QMD is smaller than the ten-inch minimum harvest diameter, cutting any tree will lower the QMD. Only diameter increases of trees greater than five-inches dbh will increase the QMD.

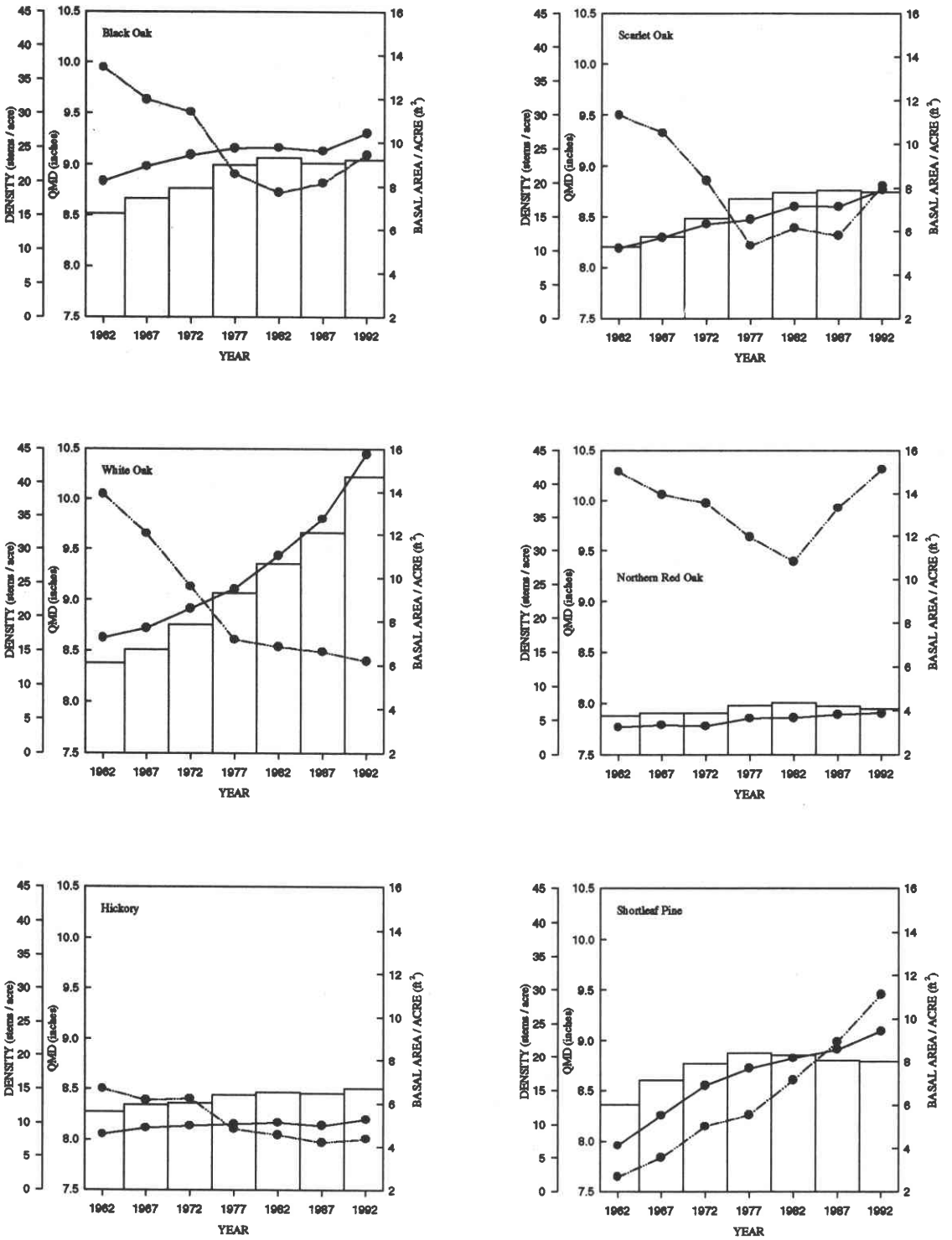

Figure 2. Changes in quadratic mean diameter (broken line), basal area per acre (solid line) and density per acre (hollow bars), for six species/species groups under uneven-aged management practices.

Average basal area has remained relatively stable for northern red oak and hickory, slowly increased for black oak and scarlet oak, markedly increased for shortleaf pine and increased exponentially for white oak (Figure 2). With black oak, scarlet oak and shortleaf pine, this increase in basal area mirrors the increase observed in QMD. Since the density of these species is stable (ingrowth equals the sum of harvest and mortality), both basal area and QMD will also likely stabilize when the growing capacity of the forest is reached.

The large increase observed in white oak basal area paralleled a similar increase observed in density. The fact that the QMD has declined only slightly since 1977 (0.2 inches) while density has continued to increase, indicates that considerable diameter development is occurring. Numbers and size of mortality and trees harvested were not available at the time of manuscript preparation; therefore, changes in density that are reported represent an extremely conservative estimate of ingrowth and its consequent effect on QMD suppression. The sharp early drop observed in white oak QMD is believed to be due to the ten stem per acre increase which occurred between 1962 and 1977, almost doubling the species' density (Figure 2). In this same 15-year period, basal area increased by 2.2 ft^2 per acre. Since ten, five-inch stems account for only 1.4 ft^2, the remaining 0.8 ft^2 (plus any basal area removed by mortality or harvesting) had to result from the growth of the remaining stems. Since 1977 the evidence for diameter development is even more dramatic. Again, almost a doubling of density was observed (23.5 to 41 stems per acre). However, during this time period the sharp decline in QMD observed earlier, did not occur. The 17 additional white oak stems tallied account for 2.4 ft^2 of the 6.2 ft^2 of basal area added since 1977.

As of the last remeasurement period (1992), the single-tree selection management approach appears to be maintaining a healthy, sustainable forest. Ingrowth into the five-inch diameter class is occurring at a rate sufficient to maintain or increase density for all of the principal forest species even after accounting for harvested stems. Diameter development is occurring as evidenced by the stabilization of QMD for the forest as a whole while average basal area and density are steadily increasing (Table 1). However, even though single-tree selection has been applied for 40 years, the forest can not yet be considered regulated. This is evidenced by the continuing increase in basal area and density which can only occur while there is available growing space. The forest is obviously still recovering from the low stocking levels which were present at the time ownership was transferred; as of 1992, approximately 45% of the forest still supported less than 60 ft^2 of basal area per acre (Figure 1). Despite the lack of full regulation, initial indications support Johnson's (1992) suggestion that under the conditions present in this xeric to dry-mesic, oak dominated ecosystem, selection silviculture may result in sufficient recruitment to perpetuate the oak-hickory type.

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FOREST FLOOR CO_2 FLUX FROM TWO CONTRASTING ECOSYSTEMS IN THE SOUTHERN APPALACHIANS

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Abstract: We measured forest floor CO_2 flux in two contrasting ecosystems (white pine plantation and northern hardwood ecosystems at low and high elevations, respectively) in May and September 1993 to quantify differences and determine factors regulating CO_2 fluxes. An automated, IRGA based, flow through system was used with chambers inserted into the soil. This approach allowed quantification of diurnal flux patterns which were subsequently averaged to estimate daily mean flux rates (μmol m^{-2} s^{-1}). Mean flux rates were 60 percent greater in the white pine ecosystem (8.9 μmole m^{-2} s^{-1}) than in the northern hardwood ecosystem (5.6 μmole m^{-2} s^{-1}). Across ecosystems and sample dates, the most important regulating factor was soil temperature ($r^2 = 0.70$; $p < 0.0001$). Mean (24-hr) soil temperature (at 5 cm depth) was 2.5 °C lower in the northern hardwood stand relative to the white pine stand. All other parameters considered (i.e., soil C:N, root mass, root C:N, litter C:N, litter mass) did not explain the differences in flux rates between sites, but variation in fine root mass and litter C:N did explain spatial and temporal variation within the northern hardwood site. These results indicated that at large spatial scales, variation in soil temperature was more important in regulating forest floor CO_2 flux than factors more closely associated with the species composition and productivity of the sites (e.g., litter and root mass and quality).

INTRODUCTION

Carbon dioxide (CO_2) evolution from the forest floor is due to the metabolic activity of roots, mycorrhizae, and soil micro- and macro-organisms. Although precise estimates of carbon (C) recycled to the atmosphere from belowground sources are unavailable, Raich and Schlesinger (1992) propose that the belowground contribution exceeds 70 Pg $year^{-1}$ globally. This represents a major component of C flux in the global C cycle. Belowground C cycling processes and subsequent forest floor CO_2 fluxes are equally important at ecosystem scales; however, we have limited knowledge of the magnitude of fluxes within and across ecosystems. Increased knowledge of the magnitude of C fluxes, as well as the factors which regulate these fluxes is critical for understanding ecosystem C cycling and potential effects of forest management or other factors such as climatic change. In this study, we quantified forest floor CO_2 flux in two contrasting ecosystems: a low elevation 36-yr-old white pine plantation and a high elevation mature northern hardwood stand.

Separating the contributing sources (i.e., roots vs. microbes) of forest floor CO_2 flux has proven difficult. The relative contribution of roots versus other soil components has been estimated to vary between 35 to 65% of the total CO_2 evolved (Edwards and Harris 1977, Ewel and others 1987, Bowden and others 1993). Factors influencing the rate of CO_2 evolution include soil temperature and moisture (through their influence on metabolic activity of both roots and microbes) (Edwards 1975, Schlentner and Van Cleve 1985, Weber 1985), soil organic matter (Ewel and others 1987), soil and root nitrogen (N) levels (Söderström and others 1983, Ryan 1991), and root biomass (Behera and others 1990).

Several techniques are available for measuring CO_2 evolution from the forest floor. Static chamber methods include soda lime or bases (KOH or NaOH) which measure CO_2 "trapped" over the measurement interval (see Cropper and others 1985). Static measures of CO_2 evolution may also be made by gas chromatograph analysis of air samples collected from sealed chambers on the soil surface (Raich and others 1990). de Jong and Schappert (1972) describe a

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variation of the static method by using a chamberless technique based on CO_2 profiles (pCO_2) in the soil. Dynamic chamber methods quantify CO_2 evolution by continuously monitoring CO_2 levels in chambers with either a closed or flow-through system and an infrared gas analyzer (IRGA). Studies comparing measurement techniques have found wide disparity between static chamber, static chamberless, and dynamic chamber methods (Edwards and Sollins 1973, Cropper and others 1985, Raich and others 1990, Rochette and others 1992, Norman and others 1992). In general, static chamber techniques provide lower estimates of CO_2 evolution than dynamic chamber techniques, while pCO_2 techniques provide higher CO_2 evolution estimates than dynamic chamber techniques (de Jong and others 1979). Although more difficult and expensive to conduct, dynamic, IRGA based techniques are considered more reliable (Ewel and others 1987) and they can be configured to quantify diurnal patterns.

The objectives of our study were: (1) to quantify and contrast forest floor CO_2 evolution in two ecosystems in late spring and summer using a dynamic, IRGA based measurement system, and (2) to qualitatively assess the importance of regulating factors such as, fine and coarse root biomass and C:N, soil temperature, litter mass and C:N, and soil C:N.

METHODS

Site Description

The study was conducted at the Coweeta Hydrologic Laboratory in the southern Appalachians of western North Carolina, USA. Two sites were selected for the present study (Table 1). Watershed one (WS1) is a 16.1 ha, 36-year-old white pine plantation (*Pinus strobus* L.). The watershed has a southerly aspect and spans an elevation range of 705 to 988 m. The site selected for study was located in the lower portion (≈715 m) of the watershed. Watershed 27 (WS27) is a 39 ha, ≈85-year-old mixed hardwood watershed. The watershed has a northeast aspect and spans an elevation range of 1061 to 1454 m. The site selected for study is in the upper portion (≈1375 m) of the watershed and contains a northern hardwood forest type.

The range in elevation and aspect between the two watersheds results in differences in climatology. At lower elevations, mean annual precipitation averages ≈1800 mm, while at higher elevations mean annual precipitation averages ≈2200 mm (Swift et al. 1988). Air temperature is also substantially lower (10-15%) at higher elevation sites (Swift et al. 1988).

Table 1. Summary of stand and site characteristics for the white pine and northern hardwood study sites.

Variable	White Pine	Northern Hardwood
elevation (m)	715	1375
stand age (years)	36	≈85
aspect	S	NE
trees ha^{-1}	1015	405
basal area (m^2 ha^{-1})	53.2	32.1
major species	*Pinus strobus* L.	*Quercus rubra* L. *Quercus prinus* L. *Acer rubrum* L.

Forest Floor CO_2 Flux Measurements

Sampling was conducted on four consecutive days in May and September 1993. In the southern Appalachians, May is a period where biological activity is beginning to occur and September is a period of active biological activity. Soil CO_2 flux was measured for 20-22 hrs (i.e., a diurnal cycle) using an automated, flow-through, IRGA based measurement system. Data were averaged to provide an average flux rate (μmol m^{-2} s^{-1}) over the entire sampling interval. The system measured flux sequentially from five soil chambers (10 cm diameter, 10 cm height, 785 cm^3 volume) constructed of PVC pipe. Soil chamber edges were sharpened on the open end and driven approximately 2 cm into the soil surface (at random locations) with a rubber mallet. All tubing was 5 mm (i.d.) flexible PVC. Air was passed through the chambers via inlet and outlet fittings attached to the upper sides of the chamber. Air flow through the chambers was regulated with a dual-sided air pump (Spec-Trex Corp.) which balanced flow into and out of the chambers. Actual flow rate (ml min^{-1}) was controlled by varying voltage (0-12 VDC) supplied to the pump and was measured and logged electronically with a flow meter and data logger (Campbell 21X). An air flow rate of 800 to 1000 ml min^{-1} provided stable readings within 7 to 8 minutes. Chamber sampling was controlled with a multiplexer, data logger, and solenoids which opened sequentially (chambers 1 - 5) at ten minute intervals. Carbon dioxide concentrations of air entering and exiting the chambers was measured and logged electronically with an IRGA (ADC LCA3) operating in differential mode and a data logger (Campbell 21X), respectively. Forest floor CO_2 flux (μmole CO_2 m^{-2} s^{-1}) was calculated based on the difference in CO_2 entering and exiting the chamber, the soil area sampled beneath the chamber, and the flow rate. Only data from the last minute of sampling were used in flux calculations.

Litter and Humus Measurements

Litter and humus were removed from beneath the chambers after flux measurements, dried, weighed, ground, and analyzed for N and C concentration (Perkin-Elmer 2400 CHN Analyzer).

Root Biomass Measurements

Root biomass, separated into fine (< 2 mm) and coarse (>2 mm) fractions, was determined using coring. After flux measurements, a 10 cm diameter metal pipe was placed in the exact chamber location and driven to a 30 cm depth with a mallet. The core was removed and the contents were placed in paper bags, transported to the laboratory, and washed over a fine mesh screen where live and dead roots were visually separated. Roots were dried, weighed, ground, and analyzed for N and C concentration (Perkin-Elmer 2400 CHN Analyzer).

Soil Measurements

Soil N and C (Perkin-Elmer 2400 CHN Analyzer) were determined on a sub-sample of soil from the cores. Soil temperature (5 cm depth) was measured with Type-T thermocouples and a datalogger (Campbell 21X). Soil moisture was not measured.

Statistical Analyses

Differences in temporal and spatial site means of average flux rates were determined with analysis of variance (SAS 1987). Stepwise regression analyses were used to relate between and within site variation (spatial and temporal) in average flux rates to soil temperature, litter mass, fine and coarse root mass, and the quality (i.e., C:N ratio) of soil, roots, and litter (SAS 1987). In all cases, $\alpha = 0.05$ was used for statistical significance and selection of significant parameters in multiple regression analyses.

RESULTS AND DISCUSSION

Forest Floor CO_2 Flux

The magnitude of forest floor CO_2 flux varied considerably between ecosystems and sample dates (Table 2). For example, averaged across sample dates, the flux rate for the white pine stand was 8.9 *u*mole m^{-2} s^{-1} versus 5.6 *u*mole m^{-2} s^{-1} for the northern hardwood stand (differences significant at $p<0.05$). Averaged across sites, May flux rates were also significantly ($p < 0.01$) lower than September flux rates (Table 2). Variation in flux rates within and between ecosystems has been observed in other studies (Garrett and Cox 1973, Hanson and others 1993). For example, Hanson and others (1993) found a maximum 2-fold variation in forest floor flux rates between ridge and valley locations within the same watershed. The values obtained in our study are in the upper range of those observed for many ecosystems (e.g., Weber 1985, Hanson and others 1993); however, comparison of rates with studies using other measurement techniques should be done with caution. Where measurement techniques were similar, our rates are in the range of values obtained by others (e.g., Edwards and Sollins 1973, Ewel and others 1987).

Table 2. Forest floor CO_2 flux by site and date (n = 5 for each sample date and site; † indicates significant [$p < 0.05$] difference between sites for mean flux rate; ‡ indicates significant [$p < 0.05$] difference between sample dates within a site).

Site	Date	Forest Floor CO_2 Flux (standard error)
White Pine	May	5.20(1.28)
	September	11.80(1.59)‡
		Mean = 8.87(1.52)
Northern Hardwood	May	3.22(0.12)
	September	7.46(1.61)‡
		Mean = 5.57(1.13)†

Regulating Abiotic and Biotic Factors

There was substantial variation in most abiotic and biotic factors between and within sites (Table 3). Coefficients of variation ranged from 12 to 118% for the northern hardwood ecosystem and from 19 to 87% for the white pine ecosystem. Based on the results from previous studies, higher forest floor CO_2 flux rates should occur in conjunction with warmer soils, lower C:N ratios in soil and litter, higher root biomass (especially fine roots). Regression analyses using data from both sites and sample periods indicated that temperature was the primary factor regulating spatial and temporal variation in forest floor CO_2 flux across ecosystems (Table 4). This emphasizes the importance of temperature in regulating heterotrophic and autotrophic activity in these ecosystems and indicates that temperature regulation may override variation in biotic factors at large spatial scales (i.e., between ecosystem types occurring at different climatic regimes). In our study, this was true even when the variation in ecosystem type (i.e., pine vs. hardwood ecosystems) and corresponding biotic components was quite large (Table 3). Other studies have also demonstrated the importance of temperature in determining forest floor CO_2 flux (Hanson and others 1993, Peterjohn others 1993). Soil and litter moisture has been shown to influence CO_2 flux in some studies (e.g., Hanson and others

Table 3. Means (n = 10), standard errors (SE), and coefficients of variation for ecosystem parameters used in regression analyses relating forest floor CO_2 flux to abiotic and biotic parameters across and within ecosystem types.

	White Pine		Northern Hardwood	
Parameter	Mean (SE)	Coefficient of Variation %	Mean (SE)	Coefficient of Variation %
Fine Root Mass ($g\ m^{-2}$)	432.4(72.3)	50.2	517.4(102.5)	59.4
Coarse Root Mass ($g\ m^{-2}$)	491.7(151.2)	87.0	762.0(299.9)	118.1
Fine Root C:N	63.9(4.1)	19.1	55.1(2.6)	14.3
Coarse Root C:N	110.6(10.7)	27.3	92.6(9.7)	31.4
Litter Mass ($g\ m^{-2}$)	1018.7(201.8)	59.4	938.0(114.8)	36.7
Litter C:N	40.1(2.5)	18.7	37.4(2.5)	20.0
Soil C:N	23.6(2.9)	36.3	21.2(0.9)	12.4
Soil Temperature (°C)	17.4(1.1)	18.8	15.6(0.7)	13.9

Table 4. Regression equations relating forest floor CO_2 flux to abiotic and biotic driving variables. All variables are significant at $P < 0.05$.

Model Type	Model	r^2	F	P>F
Across Ecosystems	Flux = -14.211 + 1.321 (soil temperature)	0.70	34.6	0.0001
w/in Northern Hardwood	Flux = 13.884 + 0.0099 (fine root biomass) -0.3604 (litter C:N ratio)	0.90	26.7	0.0010
w/in White Pine	Flux = -13.381 + 1.325 (soil temperature)	0.84	32.3	0.0013

1993). While we did not measure soil moisture, litter moisture in our study was always greater than 50%. In addition, we explained from 70 to 90% (see below) of the variation in forests floor CO_2 flux without accounting for variation in soil moisture. This suggests that soil moisture was not a dominant factor regulating spatial and temporal variation in forest floor CO_2 flux in our study.

Within the northern hardwood ecosystem, spatial and temporal variation in fine root mass and litter C:N ratio were important regulators of forest floor CO_2 flux (Table 4). Roots can contribute as much as 60% to forest floor CO_2 flux so it is not surprising that fine root mass is significantly and positively related to forest floor CO_2 flux. Litter quality (i.e., C:N ratio) is an important parameter regulating decomposition rate and the negative regression coefficient indicates less forest floor CO_2 flux (i.e., decomposition) as litter quality decreases. These results contrast with those found across ecosystems, where only soil temperature was related to spatial and temporal variation in forest floor CO_2 flux. Hence, during late spring and summer, within site variation in forest floor CO_2 flux was driven primarily by variation in biological components (i.e., root mass and litter quality) rather than soil temperature. We are reasonably certain, however, that temporal variation in soil temperature within the northern hardwood ecosystem would be an important variable if measurements in winter months were also included.

In the pine ecosystem, soil temperature was the only statistically significant factor regulating temporal and spatial variation in forest floor CO_2 flux (Table 4). It is noteworthy that some of the other parameters (i.e., soil C:N, coarse root C:N, and coarse root mass) were marginally significant ($p < 0.10$) when included in multivariable regressions. This indicates that while temperature is the most important factor, other factors may also be important and larger sample sizes are required to detect statistical significance.

SUMMARY AND CONCLUSIONS

Based on measurements in early spring and summer, forest floor CO_2 flux rates varied considerably (60 percent) between the white pine and northern hardwood ecosystems. Flux rates are a function of multiple and complex abiotic and biotic factors which vary in time and space. Between ecosystems, temperature was the most important driving variable; however, within the hardwood ecosystem, variation in fine root mass and litter quality were important. Hence, the relative importance of driving variables depends on the scale of study and the magnitude of variation in climatic, edaphic, and biological parameters within and between ecosystems. The short-term study presented here provides some interesting preliminary insights, however a more complete understanding of these relationships will require a much more intensive and extensive study. Our current research is focusing on including more ecosystem types and more intensive measurements (i.e., monthly sampling intervals).

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ACID-BASE STATUS OF UPPER ROOTING ZONE SOIL IN DECLINING AND NON-DECLINING SUGAR MAPLE (*ACER SACCHARUM* MARSH) STANDS IN PENNSYLVANIA

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Abstract: Sugar maple (*Acer saccharum* Marsh) is an important commercial tree species of the central hardwood region which is valued for its wood and maple sugar products. High elevation sugar maple stands in northcentral Pennsylvania have been in serious decline for about the last 15 years with more than 1,200 hectares of maple forest affected. The decline appears to be largely irreversible, leading to the death of the affected trees. Stands at lower slope positions remain generally healthy. The objective of this study was to investigate the role of soil conditions in causing sugar maple decline by studying the acid-base status of soils, plant available soil aluminum and the relative differences in soil Al toxicity to maple seedling roots in declining (47-79 percent mortality) and non-declining (<10% mortality) stands. The acid-base status comparisons for the upper rooting zone (A, B1, B21 horizons) of soils of the sugar maple stands under study are reported here. Five declining and five non-declining stands in close proximity to one another were sampled with soil samples taken from the side walls of hand-excavated shallow soil pits in each stand. The samples were analyzed at the Agricultural Analytical Services Laboratory at The Pennsylvania State University for pH (water paste), and exchangeable Ca, Mg, K and P. Percent base saturation was also calculated. Results indicated that soils from the declining stands had significantly lower exchangeable Ca, Mg and K than the non-declining stands. pH and base saturation were also consistently lower in the declining stands while P was higher. The significance of these differences in acid-base status as a predisposing factor in the observed sugar maple decline requires additional study.

INTRODUCTION

Sugar maple decline is not a new phenomenon. Sugar maple declines have been reported in eastern North America since the early 1950's (Hendershot and Jones 1989). In most cases unusual weather events such as droughts and early frosts or episodes of insect defoliation have been cited as factors contributing to or causing declines (Bauce and Allen 1991, Kolb and McCormick 1993). However, recent severe declines of sugar maple in the Canadian provinces of Ontario and Quebec have prompted investigations into the likelihood that soil acidification and its concomitant suite of potential nutrient deficiencies and imbalances may be a contributing or predisposing factor in sugar maple decline (Kinch 1989, Hendershot and Jones 1989, Bernier and Brazeau 1988a and 1988b). Aluminum is also a potential contributor to these problems through direct toxicity to fine roots (Thornton and others 1986), by interference with nutrient uptake (Thornton and others 1986, Kelly and others 1990), and by causing precipitation of phosphorus to nonavailable forms (Kinch 1989). Liming and fertilization of declining sugar maple stands in Canada with P and K has been shown to improve decline symptoms (Adams and Hutchinson 1992, Hendershot 1991, Kinch 1989, Ouimet and Fortin 1992).

The purpose of this study was to test the hypothesis that the soils of declining ridge-top sugar maple stands were more acidic than those of the non-declining stands and that this condition has resulted in nutrient deficiencies, reduced radial growth and root development. We report here on the acid-base status of the upper rooting zone (A, B1, and B21 horizons) of soils from the ten stands under study.

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METHODS

Study Area

Ten sugar maple stands were chosen for study based on the severity of decline within each stand (Table 1). All of the stands were located within a 10 km^2 area along Rock Ridge Road in southern Potter County, Pennsylvania (Figure 1). The study area is part of the Susquehannock State Forest. The declining stands are located at elevations around 700 m. The non-declining stands were located at elevations around 560 m. Stands that had been salvage logged were not included in the study. This made selection of declining stands more difficult since extensive salvage logging of declining sugar maple stands has occurred along Rock Ridge Road.

Five stands with presently identifiable mortality of dominant and co-dominant sugar maples ranging from 47 to 79 percent were characterized as declining. Five additional stands with estimated mortality of 0 to 8 percent were characterized as non-declining. Mortality was assessed based on the mean number of dead sugar maples on three 0.1 ha plots within each stand. Any tree with a living branch, no matter how small, was considered living. Any dead standing or down tree with sufficient attached bark to allow for field identification was counted as dead. Mortality was no doubt underestimated particularly in the declining stands because many trees had been lost in years previous to this study that could no longer be identified as sugar maple and thus were not included in the tally. This fact was evident both from the number of stems on the ground and the very poor stocking of stands exhibiting the worst decline.

Trees with greater than 50 percent crown loss were also tallied. If these trees are added to the dead trees in each stand, the declining stands would have 76 to 89 percent of sugar maples in this category and the non-declining stands would have 0 to 11 percent.

Soils

Soils are classified according to the Soil Survey of Potter County (SCS 1953). The non-declining stands based on their plotted locations on the soil survey map are mostly of the Lackawanna Soil Series. These are for the most part channery silt loams. Declining stands are mapped primarily as Leetonia, Wharton, Bath or Nolo Soil Series. These are stony loamy sands and silt loams.

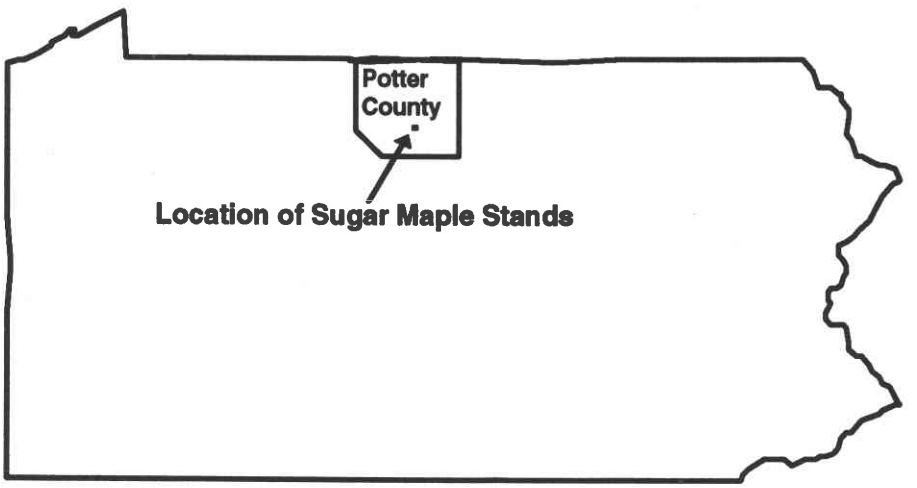

Figure 1. Location of the sugar maple stands within the Susquehannock State Forest in Potter County, Pennsylvania.

Table 1. Sugar maple mortality in the stands under study.

Stand	Mean % dead trees	Mean % dead and >50% crown dead trees
Declining		
D7	78.9[1]	89.0
D8	74.9	88.6
D4	63.8	85.2
D5	53.1	84.8
D6	47.4	76.2
Non-declining		
ND6	8.3	11.0
ND7	7.0	7.0
ND2	2.2	4.3
ND5	0.0	0.0
ND1	4.5	4.5

[1]Mean percentages of all declining vs. non-declining stands in both dead and dead plus >50% crown dead categories were significantly different at $\alpha \leq 0.05$.

Soil samples were obtained from the face of hand-excavated shallow soil pits. Samples were collected by individuals wearing plastic gloves by scraping soil from a clean pit face with a large plastic spoon. Soil descriptions for the soils mapped on each site were used to determine soil profiles in the field. Composite samples were collected from the A, B1, and B21 horizons in each soil pit. A total of 15 samples were collected from each stand. All samples were collected near living sugar maple trees. Samples were placed in plastic bags and transported to the Agricultural Analytical Services Laboratory where they were analyzed for pH, CEC, exchangeable Ca, Mg, K, and P, and acidity by the methods described in NDSU (1988). A percent base saturation was also calculated.

Statistical Analysis

Soil chemistry data were analyzed using the SAS mainframe computer package (SAS 1985). Summary statistics, including means, standard deviations, and standard errors, were calculated for each soil chemical parameter in each forest stand using the MEANS procedure. Significant differences in soil chemistry between declining and non-declining stands were determined using a two-factor nested analysis of variance (PROC NESTED in SAS). Significant differences were determined at $\alpha \leq 0.05$ unless otherwise noted. The Tukey Studentized Range Test was used to test for significant differences in mortality between stands.

RESULTS AND DISCUSSION

The results of the soil chemical analysis are presented in Table 2. The pH of the soils for the declining stands was lower than that of the non-declining stands but the difference was not significant (P=0.29). Base saturation was also lower on the declining stands but was also not significant (P=0.14). The mean base saturation of the soils in declining stand 5 was 10.2 percent while the mean of the other four declining stands was only 5.25 percent. Without stand 5 the difference in base saturation between declining and non-declining stands was significant. There were very few live trees remaining in stand 5 and most of these were at the edge of the former stand; consequently, it is possible that sampling near these trees in this stand biased the results toward more favorable soil chemistry.

Table 2. Comparisons of mean soil chemistry for declining and non-declining stands.

	Declining	Non-declining	P-value[1]
pH	4.24	4.46	0.29
exch. Ca (meq/100 g)	0.56	1.04	0.03
exch. Mg (meq/100 g)	0.17	0.25	0.05
exch. K (meq/100 g)	0.09	0.14	0.04
exch. P (lbs/acre)	50.6	18.9	0.06
% base sat.	6.36	9.39	0.14

[1]Probability that the means are not significantly different.

Base saturation values for forest soils of less than 10 percent are problematic with respect to aluminum availability (Cronan and others 1989). Both the declining and non-declining stands had base saturation values of less than 10 percent although the non-declining mean was 9.39 percent. Our estimates of exchangeable Al (0.01 M $SrCl_2$ extractable) are not complete as of this writing, but it is expected that they will be high in both declining and non-declining stands.

Estimates of exchangeable Ca, Mg and K indicate significant differences between declining and non-declining stands (Figures 2, 3 and 4). All of these essential nutrients are less available in the declining stands.

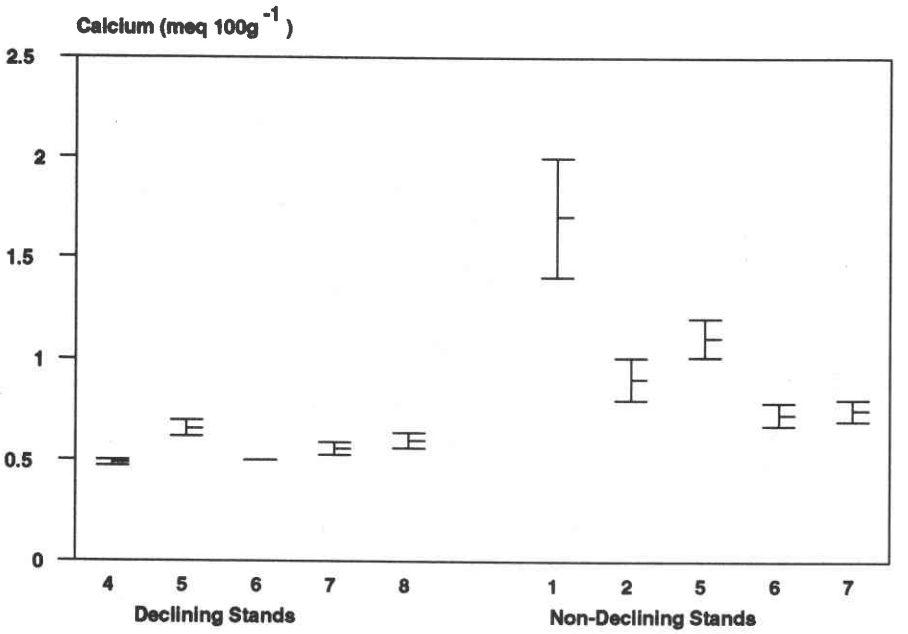

Figure 2. Mean calcium concentrations in soil from each of five declining and five non-declining stands studied. Bars around each mean indicate ± one standard error.

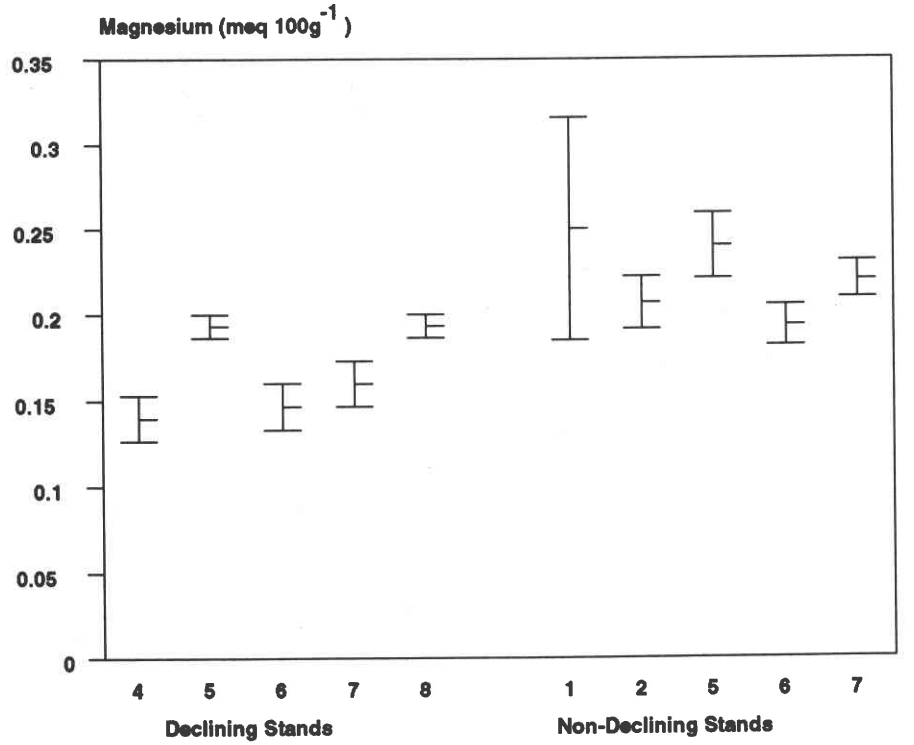

Figure 3. Mean magnesium concentrations in soil from each of the five declining and five non-declining stands studied. Bars around each mean indicate ± one standard error.

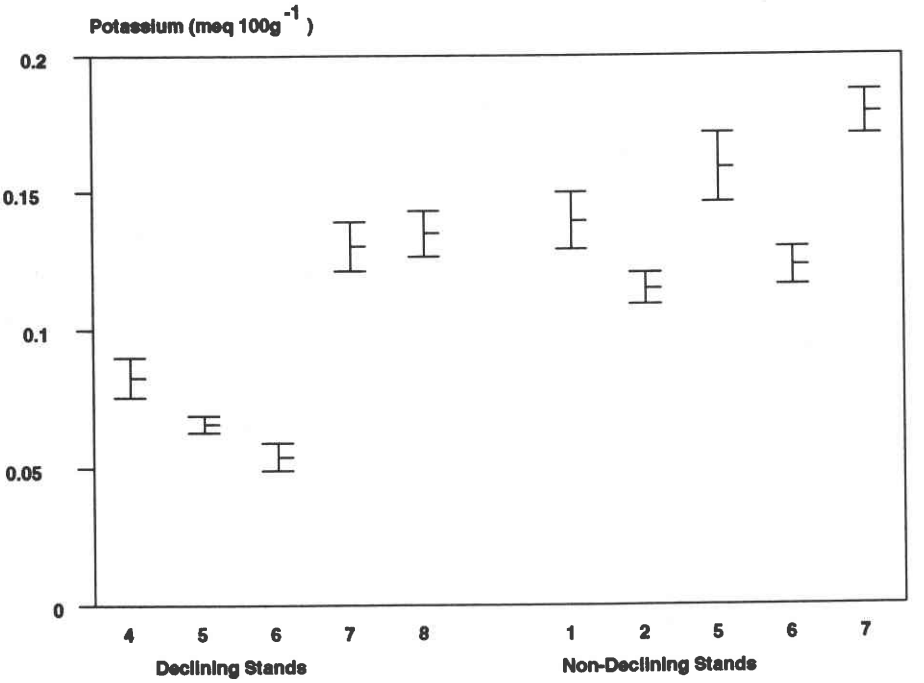

Figure 4. Mean potassium concentrations in soil from each of the five declining and five non-declining stands studied. Bars around each mean indicate ± one standard error.

This condition alone could be sufficient to explain the difference in decline severity between the stands studied. Kolb and McCormick (1993) reported Mg and possibly Ca deficiencies in the foliage of our declining stand 4 and an additional declining stand on the Susquehannock State Forest. Kinch (1989) reported that Ca, Mg and P were potentially deficient in sugar maples growing in declining stands and that the foliar concentrations of these elements were strongly predicted by soil Al and Ca concentrations and pH. Similarly Roy and others (1985) reported that declining sugar maple stands were often deficient in soil Ca and Mg.

The declining stands had higher exchangeable P than the non-declining stands (P=0.06). We tested whether or not this was a consequence of the extraction procedure (Mehlich III) utilized by the Agricultural Analytical Services Laboratory versus a water extraction method suggested by Bingham (1966). Estimated available P was much lower with water extraction. However, declining stands still had higher P although the difference betweeen declining and non-declining stands was not significant. Precipitation of insoluble aluminum phosphate in the presence of high concentrations of available soil Al and low soil pH (<pH 6) has been reported (Cole and Stewart 1982, Cook 1983, Hsu and Bates 1964). If insoluble P were extracted along with available P, as may be the case with the Mehlich III procedure, acidic soils with high P values would have the most fixed, unavailable P; consequently, they could test high for available P but be P limited. Data presented by Kolb and McCormick (1993) did not indicate a P deficiency in our declining stand 4. Foliar analysis currently underway should also indicate whether or not there may be less P available in the declining vs. non-declining sugar maple stands in this study.

SUMMARY AND CONCLUSIONS

The acid-base chemistry of the upper rooting zone soil from five declining and five non-declining sugar maple stands was evaluated. Upper rooting zone soil from the declining stands had lower pH and base saturation, higher exchangeable P and significantly lower exchangeable Ca, Mg and K. This finding is consistent with the results of other studies of soil chemistry in declining sugar maple stands and with the hypothesis that reduced availability of Ca, Mg and K is a potential predisposing factor in the observed decline. Addition of Ca, Mg and K in proper balance should be attempted as a remedy for decline and as a way to encourage sugar maple reproduction in the damaged stands in this area.

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TREE-RING CHEMISTRY RESPONSE IN BLACK CHERRY TO AMMONIUM SULFATE FERTILIZATION AT TWO WEST VIRGINIA SITES

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Abstract: The chemical element content of black cherry (*Prunus serotina* Ehrh.) tree rings showed significant changes related to annual ammonium sulfate treatments on one watershed (Fernow WS-3) which exhibited a significant increase in streamflow N export due to treatment. However, tree-ring, soil and streamflow chemistry did not respond to the same treatment on another watershed (Clover Run WS-9). On WS-3, tree-ring concentrations of P, K, Ca, Mg, B, Zn, S and Sr were higher and concentrations of Mn, Fe, Cu and Al were lower in wood formed either before or after treatment. Lack of response to treatment on WS-9 was largely due to soil retention of N, lack of nitrate leaching and possibly the location of the sampled trees near the watershed mouth. Overall, black cherry tree-ring chemistry was consistent with streamflow chemistry changes caused by treatment and the method shows promise as an index to soil chemistry changes. Sapflow in rings formed prior to treatment and/or radial translocation prevented an exact determination of the year of treatment initiation.

INTRODUCTION

Study of the chemical element content of tree rings, or dendrochemistry, is receiving increased attention as a method of determining changes in the soil chemical environment. Several studies have been conducted which showed, or attempted to show, a link between tree-ring chemistry and soil chemistry. A study by Bondietti and others (1990) with red spruce (*Picea rubens* Sarg.) suggested a link between atmospheric deposition and altered soil chemistry. The relationship between soil pH and tree-ring chemistry was examined by Guyette and others (1992) with Eastern redcedar (*Juniperus virginiana* L.) and DeWalle and others (1991) with a variety of species. Effects of liming were studied by Kashuba-Hockenberry and DeWalle (1994) with scarlet oak (*Quercus coccinea* Muenchh.), DeWalle and others (in press) with red oak (*Q. rubra* L.) and McClenahan and others (1988) with tulip-tree (*Liriodendron tulipifera* L.). No known studies have focused on tree-ring response to soil treatment with known quantities of ammonium sulfate. In addition, concern about the effects of radial translocation or sapflow in xylem raises doubt about the utility of using tree rings to measure the timing of soil changes.

Experiments at two research watersheds maintained by the USDA Forest Service, Northeastern Forest Experiment Station in north central West Virginia (Adams and others 1993) provided an opportunity to determine the dendrochemical response to soil fertilization with ammonium sulfate in black cherry (*Prunus serotina* Ehrh.) trees. In earlier studies with black cherry, DeWalle and others (1991) showed that cation concentrations, such as Mn and Sr, in sapwood were sensitive to soil pH variations. In this paper, we compare the black cherry tree ring chemistry on control and treated areas and in wood formed before and after treatment to determine possible effects of ammonium sulfate treatment and radial translocation/sapflow.

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STUDY AREAS

Experiments were conducted on watersheds maintained by the USDA Forest Service, Northeastern Forest Experiment Station on or near the Fernow Experimental Forest near Parsons in north central West Virginia (39°3'15"N, 79°41'15"W). Treatment and control watersheds on the Fernow were WS-3 and WS-7, respectively. In the Clover Run area, watershed WS-9 and a nearby control area were sampled. WS-9 and the control area are on the Monongahela National Forest near Parsons. At both the Fernow and Clover Run sites, the treatment and control areas are adjacent to one another. Land use, current and past experimental treatments, and dominant vegetation are described for each watershed in Table 1.

Table 1. Comparison of treatments and land use history for watersheds WS-3, WS-7 and WS-9.

Watershed	Treatments	Dates	Area (ha)	Dominant Vegetation
WS-3	Weir Installation	5/51	34.3	*Prunus serotina*
	Intensive selection cut	10/58-2/59		*Acer rubrum*
	Repeated cut	9/63-10/63		*Betula lenta*
	Patch cuttings w/herbicide	7/68-8/68		*Fagus grandifolia*
	Clearcut, left stream buffer	7/69-5/70		
	Cut buffer, clear channel	11/72		
	Begin ammonium sulfate treatment	1/89		
WS-7	Weir Installation	11/56	24.2	*Acer saccharum*
	Upper 12.1 ha clearcut	11/63-3/64		
	Herbicide upper cut	5/64-10/69		*Betula lenta*
	Lower 12.1 ha clearcut	10/66-3/67		*Liriodendron tulipifera*
	Herbicide lower cut	5/67-10/69		*Prunus serotina*
WS-9	Farmed	1800s to 1920s	11.6	*Larix leptolepis*
	Weir Installation	1957		*Quercus rubra**
	Bulldozed forest regrowth, left stream buffer	11/83		*Prunus serotina**
	Planted to Japanese larch	Spring 1984		*Smilax rotundifolia*
	Began ammonium sulfate treatment	4/87		*Rubus* spp.

*buffer zone along stream.

Chemical applications on the two treatment watersheds for this study, WS-3 and WS-9, consisted of aerial applications of granular ammonium sulfate fertilizer (21:0:0:24= N:P:K:S) three times per year to give an annual total application of 167 kg ha^{-1}. Treatments were designed to double the seasonal and annual inputs of N and S occurring in throughfall on these watersheds. Applications varied seasonally with 33 kg ha^{-1} added generally in March and November and 101 kg ha^{-1} added in July on each watershed. Applications began in April 1987 on WS-9 and in January 1989 on WS-3. Details of the treatment are given by Adams and others (1993).

The region is part of the Allegheny Plateau with steep slopes and shallow soils. Soils are predominantly loamy-skeletal, mixed mesic Typic Dystrochrepts of Calvin channery silt loam type derived from shale and sandstone from the Hampshire formation. Precipitation averages about 145 cm per year with even seasonal distribution. Mean monthly air temperatures average about -1 to 5° C in January to about 25° C in July. Adams and others (1994) give a detailed description of the hydrometeorology of the Fernow Experimental Forest region.

METHODS

Wood samples were taken from trees growing on two pairs of treated and control areas using different methods due to size of trees. On WS-3 and WS-7, 3-cm thick wood disks taken at breast height were harvested from five pole-sized black cherry trees felled at each site on July 29, 1992. On July 2-3, 1992, 4-mm diameter increment cores were collected with a standard increment borer at breast height at four locations around the circumference of five mature black cherry trees in the streamside buffer zone on WS-9 and the control area. Other species also were sampled, but black cherry was the only species sampled that occurred at Fernow and Clover Run sites.

Wood disks from WS-3 and WS-7 were sanded, then a band saw was used to cut the wood disks into quadrants and quadrants into radial strips. Radial strips were separated into age-class sections using a utility knife under low power magnification. Wood from each quadrant was divided into the following sections according to the years in which wood was formed:

Treatment Period	1989-1992
Pre-treatment I	1986-1988
Pre-treatment II	1981-1985

The sanded edges were removed prior to chemical analysis with a stainless-steel chisel to avoid chemical contamination by the sand paper. Samples from each quadrant were analyzed separately, but for the purposes of this paper data for all quadrants were combined.

For Clover Run sites, increment cores also were separated by hand with a scalpel under low magnification into three sections based on ring age:

Treatment Period	1987-1992
Pre-treatment I	1982-1986
Pre-treatment II	1977-1981

Wood from each of the four cores per tree was combined by sections for each tree to give enough mass for chemical analysis.

Sterile laboratory rubber gloves were used for all sample collection, handling and preparation. Samples were rinsed with deionized water several times during processing and were stored frozen until submitted for analysis. Wood samples were analyzed for P, K, Ca, Mg, Mn, Fe, Cu, B, Al, Zn, Na, Sr, N and S at the Agricultural Analytical Services Laboratory operated by the College of Agricultural Sciences at Penn State University, University Park, PA. Nitrogen analysis was conducted using the Dumas method (Campbell 1991) and sulfur analysis using ICP emission spectroscopy (Huang and Schulte 1985). Analysis for the other elements was also by ICP emission spectroscopy (Dahlquist and Knoll 1978).

Soil samples were collected in spring 1994 at rooting depth (0-15 cm) around sample trees on WS-9 and its control area. Samples were collected around black cherry (n=5) and red oak trees (n=5) in the treated and control areas. Soil chemical analyses were performed at the Agricultural Analytical Services Laboratory at Penn State using standard soil tests (North Dakota State Univ. 1988). Since soil chemistry did not vary between species at a site, soil data were

grouped for later comparisons between treatment and control areas. Statistical analysis involved simple t-tests to determine significant differences between treatment and control areas in soil data and tree-ring data by age classes.

RESULTS AND DISCUSSION

Treatment Effects at Each Site

Black cherry tree-ring chemistry changes in response to ammonium sulfate additions on WS-3 were quite pronounced (Table 2). Wood formed during the period 1989-1992, after initiation of treatment on WS-3, showed significant increases in P, K, Ca, B, Sr and S and significant decreases in Mn concentrations relative to WS-7. Pre-treatment wood on WS-3 formed during the 1986-88 (Pre-treatment I) period showed significant increases in P, K, Ca, Mg, B, Zn, Sr, and S and significant decreases in Al, Cu, Fe, and Mn concentrations. For the 1981-85 (Pre-treatment II) period, wood on WS-3 also showed significantly higher Ca, Mg, and Sr and significantly lower Mn, Cu, and Na relative to WS-7. Interestingly, more numerous and generally larger significant differences occurred in wood formed immediately prior to initiation of treatment (Pre-treatment I) than in wood formed after treatment. Nearly all of the differences between WS-3 and WS-7 were significant at the $p \leq 0.01$ level.

Table 2. Mean tree-ring element concentrations (mg/kg) for five black cherry trees on watersheds WS-3 (T=treated) and WS-7 (C=control) for periods of wood formation after and before initiation of soil ammonium sulfate treatments.

Element	Treatment 1989-92		Pre-treatment I 1986-88		Pre-treatment II 1981-85	
	C	T	C	T	C	T
P	122	146.0**	50.5	66.0**	20.0	29.5
K	1015	1213.0*	356	421.0**	225	234
Ca	431	502.0**	331.5	504.0**	171	320.0**
Mg	103	113	65	98.0**	27.5	57.0**
Mn	76.3	22.4**	66.4	22.5**	29.0	14.0**
Fe	10.9	10.9	8.1	5.25*	6.25	8.4
Cu	1.87	1.78	1.5	1.29*	1.76	0.98*
B	3.45	3.73*	2.72	2.98*	2.62	2.73
Al	8.6	1.7	2.9	1.30*	1.1	1.1
Zn	2.65	5.7	1.72	3.49*	2.49	2.06
Na	11.15	10	9.85	8.3	7.35	5.85*
Sr	3.91	5.22*	3.47	5.36*	2.78	4.08*
N	1273	1321	897	950	743	722
S	82.3	90.4*	60.0	64.4*	61.8	53.9

* = significant difference between C and T at alpha ≤ 0.05 level, ** at alpha ≤ 0.01 level

In contrast, mean black cherry tree-ring chemistry at WS-9 showed relatively minor response to ammonium sulfate applications (Table 3). Tree rings on WS-9 formed during the 1987-92 period, after treatment initiation, showed significant increases in only Ca and Fe concentrations relative to the control area. Wood formed for the five-year period immediately prior to treatment (1982-86) also showed Fe and Na increases and a decrease in S concentrations.

Table 3. Mean tree-ring element concentrations (mg/kg) for five black cherry trees on watershed WS-9 (T=treated) and a nearby control area (C) for periods of wood formation after and before initiation of soil ammonium sulfate treatments.

Elements	Treatment 1987-92		Pre-treatment I 1982-86		Pre-treatment II 1977-81	
	C	T	C	T	C	T
P	90	80	60	50	20	20
K	670	700	460	420	230	290
Ca	390	430.0*	390	390	200	250
Mg	120	130	110	90	40	50
Mn	49.2	56.4	45.6	46.8	18.6	24.6
Fe	49.8	97.8*	27.6	84.2*	27.4	94.8
Cu	2.1	1.8	1.7	1.4	1.5	1.4
B	3.4	3.4	3.0	2.9	2.9	2.7
Al	3.8	4.4	2.9	3.3	2.9	2.4
Zn	154.0	30.2	112.2	17.6	68.0	22.0
Na	17.2	11.4	13.0	9.4*	11.0	7.4
Sr	3.6	3.6	3.6	3.5	2.7	2.9
N	840	910	640	610	520	570
S	460	460	440	430.0*	440	440

* = significant difference between control and treatment at alpha $\leq$ 0.05 level

All of these changes were significant at the $p \leq 0.05$ level. Lowering $p \leq 0.1$ would only have added significant changes in Zn and Na in Pre-treatment II samples and in Zn in Post-treatment samples, respectively. Pre-treatment wood formed during the 1977-1981 period showed no significant differences relative to the control area.

Causes of Treatment Differences between Sites

Large differences in tree-ring response to ammonium sulfate treatment occurred between watersheds WS-3 and WS-9. Increased base cation concentrations in wood on WS-3 are consistent with the mobilization of base cations in soil solution to maintain a charge balance with the anions added by treatment. Sulfate anions were added directly in the ammonium sulfate treatment and nitrification of added ammonium would produce NO_3 anions. Nitrogen fertilization from the added ammonium also may have increased rates of decomposition of soil organic matter which increased the overall supply of base cations. Sulfur increases in wood are most likely a direct result of the soil treatment. Reductions in trace metal concentrations (Al, Cu, Fe, Mn) may have been caused by increased base cation availability. Tree-ring trace metal concentration reductions on WS-3 also imply that soil acidification due to three years of treatment has not occurred yet, since soil acidification should generally lead to increased availability of trace metals.

In contrast to WS-3 results, six years of ammonium sulfate treatment on WS-9 produced very small to negligible tree-ring chemistry effects. Lower site fertility caused by years of mountain farming on Clover Run WS-9 (Adams and others 1993) may have depleted available soil base cations and produced low soil N pools. Consequently, N may be very limiting on WS-9 and more strongly retained. Data for treated watersheds show lower tree-ring Ca and N content on WS-9 than WS-3 in all years (Tables 2 and 3), supporting the suggestion of lower site fertility on WS-9 as a cause of reduced treatment response in tree rings. However, trees sampled on WS-9 were located in a streamside

buffer zone near the mouth of the watershed and may reflect soil conditions in this zone only. Helicopter additions of ammonium sulfate included the streamside buffer zone.

Increases in tree-ring Fe concentrations in wood formed during the 1987-92 and 1982-86 periods on WS-9 may signal initial stages of soil acidification caused by treatment. Treatment and control area differences in tree-ring Na and S at Clover Run cannot be related directly to treatment and may be due to natural site differences.

Soil chemistry data were used to test the hypothesis that ammonium sulfate treatments have affected soil properties near sampled trees at the Clover Run sites. Mean chemistry for 10 soil samples around sampled trees in the streamside buffer zone on WS-9 and the control area (Table 4) showed no significant differences at either the $p \leq 0.05$ or $p \leq 0.1$ levels. No significant change in measured soil properties had occurred due to treatment around sample trees in the buffer zone on WS-9.

Table 4. Comparison of root-zone soil chemical properties around sample trees in the streamside buffer zone on watershed WS-9 and nearby control site.

Parameter	Control	WS-9
pH	4.57	4.43
P (kg/ha)	17.3	11.4
Acidity (meq/100g)	8.37	9.59
K "	0.075	0.084
Mg "	0.16	0.18
Ca "	0.63	0.61
CEC "	9.24	10.44

* indicates significant difference at the 0.05 level

Published streamflow chemical export data provide some checks on the tree-ring results for both sites. Differences in black cherry wood chemistry response to ammonium sulfate treatment between watersheds WS-3 and WS-9 were consistent with differences in NO_3 and Ca export in streamflow in response to treatment on these watersheds (Adams and others 1993). On WS-3, where tree-ring chemistry was markedly changed by treatment, large increases in NO_3 and Ca export were noted. On WS-9, where little tree-ring response was found, no significant increases in NO_3 or Ca were found. No significant increases in streamflow export of SO_4 were found on either watershed. Soil sulfate adsorption probably is responsible. Increased export on WS-3 indicates that treatment indeed did mobilize cations and tree-ring chemistry was able to record those changes. Lack of streamflow export changes on WS-9 also is consistent with no soil chemical changes and limited tree-ring chemistry changes in response to treatment on this watershed.

Some soils data do exist to help interpret dendrochemical results. Gilliam and others (1994) found that inorganic soil NO_3-N concentrations at the 0-10 cm depth were significantly higher on WS-3 than on WS-7 after 3 years of treatment in support of the stream chemistry differences. Mean soil calcium concentrations on WS-3 were also much greater than on WS-7 but the differences were not significant due to high soil Ca variability on WS-3. Differences in soil K, Mg, P, NH_4-N, Cu, Fe, Mn, and Zn between WS-3 and WS-7 were not significant. The authors do cite unpublished data for the 0-5 cm soil depth which showed that soil on WS-3 was significantly more acidic than on WS-7. Although these soil data show NO_3 changes on WS-3 due to treatment, large changes in base cation availability as suggested by tree-ring chemistry data are not supported.

On the Clover Run WS-9 basin, Pickens and others (1995) present soils data which show that significant changes in soil Al, Mn, N, S, Ca, Mg, and pH occurred due to ammonium sulfate treatments. Only small tree-ring Ca increases could be attributed to treatment on WS-9. Tree ring samples were collected in the streamside buffer zone near the mouth of the watershed on WS-9, while soils data from Pickens and others (1995) were collected in the uplands portion of this basin. Considering that both tree-ring and stream chemistry data (Adams and others 1994) do not indicate acidification of WS-9 due to ammonium sulfate treatments, soils data presented by Pickens and others (1995) suggest a gradual acidification of soils on WS-9 from the uplands to the lowlands which has not yet affected the tree-rings, soils and water at the mouth of the basin.

Apparently, cation mobilization is largely due to NO_3 rather than SO_4 anions on WS-3. No significant changes in tree-wood N concentrations were found, even though NO_3 concentrations were increased in soil solution. Kashuba (1992) also found that scarlet oak tree-ring chemistry did not change in response to soil N fertilization. Tree-ring chemistry is more sensitive to base cation element availability changes, than N changes, since base cations can bind to pectic compounds in the cell wall of xylem. (Ferguson and Bollard 1976; Tomlinson and Tomlinson 1990). Nitrogen is not similarly adsorbed.

Radial Variations of Treatment Effects

Contrary to expectations, wood formed prior to, rather than following, treatment appeared to be affected most on WS-3. A larger number of significant differences were observed for the 1986-88 period, just prior to treatment, than for wood from the 1989-92 period occurring after treatment. Several explanations can be given for this occurrence: 1) radial translocation of elements could have occurred which would redistribute elements from post-treatment to pre-treatment wood, 2) previous watershed treatments could have affected tree rings on WS-3 or WS-7 in such a way that wood formed before ammonium sulfate treatment was inherently different between watersheds, and 3) ammonium sulfate treatment effects on WS-3 may be changing gradually over time.

Radial Translocation/Sapflow. Radial translocation of elements in xylem often has been suggested as an explanation of a treatment response in tree rings formed in years prior to treatment (McClenahen and others 1988). Parenchyma cells in wood rays represent living tissue, and radial translocation of carbohydrate in rays has been documented. Thus, radial translocation of chemicals could have occurred from rings affected by treatment to rings formed before treatment. However, this issue is controversial since tree rings formed in years just prior to treatment can also be affected by treatment, if those rings were participating in vertical sapflow at the time of treatment. In tree species with diffuse porous wood, such as black cherry, sapflow occurs in a band of sapwood of indeterminate thickness. By definition, sapflow cannot occur in heartwood. The number of annual rings involved in sapflow during treatments in this experiment is not known, but observations of sapwood thickness can at least be used to indicate the number of rings potentially available for sapflow.

Sapwood included most if not all of the wood analyzed on the two watersheds. The heartwood/sapwood boundary observed when the wood samples were cut in 1992 was located between 4-9 annual rings from the bark (mean 6.1 years) on WS-7 and 7-11 annual rings from the bark (mean 8.5 years) on WS-3. Since wood only up to 11 years of age at the time of sampling was considered in this study and the sapwood/heartwood boundary probably migrated toward the bark after treatment, sapflow probably could have occurred at some time after treatment was initiated in most, if not all, of the 11 annual rings analyzed.

Response to ammonium sulfate treatment in wood formed immediately prior to the treatment period (1986-88) could be easily explained by sapflow at treatment time in these three rings. Treatment effects on wood formed during the 1981-85 period implies that one or more annual rings formed 4-8 years prior to treatment also contributed to sapflow when treatments were initiated. Sapflow in at least some of these rings cannot be ruled out.

Effects of Past Treatments. Experimental treatments on Fernow watersheds WS-3 and WS-7 prior to the time of ammonium sulfate treatment may have contributed to differences in the chemistry of black cherry wood, but this is not considered likely. These watersheds were subjected to different forest cutting and herbicide treatments as part of

earlier experiments (Table 1); however, the trees on each basin had been allowed to regrow naturally for at least 19 years prior to ammonium sulfate treatments. Wood formed only up to 8 years prior to ammonium sulfate treatment was analyzed in this study. This wood probably was not affected significantly by prior experimental treatment, but this can not be established with certainty.

Changing Treatment Effects. Since ammonium sulfate treatments were applied each year from 1989 through 1992 on WS-3, gradually changing soil response to treatments also could have influenced radial variations. Continued ammonium sulfate treatments are expected eventually to deplete base cations, cause soil acidification, and increase availability of trace metals. Timing of such base cation depletion/acidification cannot be predicted. That wood formed after treatment showed fewer and smaller significant differences than wood formed before treatment suggests that base cation depletion had begun after only three years of treatment, although no direct evidence for soil acidification was found. As treatment of WS-3 continues, tree-ring chemistry may eventually show soil acidification effects.

CONCLUSIONS

The chemical element content of black cherry tree rings is an indicator to the effects of ammonium sulfate fertilization on the soil which is at least as sensitive as measurements of streamflow or soil chemistry. Sapflow and/or radial translocation in black cherry wood may interfere with the detection of the precise timing of soil changes using dendrochemistry.

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ELEMENTAL CONCENTRATIONS IN FOLIAGE OF RED MAPLE, RED OAK, AND WHITE OAK IN RELATION TO ATMOSPHERIC DEPOSITION IN PENNSYLVANIA

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Abstract: Foliage was sampled in June and late August-early September in 1988 and 1989 from the outer crowns of codominant red maple (*Acer rubrum* L.), northern red oak (*Quercus rubra* L.), and white oak (*Q. alba* L.) trees in forest stands along an atmospheric deposition gradient in north-central Pennsylvania. Leaf samples from approximately 480 trees were analyzed for concentration of macroelements N, P, K, Ca, Mg, and S, as well as microelements Al, B, Cu, Fe, Mn, Mo, Na, Pb, Si, Sr, V and Zn. Foliar concentrations of Fe, S, and Sr were generally greater in the high deposition portion of the gradient. Concentrations of other foliar elements generally were not related to the spatial pattern of atmospheric input.

INTRODUCTION

Concentrations of specific elements (e.g. sulfur and trace elements) in plant tissue have been used to indicate the magnitude of atmospheric pollutant input to forest ecosystems. Several studies have been conducted in the oak forests of Pennsylvania USA which indicate that certain elements accumulate in plant tissues subjected to elevated pollution levels. For example, the sulfur content of tree foliage in woodlands of western Pennsylvania was related to distance and direction from a complex of large coal-fired power plants (Hutnik and others, 1989). Strontium concentrations of white oak (*Quercus alba* L.) xylem in a mixed-oak forest of central Pennsylvania were related to distance from a small coal-fired power plant (Long and Davis, 1989). In addition to these point-source studies, Lynch (1990) documented the existence of an atmospheric deposition gradient in north-central Pennsylvania based on precipitation monitoring conducted since 1982. He reported that pH, sulfate, nitrate, ammonium, and calcium ions in precipitation generally followed a west-to-east pattern of decreasing concentrations. The westernmost part of this forested gradient receives high sulfate loadings, as well as some of the most acidic rainfall in the USA (Figs. 1, 2). Lynch (1990) reported that wet $SO4^{=}$ deposition varied from 38 Kg.ha-1.yr-1 in study area 1 in the west to 30 Kg.ha-1.yr-1 in study area 4 in the east. Likewise, wet $NO3^{-}$ deposition varied from 23 to 20 Kg.ha-1.yr-1 across this gradient, and mean annual precipitation pH varied from 4.07 to 4.14. Historical seasonal mean temperatures have not varied significantly across the region of this study (Kingsley, 1985), but annual precipitation varies from 109.2 cm.yr-1 in study area 1 to 96.5 in study area 4 (Davis and others, 1990). Physiographic and soil factors along the gradient have been characterized (McClenahen and Long, 1993), but have not been analyzed with respect to atmospheric deposition. Variation in heavy metal burdens within vegetation may also occur along this gradient, as evidenced by Showman and Long (1990) who reported that concentrations of Al, Cr, Cu, and Fe in lichen thalli were significantly greater in the western, more polluted part of this deposition gradient.

A multi-disciplinary research project was initiated in 1986 to examine forest health along this deposition gradient (Davis and others, 1990). Using a rigorous statistical procedure, Long and others (1991) identified 13 ecologically analogous forest stands, clustered in four distinct "core areas" on the gradient (Fig. 1). These core areas served as focal points for a series of multi-faceted studies. At or near these four core areas, elemental analyses were conducted on samples of rainfall, soils, and plant tissue collected from both the herb-layer and canopy-layer. This report deals

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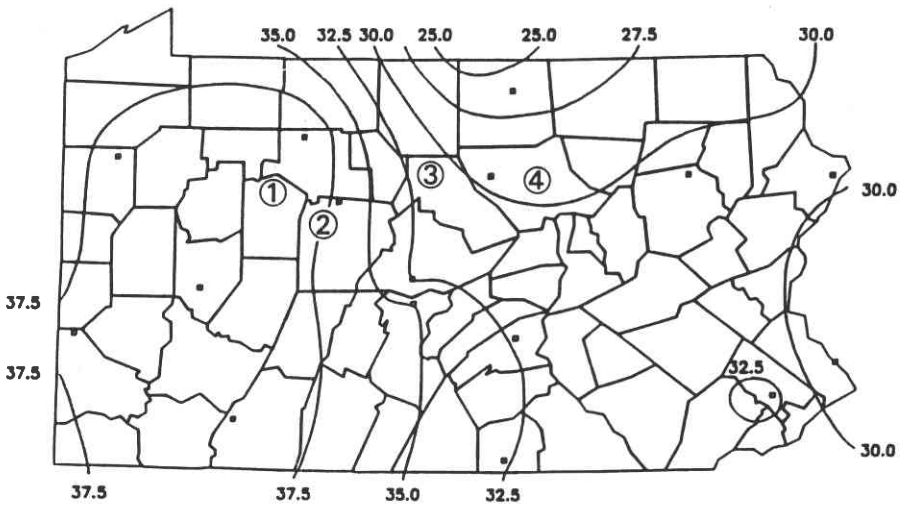

Figure 1. Spatial distribution of multiannual wet sulfate ion deposition (kg/ha) in Pennsylvania from 1982 - 1988. Numbers 1 - 4 indicate approximate location of the four core areas which served as focal points for the Pennsylvania Deposition Gradient research. The small squares represent atmospheric monitoring locations (Lynch, 1990).

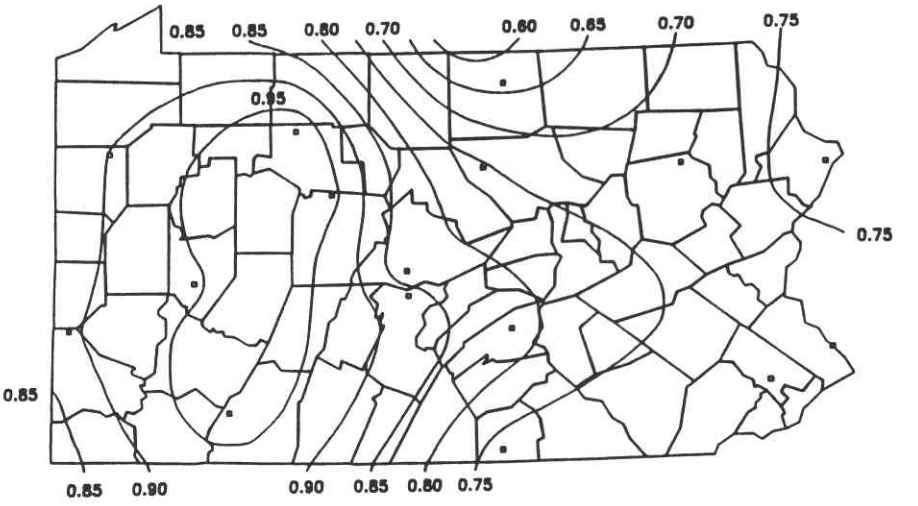

Figure 2. Spatial distribution of multiannual mean wet hydrogen ion deposition (kg/ha) in Pennsylvania from 1982 - 1988 (Lynch, 1990).

with the elemental content of the canopy-layer foliage of three native tree species. The primary objective of this study was to determine if elemental concentrations of foliage differ between the western and eastern forest stands receiving different levels of atmospheric deposition loadings. A secondary objective was to establish a database which would allow interpretation of future trends in changing elemental concentrations in forest tree foliage subjected to anthropogenic inputs.

METHODS

Four ecologically analogous forest stands, one per core area (Fig. 1), were sampled in 1988 and 1989. For this paper, "stand number" and "core area number" are identical. Ten dominant or co-dominant northern red oaks, white oak, and red maple trees growing within each stand were randomly located and tagged. Ten trees per species were tagged in all areas except in the easternmost stand, where 10 red maples, 9 red oaks and 8 white oaks were located in 1988; 10, 10, and 7 trees of each species, respectively, were tagged in stand 4 in 1989. Ten trees per species were not always present because the number of oak trees in each stand was restricted due to the number of individual trees being utilized in previously established research plots. These restrictions resulted in a total of 117 trees sampled in the four areas in each of the two years. Two samplings were conducted per year . Due to personnel constraints and inclement weather, sampling could not be done on the same dates each year. In 1988 the trees were sampled during June 21-30 and August 30 - September 2. In 1989 a different set of trees were sampled during June 27 - July 10 and August 21-25. The location of each sample tree relative to other sample trees, as well as height, and diameter of each sample tree was recorded.

Four 1-m long branches (one branch per cardinal direction) were cut from the outer canopy of each tree by a climber positioned within the crown. On the ground, excised branches were placed in plastic bags for transport to the laboratory, where 30 leaves per branch were randomly selected and removed. Leaves were not washed in order to retain external contaminants entrapped as dry (or wet) deposition. Foliage from each tree was air dried, composited per tree, ground, and sent to Micro-Macro International (Athens, GA 30607) for determination of the macroelements N, P, K, Ca, Mg, and S, as well as microelements Al, B, Cu, Fe, Mn, Mo, Na, Pb, Si, Sr, V and Zn.. Total N was analyzed using Kjeldahl digestion with generated ammonium determined by automated colorimetry. Sulfur content was determined using a LECO (LECO Corp., St. Louis, MO) sulfur analyzer. Concentrations of remaining elements were determined using dry ashing of tissue with remaining ash dissolved in dilute aqua regia, and assayed by inductively coupled plasma (ICP) emission spectrometry.

The general approach to the statistical analysis was based on results of on-going, intensive atmospheric deposition monitoring and lichen studies. The historical deposition trends (Fig. 1) are based on regional network data from Pennsylvania and adjacent states. However, intensive monitoring conducted during this study at approximately 16-km intervals along the east-west gradient revealed that depositon values at study areas 1 and 2, characterized by greater pollutant deposition loadings, were similar and were different from those monitored at 3 and 4, which in turn were similar to each other (Lynch, 1990). In complementary studies, Showman and Long (1992) concluded that lichen richness values did not form a continuum along the deposition gradient, but rather separated into two distinct groups. The lichen communities growing in study areas 1 and 2 were significantly less rich compared to those communities in study areas 3 and 4, and had significantly greater concentrations of Al, Cr, Cu, and Fe in their thalli.

Therefore, we calculated a mean concentration for each element per tree species within study areas 1 and 2, as well as within study areas 3 and 4. Significant differences between the mean concentration per element for the two areas were then tested. A GLM (Ray, 1982) analysis of variance was conducted separately for each element, tree species, and year of sampling using a factorial design, with sampling "month" (two levels = June and August-September samplings) and location (two levels = "Low" and "High" deposition portions of the gradient) as factors. Mean separation tests (Scheffe) were conducted if main effects were significant and if significant month x location interactions did not occur. Results from the two sampling times per year were combined and presented as annual means. Significance tests were conducted at $p < 0.05$.

RESULTS AND DISCUSSION

Although considerable variation existed in the data, foliar Fe, S and Sr concentrations were often directly related to the atmospheric deposition pattern, especially in red oak and red maple (Table 1). These three elements likely are emitted from the numerous anthropogenic sources such as coal-fired power plants or heavy industries located upwind in the Ohio River Valley, other states west of Pennsylvania and/or in southwestern Pennsylvania. Atmospheric S from these sources may enter Pennsylvania as gaseous sulfur dioxide (Fig. 3), or in the case of long-range transport, as sulfate. Trace metals, especially heavy metals, are most commonly associated with fine particles in contaminated atmospheres (Smith, 1981). Prevailing southwesterly winds during the growing season carry these industrial pollutants to the northeast, until they reach the front of the Allegheny Plateau in western Pennsylvania. As the pollutant-laden air masses rise up the plateau, Fe, S, and Sr are transferred from the atmosphere to the forest ecosystem through absorption by vegetation and soils as wet or dry deposition, with the greatest pollutant loadings occurring in forest stands located in the western end of the gradient (Lynch, 1990).

The use of increased foliar sulfur as a marker for delineating general polluted areas supports previous findings in Pennsylvania (Hutnik and others, 1989), in southeastern Ontario (Linzon and others, 1979), and Minnesota (Krupa and others, 1980). However, these cited studies were all conducted near point sources such as coal-fired power plants or smelters. Shriner and others (1980) reported that 82% of the sulfate deposited in Tennessee forest ecosystems, distant from point sources, was dissolved in rainfall, whereas only 18% occurred as dry particulate fallout. Similarly, it is likely that the majority of S input into the Pennsylvania forests occurs as wet sulfate, with lesser amounts occurring as gaseous sulfur dioxide. Whatever the form of deposited S, the primary available form of S in the soil solution is the sulfate anion, most of which is found in the subsoil (Jones and others, 1991). In our study, the sulfate-S of the B1 and B2 soil horizons were fairly uniform across the gradient (Davis and others, 1990). This would indicate that the greater S content of foliage in the western end of the gradient may have been due to greater foliar adsorption or absorption of wet or dry deposition.

Heavy metals are deposited in rainfall (Lindberg, 1982) but were not measured by Lynch and co-workers in precipitation sampling conducted across the study area. Likewise, Fe and Sr content of soils along the gradient were not measured. The elevated concentrations of Fe and Sr in tree foliage in the western end of the gradient could have been deposited as wet or dry deposition. Since the leaves were not washed, the Fe and Sr could have existed as particulates on the external leaf surfaces. Alternatively, soluble forms of Fe and Sr could have been directly absorbed by foliage. If deposited onto the soil, metal activity and subsequent uptake by the plant, is affected strongly by soil pH, in addition to metal equilibria among clay mineral, organic matter, hydrous oxides of Fe, Mn, Al, and soluble chelators (Adriano, 1986). Iron and Sr are more readily available in the soil solution at an acidic pH (Friedland and others, 1984). However, since the soils across the gradient were rather uniform in acidity with no east-to-west pattern (Davis and others, 1990; McClenahen and Long 1993), the ions should have been relatively available at all four stands. Soil acidity probably did not play a major role in the observed patterns of Fe, S, and Sr accumulation.

Total (Kjeldahl) N concentrations in red maple (only) foliage were significantly greater in the high deposition part of the gradient (Table 1). It has been hypothesized that elevated nitrogen inputs could promote succulent twig growth and delay hardening-off in the fall, which might lead to twig and crown dieback, and ultimately to decline (Friedland and others, 1984). We did not observe elevated twig dieback in the high deposition end of the gradient, based on observations of tree crowns (Nash and others, 1992). Perhaps the increases were not great enough to induce succulent growth. However, more detailed studies related to N accumulation and twig dieback in red maple are warranted. Alao, accumulation of N in red maple foliage may be useful as a sensitive indicator of N deposition.

The general values for elements reported herein are within the range reported in comparable studies for foliage of red and white oak (Majumdar and others, 1989) and red maple trees (Lea and others, 1980) growing in other parts of the Northeast. Thus, although the concentrations in foliage were relatively elevated in the western end of the gradient, and/or were relatively less in the eastern part of the gradient, there is no reason to suspect toxicity nor deficiency symptoms to occur in either section of the gradient. Indeed, our field observations of crowns (Nash and others,

Table 1. Concentration of elements in foliage of three species collected from the High vs Low deposition portions of the Pennsylvania gradient.

	Quercus rubra				*Quercus alba*				*Acer rubrum*			
	1988		1989		1988		1989		1988		1989	
Element	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low
%												
N	2.23[a]	2.26	2.24	2.32	2.24	2.31	2.40	2.37	1.62	1.53[b]	1.66	1.54
P	0.16	0.16	0.16	0.16	0.16	0.17	0.17	0.18	0.15	0.14	0.14	0.14
K	0.85	0.84	0.75	0.72	0.77	0.82	0.71	0.71	0.71	0.75	0.62	0.55
Ca	0.39	0.44	0.46	0.43	0.43	0.55	0.41	0.53	0.37	0.37	0.37	0.34
Mg	0.08	0.09	0.11	0.10	0.10	0.10	0.11	0.09	0.09	0.09	0.09	0.08
S	0.21	0.19	0.22	0.19	0.22	0.21	0.25	0.20	0.15	0.14	0.14	0.13
ppm												
Al	52	51	44	48	42	43	42	49	34	26	24	19
B	-[c]	-	35	33	31	29	26	27	24	22	22	18
Cu	-	-	4.2	3.9	-	-	5.2	5.8	4.6	4.4	4.2	3.7
Fe	105	102	100	94	97	92	95	92	110	100	84	78
Mn	2396	2526	2240	2333	2264	2446	2184	2569	1690	1676	1565	1521
Mo	-	-	-	-	-	-	0.17[d]	0.20	-	-	0.16	0.15
Na	49	47	61	51	36	42	48	36	37	28	36	28
Pb	-	-	2.9	3.1	1.6	2.1	2.7	3.0	1.6	1.7	1.7	1.5
Si	145	132	111	112	238	259	222	252	357	358	339	278
Sr	5.1	3.8	7.6	5.4	7.0	7.1	7.9	7.8	7.6	5.9	11.7	7.2
V	0.36	0.43	0.95	0.96	0.54	0.74	0.87	0.89	0.61	0.64	0.56	0.49
Zn	18	21	27	25	13	14	18	17	17	20	23	18

[a] each mean is based on approximately 40 samples averaged across the two sampling months within a year.
[b] underline indicates a significant difference (p=.05) between mean values from "High" (core areas 1 and 2) and "Low" (core areas 3 and 4) deposition portions of the gradient.
[c] - indicates no data.
[d] significance not tested between these two means.

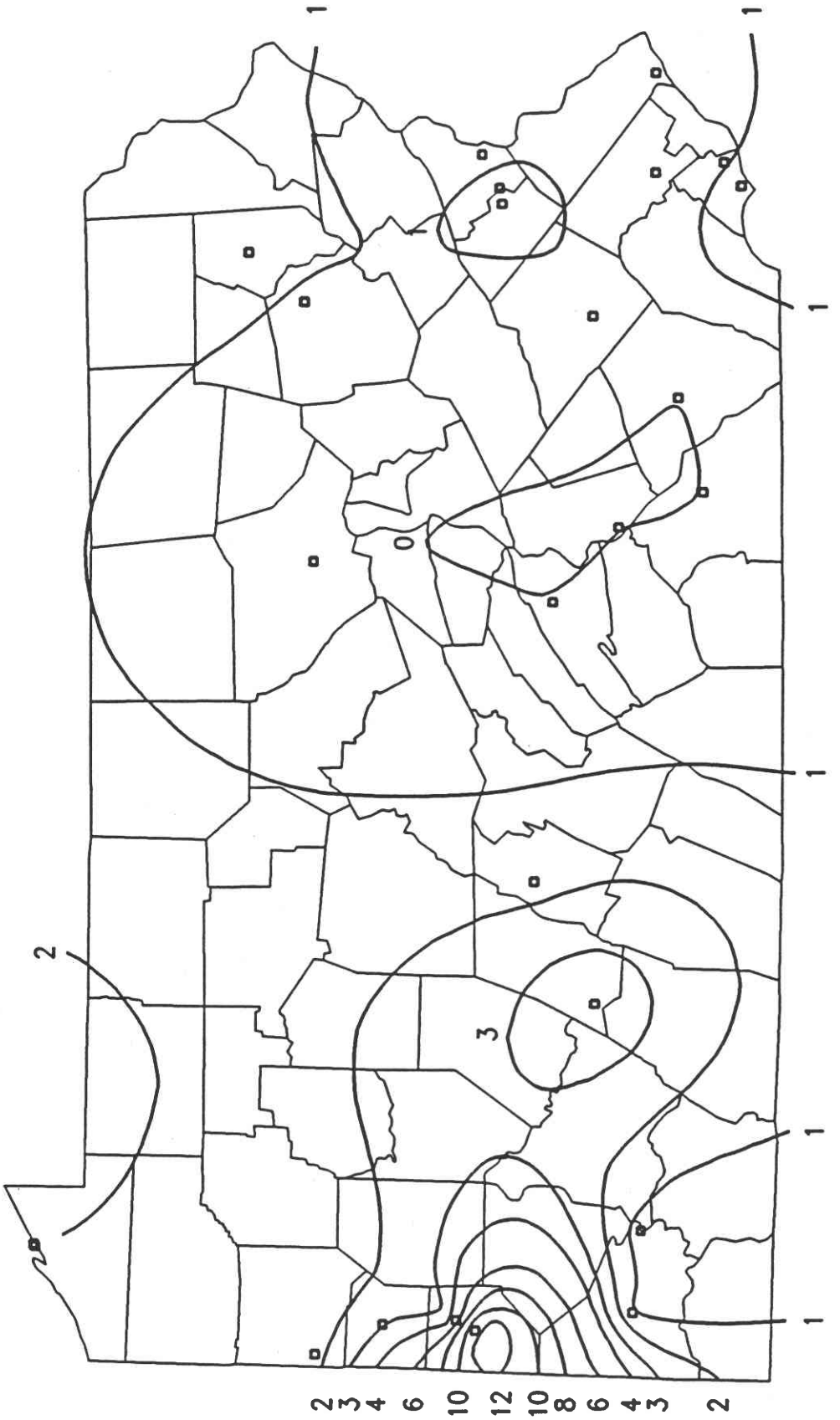

Figure 3. Percentage of hrs in 1988 that hourly-average SO2 concentrations exceeded 0.04 ppm in Pennsylvania (Data from PA Dept. Environ. Res., Bur. Air Qual. Map drawn by J. A. Lynch)

1992), as well as yet-unpublished laboratory evaluation of approximately 40,000 individual leaves indicated no apparent toxicity or deficiency symptoms.

In summary, our findings complement those of other researchers (Hutnik and others, 1989; Linzon and others, 1979; Krupa and others, 1980) who reported that S content of the foliage can be used to indicate areas affected by emissions from point sources such as coal-fired power plants and industries. Our findings extend the use of foliar S content as a marker of more widespread S or sulfate pollution. These results also complement the findings of others who reported that increased Sr content of white oak xylem (Long and Davis, 1989), sagebrush foliage (Connor and others, 1976) and grass (Wangen and Turner, 1980) as well as increased levels of Fe in grass, maple leaves, and pine needles (Klein and Russell, 1973) were increased immediately downwind from point sources such as coal-fired power plants. Since our unpublished data indicate that Fe, S, and Sr show an apparent increase during the growing season, late season collections should be utilized to ensure that concentrations of these three elements are great enough to delineate among areas of concern, and that levels are above the threshold limit for chemical analysis.

Thus, it appears that accumulations of Fe, S, and Sr may be indicative of ecosystems being subjected to atmospheric deposition originating as emissions from nearby point sources, or from pollutants which have undergone long range transport. Biological relationships between these long-term accumulations and forest health and productivity remain to be determined. In addition, the data set reported herein serves as a baseline from which future comparisons can be made as the Amended Clean Air Act goes into effect in the United States and industries reduce their emissions.

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LANDSCAPE-LEVEL REGENERATION ADEQUACY FOR NATIVE HARDWOOD FORESTS OF PENNSYLVANIA

William H. McWilliams, Todd W. Bowersox, David A. Gansner, Larry H. McCormick, and Susan L. Stout[1]

Abstract: Studies of advance regeneration and post-disturbance regeneration adequacy were conducted during the recent USDA Forest Service inventory of forest resources in Pennsylvania. The first study examined advance tree-seedling regeneration in stands where stocking levels would suggest that advance regeneration should be abundant. A range of metrics was used to describe regeneration adequacy. Findings indicate that advance regeneration is generally lacking in the State. The results ranged from only 4% to 40% of the sample locations being adequately stocked (based on the most and least stringent metric, respectively). Levels of fern and grass cover were sufficient to make treatment to control herbaceous vegetation an option at 54% of the sample locations. The second study focused on mixed-oak (*Quercus* spp.) stands that had undergone significant disturbance since the time of the previous inventory. Results indicate that 92% of the mixed-oak stands were adequately stocked with woody species. Stocking of oak species was far below what it was before disturbance. The least stringent metric of oak regeneration showed that only 16% of the sample locations were adequately stocked with oak species.

INTRODUCTION

At a time when many within the environmental community are concerned about regeneration of native hardwood forests, there is little information that describes regeneration adequacy at large geographic scales. Practicing foresters need information on regeneration adequacy to aid decisions about silvicultural investments and future markets for timber products. Studies of advance tree-seedling regeneration and post-disturbance regeneration were carried out as part of a recent inventory of forest conditions in Pennsylvania (Alerich 1993). The inventory was conducted by the USDA Forest Service, Forest Inventory and Analysis Unit of the Northeastern Forest Experiment Station (NE-FIA). The advance regeneration study evaluated tree-seedling abundance in a variety of forest types. The post-disturbance study focused on heavily disturbed mixed-oak stands.

METHODS

Advance Tree-Seedling and Herbaceous Cover Sample

Measurements of tree seedlings and herbaceous cover were adapted from the understory sampling procedure of Marquis et al. (1992) and added to the standard NE-FIA inventory procedures. The design for each sample location consisted of two nested circular plots--a 6-foot-radius plot and a 16-foot-radius plot located at each of five satellite points. On the 6-foot plot, tree seedlings (more than 2 in. in height and less than 1 in. in diameter) were tallied by species and height. On the 16-foot plot, percent cover was estimated for four classes of herbaceous vegetation: bracken fern (*Pteridium aquilinum* L.) , hayscented fern (*Dennstaedtia punctilobula* Michx.), and New York fern

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(*Thelypteris noveboracensis* L.); other fern; grass; and blackberry (*Rubus* spp.). Sample locations were screened for the range of stand-level stocking (or relative density) where advance tree-seedling regeneration should be abundant, that is, in stands with overstory stocking of 40- to 75-percent. Stands in this stocking range have a significant overstory, but available light is not likely to limit the establishment and growth of tree seedlings. The screening resulted in 499 sample locations. Six metrics were developed to represent a range of expectations about future conditions for seedling development. The metrics assess the adequacy of advance tree-seedling regeneration for three species groups: woody, commercial, and desirable; and two levels of stocking: high and low. The specifics of deriving the metrics are described in the Appendix. The low-density measures follow the guidelines of Sander et al. (1976) and Leak (1988). The more conservative high-density measures parallel the recommendations of Marquis and Bjorkbom (1982) and reflect regeneration needs under conditions that are unfavorable for seedling development, i.e., high deer (*Odocoileus virginianus* L.) impact.

Post-Disturbance Sample

When the statewide inventory was completed, information on disturbance, species composition, and relative density (Ernst, R.L. and W. Knapp 1985) were used to identify heavily disturbed mixed-oak sample locations. The decision to focus on mixed-oak stands was in response to concern over oak regeneration (Loftis and McGee 1993). Three criteria were used to select sample stands. First, candidate stands must have had at least 50% of total density in oak species at the time of previous measurement. Second, the stand must have been at least 60% stocked at the time of previous measurement (or above B-level stocking). Third, the stand must have been reduced to below 40% stocking at the most recent measurement (or below C-level stocking). Stands in this range are considered to be in need of regeneration (Ginrich 1967). These selection criteria yielded a total of 49 sample locations which were then revisited, and a grid of twenty 6-foot-radius circular plots was installed. At each plot, stem counts were conducted by species and height class. To reduce collection cost, stem counts were limited to the number of stems needed to satisfy the most conservative of two guidelines for determining if the plot was adequately stocked with regeneration. Various stand disturbances were encountered, but most involved heavy cutting and (or) overstory mortality resulting from gypsy moth (*Lymantria dispar* L.) defoliation, drought, or other factors. For most of the sampled stands, 10 to 15 years had elapsed since disturbance. Sample locations were located across the State with highest concentrations in southcentral counties where gypsy moth defoliation had a strong impact.

Measurements were taken to gauge whether these mixed-oak stands were adequately stocked with woody, commercial, or oak species following disturbance. A high-density metric was adapted from the work of Marquis and Bjorkbom (1982). A low-density metric of roughly half the high-density guideline was used to provide a range for analyzing the results. The number of stems required for adequate stocking are:

	Stocking Level	
	Low	High
Tree-Size	# of stems/plot	
5 inches dbh and larger	1	1
>= 5 feet tall and < 5 inches dbh	1	2
>= 3 feet tall and < 5 feet tall	3	5
>= 2 inches tall and < 3 feet tall	15	25

Sample locations were considered adequately stocked if at least 50% of the satellite plots satisfied a given metric. The 50% rule was used because most of the sampled stands had established forest cover of sufficient age to assume that stocking of satellite plots provides a good representation of stocking once the stand reaches merchantable size--the typical objective of post-disturbance regeneration inventories.

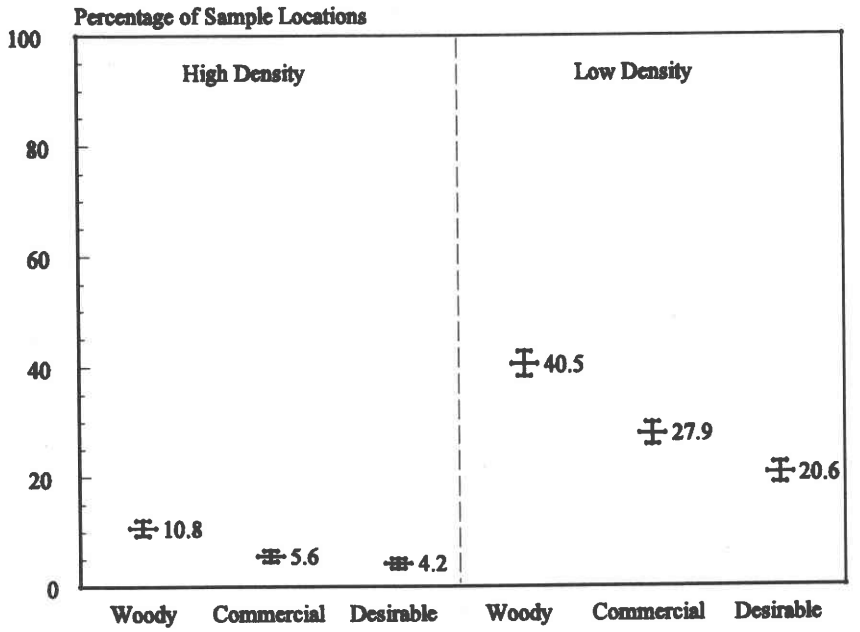

Figure 1. Mean percentage of sample locations adequately stocked with tree seedlings, by species group and density measure. (Brackets indicate the 95% confidence interval.)

RESULTS

Advance Tree-Seedling Regeneration and Herbaceous Cover

The data suggest that existing advance tree-seedling regeneration is an inadequate source for new stand establishment across most of Pennsylvania's forested landscape. Using the least stringent metric (a low density of woody stems) results in only 40% of the sample locations being adequately stocked (Fig. 1). When a high density of woody species is used, the estimate of adequacy is reduced to 11%. The woody-species measures assess the outlook for establishing woody cover following removal of the overstory without regard to how desirable the species may be for timber management. The success rates using the commercial-species measures of stocking adequacy are 28% for the low-density measure and 6% for the high-density measure. The commercial species group excludes species that contribute little to the stand's commodity value. The desirable group contains species most favored for timber production and, in general, offers more vigorous growth and a higher probability of developing a canopy of large well-formed stems that satisfy aesthetic objectives. The percentage of sample locations that were adequately stocked with desirable stems was 21 with the low-density measure and 4 with the high-density measure.

A search for statistically significant relationships between stocking adequacy and other variables collected by NE-FIA was conducted to find factors that might explain why advance tree-seedling regeneration so often is lacking in the forest understory. The variables of interest were: physiographic section (after Fenneman 1938), forest-type group, terrain position, aspect, stocking level, county-level deer density, and percentage of county-level land area occupied by forest. Continuous variables were grouped into discrete classes. Significance tests were conducted for all classes of variables and combinations of classes with at least 25 sample locations using SAS's categorical data modeling procedure (SAS Institute 1989). Of all categories tested, the only significant differences were between the oak-hickory and northern hardwoods forest-type groups. For all density measures, oak-hickory forests were associated with a higher abundance of advance tree-seedling regeneration than northern hardwood forests. This and the

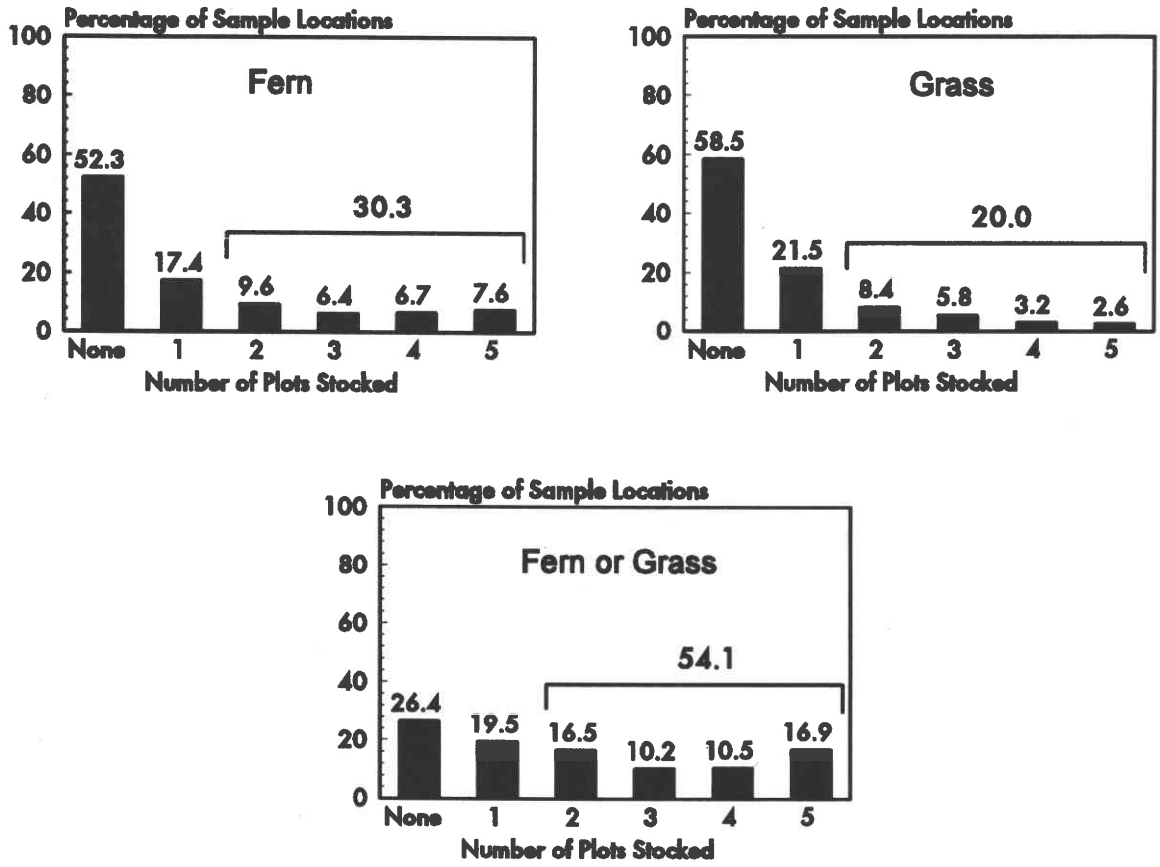

Figure 2. Mean percentage of sample locations and number of herbaceous cover plots stocked with at least 30% herbaceous plant cover, by type of vegetation.

finding that other variables and combinations were not significant suggest that poor advance regeneration is perhaps more wide ranging than was suspected.

The impact of herbaceous vegetation on regeneration ultimately depends on the abundance of advance regeneration because this determines the harvest options available (Marquis et al. 1992). In stands with sufficient regeneration for a final removal cut, the silvicultural guidelines recommend controlling herbaceous vegetation if at least 70% of the inventory plots are stocked with interfering herbaceous plants. (A herbaceous plot is considered stocked if 30% or more of the plot is covered.) If sufficient regeneration is lacking and a shelterwood seed cut is recommended, herbaceous control is considered if 30% of the plots are stocked. If the more conservative 30% threshold is used, over half of the sample locations would be potential candidates for herbaceous control due to the presence of fern, grass, or both in combination (Fig. 2). In the presence of fern alone, nearly one-third of the sample locations would qualify for treatment.

The sample also provides a general measure of the abundance of fern in Pennsylvania. Forty-eight percent of the sample locations had at least one herbaceous-cover plot stocked with 30% or more fern. The fern sample was collected for two species groups: species that have potential to spread through elongation of a perennial rhizome (bracken, hayscented, and New York fern) and other fern species. The former group is an aggressive invader of forest sites (Horsley 1988). Bracken, hayscented, and New York fern accounted for 70% of the total abundance of fern found in the sample.

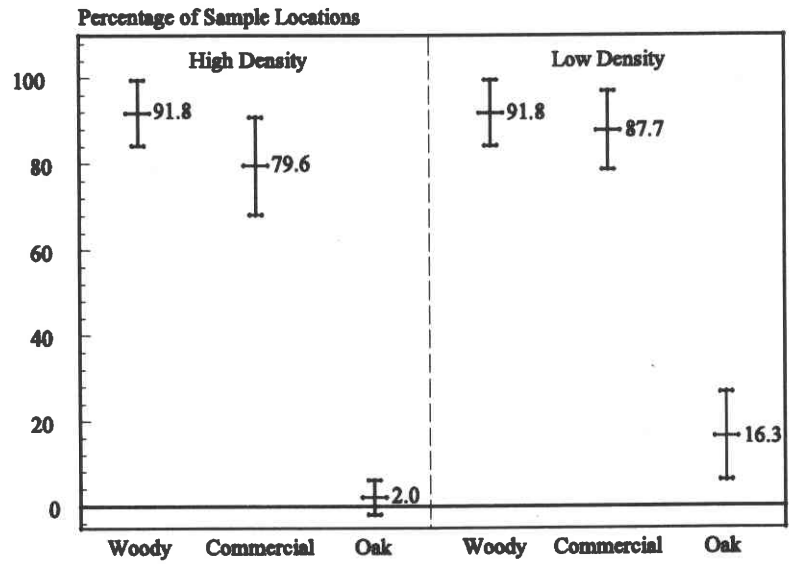

Figure 3. Mean percentage of sample locations adequately stocked with tree species following disturbance in oak-dominated stands by species group and density measure. (Brackets indicate the 95% confidence interval.)

Post-Disturbance Regeneration

The post-disturbance sample of mixed-oak stands was designed to answer two questions: 1)are stands regenerating with an adequate stocking of tree species and, 2)what is the status of oak species in the new stand? The results indicate that mixed-oak stands are regenerating adequately with tree species following heavy disturbance. Ninety-two percent of the sample locations were stocked with woody species using both the high- and low-density metrics (Fig. 3). For commercial species, 80% of the sample locations satisfied the high-density metric and 88% satisfied the low-density metric.

The findings show that the oak component is generally lacking following heavy disturbance in mixed-oak stands. When the high-density metric was used, oak regeneration was virtually non-existent. Only 2% of the sample locations were adequately stocked and this estimate was not significantly different from zero. When the low density metric was used, 16% of the sample locations were adequately stocked with oak. The low-density metric is probably a good indicator of adequacy for the oak component because of general stand conditions encountered on sample locations. Although evidence of deer was abundant on nearly all of the sample locations, most of the stands had developed to a point where sample trees were above 5 feet in height. Also, most of the oak regeneration encountered was found to be superior in terms of height, dominance, and vigor. In stands where oak regeneration was lacking, the most common species of the regeneration component were sweet birch (*Betula lenta* L.), red maple (*Acer rubrum* L.), and black cherry (*Prunus serotina* Ehrh.). As all of the stands in the study were at least 50% oak prior to disturbance, the results suggest a shift to forest types other than oak for most of the sample locations in the study. Further investigation of the data are planned to compare species composition before and after disturbance and the impact of disturbance on species diversity.

DISCUSSION

The results of these two studies have documented some significant regeneration challenges that exist across Pennsylvania's forested landscape. The advance regeneration study focused on stands where stocking levels would suggest that advance regeneration should be abundant. The most optimistic measure of advance regeneration adequacy was satisfied at only 40% of the sample locations. Fern and grass competition was significant at 54% of the sample locations. The impact of deer was not measured directly in the study, but Pennsylvania's deer herd has been a significant factor affecting understory conditions (Tilghman 1989). Other factors include soil and site characteristics, drought, stresses from diseases and insects, and other herbivores. The study of regeneration following heavy disturbance of mixed-oak stands found that regeneration was abundant, but species composition had changed due to poor regeneration of oak. The oak regeneration component was found to be inadequate in all but 16% of the sample locations. These findings are similar to those of Allen and Bowersox (1990).

It is interesting to compare the results of these two studies of regeneration adequacy even though they were conducted independently. The basic message of the advance tree-seedling study was that very few stands in Pennsylvania contained enough advance regeneration to adequately regenerate following harvest. The study of mixed-oak stands found that most stands had regenerated, but the composition tended to contain a new suite of species. The new stands typically contained light-seeded intolerant species such as black cherry and sweet birch. Black cherry and sweet birch are not preferred food sources for deer. The other prevalent species was red maple, which is very common throughout Pennsylvania and is a prolific producer of wind disseminated seed. Although not observed as part of this study, these invader species were likely not part of the advance seedling component of the sampled stands. This suggests that we need to know more about how well advance regeneration stocking guides predict future stand-level stocking and how forest composition changes as stands evolve over time.

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APPENDIX

Guidelines for determining the adequacy of advance tree-seedling regeneration.

Species Group

Desirable: Black cherry, oak, sugar maple (*Acer saccharum* Marsh.), red maple, conifer (*Juniperus virginiana* L., *Larix* spp., *Pinus* spp., *Tsuga canadensis* L.), hickory (*Carya* spp.), yellow-poplar (*Liriodendron tulipifera* L.), ash (*Fraxinus* spp.), basswood (*Tilia americana* L.), cucumbertree (*Liriodendron acuminata* L.), walnut (*Juglans nigra* L.), and butternut (*Juglans cinerea* L.).

Commercial: All desirable species and birch (*Betula* spp.), beech (*Fagus grandifolia* Ehrh.), blackgum (*Nyssa sylvatica* Marsh.), elm (*Ulmus* spp.), black locust (*Robinia pseudoacacia* L.), willow (*Salix* spp.), hackberry (*Celtis occidentalis* L.), and aspen (*Populus* spp.).

Woody: All desirable species, commercial species, and honey locust (*Gleditsia triacanthos* L.), sassafras (*Sassafras albidum* (Nutt.) Nees), ironwood (*Ostrya virginiana* (Mill.) K. Koch.), ailanthus (*Ailanthus altissima* (Mill.) Swingle), mountain ash (*Sorbus americana* Marsh.), blue beech (*Carpinus caroliniana* Walt.), hawthorn (*Crataegus* spp.), dogwood (*Cornus* spp.), redbud (*Cercis canadensis* L.), pin cherry (*Prunus pensylvanica* L.), striped maple (*Acer pensylvanicum* L.), hercules club (*Zanthoxylum clava-herculis* L.), scrub oak (*Q. ilicifolia* Wangenh.), chokecherry (*Prunus virginiana* L.), and shadbush (*Amelanchier arborea* (Michx f.) Fern.).

Satellite Plot Stocking

Seedling density	Minimum number of seedlings per plot
High	100
Low	25

To account for different seedling survival by height class, seedlings were weighted as follows:

Height	Weight
2 inches to 1 foot	1
1 to 3 feet	2
3 to 5 feet	20
5 feet and larger	50

Any combination of weighted stems that meets or exceeds the minimum number required is considered stocked. For example, a plot is considered to meet the high-density requirement for desirable species if it contains at least two stems at least 5 feet tall. Similarly, a plot is stocked with a low density of woody species if it contains 20 stems 6 inches tall and 3 stems 2 feet tall.

Sample Location Stocking

A sample location is considered stocked if at least four of the satellite plots (or at least 70%) contain the minimum number of seedlings.

LANDSCAPE VARIATION IN SPECIES DIVERSITY AND SUCCESSION AS RELATED TO TOPOGRAPHY, SOILS AND HUMAN DISTURBANCE[1]

Jeffrey N. Pearcy[2], David M. Hix[2] , and Stacy A. Drury[3]

Abstract: Three hundred and thirty-two plots have been sampled on the Wayne National Forest of southeastern Ohio, for the purpose of developing an ecological classification system (ECS). The ECS will be based on the herbaceous and woody vegetation, soils and topography of mature (80-140 year-old), relatively-undisturbed forests. Species diversity changes little across this landscape. Forty-eight woody tree species were identified among all plots and species richness (R) varied from two to thirteen on individual plots. Although variable within landform type, R varied similarly across landform type, with slightly greater variability on moderately steep slopes (3-13) than on steep slopes (3-10). Ridgetops (4-11) were comparable to sideslopes while ravines generally had fewer species (2-9) than other landforms. Again, although variable, R and species diversity (H) were not highly different across areas with differing soils characteristics. However, higher values of R and H were associated with severe, xeric sites having sandy soils; and mesic to dry-mesic sites that had evidence of nearby disturbance.

Species composition did vary predictably across the landscape. Northerly aspect and lower slope positions had more mesic communities than did southerly aspect and upper slope positions which were typically more xeric. Compositional stability of stands was determined by calculating a composition index (CI, after Fralish et al., 1993) for overstory, sub-canopy, sapling and seedling layers of woody vegetation. CI was based on the sum of the product between each species importance value (IV) and environmental adaptation value (AV). IV was the average of the relative density and the relative dominance of each species, and AV was compiled from several previously published studies (Buell et al., 1965; Curtis and McIntosh, 1951; Fralish et al, 1993; and Wells, 1976). Differences between strata of less than 100 units were considered to be fairly stable. Compositionally stable stands (based on four vegetational layers) were generally restricted to very mesic sites, xeric sites, and sites with unstable soil/slope conditions that prevent successional climax. Compositional stability on dry-mesic sites that are dominated by oak-hickory overstory is related to subcanopy and sapling layers that are predominately *Acer rubrum* (L.) and other non-oak species with AVs similar to oaks.

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CANOPY OPENINGS AND WHITE-TAILED DEER INFLUENCE THE UNDERSTORY VEGETATION IN MIXED OAK WOODLOTS[1]

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Abstract: Effects of canopy opening and white-tailed deer on ground level vegetation are being assessed in south-central Pennsylvania. Herbaceous plants and woody seedlings are being monitored in three unevenaged, mixed oak woodlots at Gettysburg National Military Park. Canopy opening levels on 0.20 ha treatment units were closed (~100% canopy), small (50-60% canopy) and large (0% canopy). Number of white-tailed deer during April have averaged about 0.40 deer per ha since 1992. Overstory treatments were conducted in March 1993. Fences to exclude deer were installed on one-half of the understory inventory plots. Before treatment inventories (1992) of the July herbaceous vegetation indicated 30% of the ground was covered by broadleafs, grasses and vines. Total herbaceous coverage increased to 54% of the area in the first growing season after treatment. Woodlot, canopy opening and fencing significantly influenced the coverage of herbaceous vegetation. Overall, total herbaceous coverage increased from 38% in the closed canopy treatment to 50% and 75% in the small and large canopy treatments, respectively. Compared to before treatment values, overall total herbaceous coverage in the first growing season after treatment increased 19% on all unfenced plots and 30% on all fenced plots. Before treatment inventories of the woody vegetation recorded an average of 5.6 shrub and 4.0 tree species seedlings per m^2. In the first growing season after treatment, number of shrub and tree seedlings increased to 6.7 and 8.6 stems per m^2, respectively. Ash dominated the tree species with an average of 2.0 seedlings per m^2 before treatment and 2.5 seedlings per m^2 after treatment. Average number of oak seedling in the first growing season after treatment (0.9 per m^2) was the same as before treatment. Woodlot was the only factor that consistently influenced the number of before and after treatment seedlings. Before treatment average numbers of shrub and tree seedlings were 12.3, 6.9 and 9.5 per m^2 for Bushman Hill, Herr Ridge and Powers Hill woodlots, respectively. In the first growing season after treatment, average number of shrub and tree seedlings for Bushman Hill, Herr Ridge and Powers Hill increased to 15.2, 7.8 and 13.1 per m^2, respectively. Overall, there were 2.1 shrub species and 3.7 tree species germinates in the first growing season after treatment. Most abundant germinates were ash ($0.6/m^2$), grape ($1.7/m^2$), redbud ($0.2/m^2$) and yellow-poplar ($2.9/m^2$). Canopy treatment and litter conditions, but not fencing, significantly influenced the abundance and species composition of the germinates.

INTRODUCTION

Gettysburg National Military Park (GNMP) was established in 1895 to commemorate the July 1863 Civil War battle. Originally established under the US War Department, GNMP was transferred to the National Park Service in 1933. GNMP presently covers about 1560 ha, with 643 ha occupied by woodlots. These woodlots range in size from 2 to 24 ha.

Before the battle, the woodlots were important cultural resources for the residents. Historic photographs and documents, and recent research reports (Fairweather and Cavanaugh 1990, Storm and others 1994) suggest a general pattern to the 1863 condition and subsequent development of these woodlots. Open-grown white oak (*Quercus*

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alba L.) trees were the dominant vegetation in 1863. In 1993, there were about 15 residual white oak trees per ha that were >135 years old. These trees typically had rapidly tapering boles with wide spreading crowns that had large diameter branches originating near the ground. Their diameters at 1.4 m above ground (DBH) generally were >60 cm. A second group of trees became established around 1900 (±25 years) when the understocked woodlots were retired from active agriculture. This group of trees was mainly northern red oak (*Quercus rubra* L.), black oak (*Quercus velutina* Lam.) white oak, bitternut hickory (*Carya cordiformis* (Wangenh.) K. Koch.), mockernut hickory (*Carya tomentosa* Nutt.) and shagbark hickory (*Carya ovata* (Mill.) K. Koch). These trees were generally between 30 and 60 cm in diameter in 1993, with closed stand architecture of slowly tapering boles with narrow crowns. A third group of trees developed in response to occasional openings in the canopy. These trees were mainly black cherry (*Prunus serotina* Ehrh.), flowering dogwood (*Cornus florida* L.), mazzard cherry (*Prunus avium* L.), white ash (*Fraxinus americana* L.) and hickories. In 1993, the third group of trees was generally between 3 and 30 cm in diameter.

Storm and others (1994) evaluated the overstory and understory of six GNMP woodlots. They concluded that the species composition, number and size of the trees >12 cm in diameter should be capable of maintaining an acceptable woodlot appearance for at least 25 years. They also concluded that there was a variety of tree species present in the seedling and sapling sizes (stems<12 cm in diameter). Overall there was an adequate number of seedlings (average of 23,584 per ha) but the number of saplings (826 per ha) was about one half the number needed to sustain the woodlots. Furthermore, oak species accounted for 44% of the number of trees in the overstory but only 6% of the sapling-sized stems.

White-tailed deer (*Odocoileus virginianus*) browsing and woodlot management practices may be influencing the height growth and species composition of the understory at GNMP (Bowersox and others 1993, Tzilkowski and others 1993). Average number of white-tailed deer on GNMP during April have increased from 0.25 deer per ha in 1987 to about 0.40 deer per ha since 1992. These deer densities far exceed the Pennsylvania Game Commission management goal of 0.08 deer per ha for the surrounding county. Shrub and tree seedlings recorded in 1987 on demonstration plots located in five GNMP woodlots averaged 7.1 per m^2 in the fenced plots and 7.3 per m^2 in the unfenced plots. By 1991, densities dropped to 6.1 in the fenced areas and 4.2 in the unfenced areas. Loss of seedlings between 1987 and 1991 was substantial for white oak, which dropped from 2.6 to 0.8 per m^2 in the fenced areas and from 3.7 to 0.5 per m^2 in the unfenced areas.

Maintaining these woodlot areas as tree-dominated communities is important to the park's mission to preserve the historic scene. Resource managers want to manage these woodlots to sustain an unevenaged structure. However, the current conditions have raised a concern for the future of these woodlots. They were created under frequent partial cutting practices before 1863 and a change in land use about 1900. There has been no tree cutting activity since 1895. The canopy trees are at or are approaching maturity. The existing woodlots were established at a time when there were no deer. Today's deer density is very high and may be influencing the understory vegetation. A research project has been developed and is being conducted to provide a basis for actively managing these woodlots. This project was designed to evaluate the effects of various size openings in the canopy, with and without white-tailed deer, on understory vegetation. This paper reports on the first growing season responses to these treatments.

PROCEDURES

The gently rolling topography of GNMP is in the Piedmont physiographic region (Lull 1968) of south-central Pennsylvania. Regeneration potential is being evaluated in three mixed oak woodlots. These woodlots are Bushman Hill (22 ha), Herr Ridge (17 ha) and Powers Hill (21 ha). Site quality was about average, and if occupied by evenaged stands the site index for oak would be ≥ 70. A more complete description of the site, soil and vegetation conditions was presented by Bowersox and others (1993).

Locations of the three replications of three canopy treatments in each of the woodlots were selected in April 1992 (Bowersox and others 1993). Felling of the canopy trees was conducted in March 1993. Directional felling was used

to uniformly distribute the residual over the 0.20 ha circular plot. Boles and branches were cut to place all material within 1 meter of the ground. No material was removed from the treatment units. Canopy treatments to the 0.20 ha areas were:

1. Closed. No changes in the overstory. Average stocking levels (based on standards developed for upland central hardwoods by Roach and Gingrich 1968) were 115, 100 and 99% for Bushman Hill, Herr Ridge and Powers Hill, respectively.

2. Small. Overstory canopies were reduced by single tree removals to achieve a modified structure (Bowersox and others 1993) goal at 50 to 60% of fully stocked level. Average stocking levels after tree felling were 53, 55 and 52% for Bushman Hill, Herr Ridge and Powers Hill, respectively.

3. Large. All stems >2 cm DBH were felled.

Wire fences 1.2 m high to exclude white-tailed deer were installed in August 1992. Estimates of the free-ranging white-tailed deer April densities for the unfenced areas have been based on actual counts since 1987 (Storm and others 1989, Storm and Tzilkowski 1992). These annual April counts are scheduled to be continued through 1997.

Abundance, composition and structure of understory vegetation were measured before and after the canopy treatments were executed. All understory inventory plots were randomly located in the central 314 m^2 of each 0.20 ha treatment unit to minimize the effect of adjacent community and site conditions. Three pairs of fenced and unfenced 2.0 m^2 circular plots were established in each 0.20 ha treatment unit to inventory the naturally established herbaceous and woody plant community. There were 27 pairs of fenced and unfenced natural regeneration plots per canopy treatment for all study woodlots. July coverage of broadleaf plants, grasses and vines on each plot was ocularly estimated to the nearest 5%. Similarly, coverage by individual herbaceous species was inventoried on a subset (n=9) of the plots. Total herbaceous was the summation of inventory categories, which due to layering, overlap and inclusions, may exceed 100%. Size and number of stems per woody species that were $\leq$ 1.50 cm in height were inventoried in August. Species were placed in Oak, Other Tree, Shrub or Total groups for some analyses. A list of species within each group is presented in Table 2.

A second set of plots was established to measure the abundance and species composition of new woody species germinates. To preserve known and unknown archeological resources in each woodlot, no disturbance to the soil was permitted when the canopy trees were felled. Seedbed conditions of no litter removed and litter removed were established to evaluate the effect of an undisturbed litter layer on the abundance and species composition of the new germinates. New woody species germinates were recorded in a set of two paired fenced and unfenced 4.0 m^2 circular plots per treatment unit. Each set of plots were randomly located within the central 314 m^2 of each treatment unit. One half of each plot was randomly assigned to each litter condition. Litter-removed treatments were conducted in May 1993 with all identifiable leaf and twig material manually removed to expose the humus or mineral soil surface. There were 18 pairs of fenced and unfenced germination plots (each with a 2.0 m^2 sub-plot for no litter removed and a 2.0 m^2 sub-plot for litter removed) per canopy treatment and a total of 54 paired plots for the study. Number of new germinates by species were recorded in August.

Analysis of variance was used to test for significant main and appropriate interaction factors on herbaceous coverage, number of seedlings and number of germinates. Mean separations were performed on those parameters which showed significant treatment effects with Tukey's method of Multiple Comparisons. Significance at the 0.05 level was used in all cases.

RESULTS

Before treatment inventories indicated 43 species of broadleaf plants, 6 species of grasses and 6 species of vines present in the three woodlots. In the first growing season after treatment, there were no overall changes in the presence of specific species on specific plots. Individual plots had an average of 8 species of broadleaf plants (1 to 14

species per plot), 1 species of grass (1 to 3 species per plot) and 3 species of vines (1 to 5 species per plot). Overall, herbaceous vegetation coverage averaged 2% for grasses, 6% for vines, 23% for broadleaf plants and 31% for total herbaceous. There was a significant difference in coverage of broadleaf plants and total herbaceous among the woodlots, and total herbaceous for canopy treatment. Bushman Hill and Powers Hill had significantly greater total herbaceous coverage (41%) than was measured at Herr Ridge (11%). Treatment units designated to receive small canopy and large canopy treatment had total herbaceous coverage of 33 and 37%, respectively. These values were significantly different from the total herbaceous coverage of 21% for those units designated for closed canopy treatment. There was no significant difference between the units designated for small and large canopy treatment. Otherwise, coverage values for all herbaceous vegetation groups were similar among the inventory plots. When averaged over woodlots, before treatment coverage values for non-fenced and fenced plots by canopy treatment ranged from 0 to 6% for grasses, 16 to 26% for broadleaf plants, 2 to 8% for vines and 18 to 37% for total herbaceous (Table 1).

Table 1. Average[1] coverage of herbaceous plant groupings by non-fenced and fenced plots, for before and after canopy treatments.

Canopy Treatment		Grasses Fenced		Broadleaf Plants Fenced		Vines Fenced		Total Fenced	
		No	Yes	No	Yes	No	Yes	No	Yes
		(----------%----------)							
Closed	Before	0	1	16	22	2	4	18	27
	After	1	2	21	27	12	14	34	42
Small	Before	1	2	24	26	5	8	30	36
	After	3	2	26	44	10	14	39	60
Large	Before	4	6	25	22	7	8	36	37
	After	11	7	31	55	23	22	65	84

[1] Based on 27 plots per canopy - fence condition.

As compared to before treatment values, canopy treatment resulted in significant increases in the first growing season coverage values for broadleaf plants, vines and total herbaceous. Overall, large canopy treatment had coverage increases of 19% for broadleaf plants, 15% for vines and 38% for total herbaceous. These increases were significantly greater than the 5 to 10% increases for broadleaf plants and vines in the closed and small canopy treatments. Overall, coverage of broadleaf plants and total herbaceous increases were significantly greater on the fenced plots than on the non-fenced plots. Broadleaf plant coverage increased by 5% on the non-fenced plots and 18% for the fenced plots. The increase in total herbaceous coverage was 18% for non-fenced plots and 29% for fenced plots.

Total herbaceous coverage in the first growing season after treatment was significantly different among the woodlots. Overall, total herbaceous coverage at Bushman Hill (58%) and Powers Hill (64%) were significantly different from the coverage at Herr Ridge (41%). Total herbaceous coverage at Bushman Hill and Powers Hill were not significantly different from each other. Coverage values for all groups of herbaceous vegetation were significantly different among canopy treatments. In all cases and when averaged over woodlots and fencing treatments, closed and small canopy treatment coverage values were not significantly different from each other but they were significantly different from the values for the large canopy treatment. Total herbaceous coverage averaged 38, 49 and 75% for closed, small and large canopy treatments, respectively. Fencing resulted in significantly different coverage values for broadleaf

plants and total herbaceous. Unfenced plot coverage averaged 26% for broadleaf plants and 42% for total herbaceous whereas fenced plot coverage averaged 42 and 62%, respectively. Averaged over woodlots, first growing season after treatment coverage values for non-fenced and fenced plots by canopy treatment ranged from 1 to 11% for grasses, 21 to 55% for broadleaf plants, 10 to 23% for vines and 34 to 84% for total herbaceous (Table 1).

Total number of before treatment shrubs and tree species of stems >2 cm DBH was 24 for all woodlots. Individual woodlots ranged from 17 to 24 species. On the individual treatment plots, there was an average of 4.8 species in the before treatment inventories. There was an average of 5.6 shrub seedlings per m^2 and 4.0 tree seedlings per m^2 for the three woodlots (Table 2). *Rubus* spp. with 2.3 stems per m^2 and spicebush (*Lindera benzoin* L.) with 1.6 seedlings per m^2 accounted for a majority of the shrub seedlings. Ash dominated the tree species with an average of 2.0 seedlings per m^2. There was an average of 0.9 Oak seedlings per m^2. The seedlings were dominated by stems in the $\leq$ 25 cm height class. Oak, Other Tree and Shrub species groups had 2, 9 and 20%, respectively, of the Total in the >25 cm height classes. There were no significant differences in the number of Oak, Other Tree, Shrub and Total seedlings between the plots assigned to be fenced and those to be unfenced. Overall, average Total seedlings per m^2 were significantly different for Bushman Hill (12.3), Herr Ridge (6.9) and Powers Hill (9.5) (Table 2). The average number of Oak seedlings per m^2 at Bushman Hill of 2.1 was significantly different from the 0.2 and 0.3 per m^2 seedlings at Herr Ridge and Powers Hill, respectively.

Table 2. Average[1] before and after treatment number of seedlings per m^2 for the three woodlots, by species group[2].

Species-Time		Bushman Hill	Herr Ridge	Power Hill	Overall
		(----------Number of Stems per Square Meter----------)			
Oak	Before	2.1a	0.2b	0.3b	0.9
	After	2.1a	.2b	.3b	.9
Other Tree	Before	4.4a	1.0b	3.9a	3.1
	After	6.4a	1.2b	6.0a	4.6
Shrub	Before	5.8a	5.6a	5.3a	5.6
	After	6.6a	6.3a	6.9a	6.6
Total	Before	12.3a	6.9c	9.5b	9.6
	After	15.2a	7.8b	13.1a	12.1

[1] Based on 54 plots per woodlot - time of inventory. Woodlot means within species group, by time, with the same letter were not significantly different at the 5% level.

[2] Oak included black, chestnut, northern red, scarlet and white.

Other Tree included American elm, bitternut hickory, black cherry, black gum, black walnut, eastern red cedar, hackberry, mazzard cherry, mockernut hickory, Norway maple, pignut hickory, red maple, shagbark hickory, sugar maple, white ash and yellow-poplar.

Shrub included blackhaw, euonymous, flowering dogwood, grape spp., Japanese barberry, redbud, rose spp., rubus spp., sassafras, serviceberry, spicebush and witch-hazel.

Over all woodlots, there was an increase in the average number of species and seedlings per plot in the first growing season after overstory treatment. There was an after treatment average of 5.4 species per plot, and 6.6 shrub species seedlings and 8.6 tree species seedlings per m^2. For the Shrub group, spicebush and *Rubus* spp. continued to have the most abundant seedlings with 1.8 and 2.6 stems per m^2, respectively. The Other Tree group continued to be dominated by ash which increased to 2.5 stems per m^2 but there was a notable gain with yellow-poplar (*Liriodendron tulipifera* L.). Yellow-poplar increased from 0 seedlings before treatment to 0.9 seedlings per m^2 after treatment. There was no change in the average number of Oak seedlings. There was an increase in the relative number of seedlings in the >25 cm height classes but none of the increases were significantly related to any of the treatments. Oak, Other Tree and Shrub groups had 3, 14 and 34% of the Total seedlings in the >25 cm height classes.

As with before overstory treatment values, woodlot continued to be a significant factor in the number of seedlings present after treatment in the Oak, Other Tree and Total groups. Average woody species seedlings per m^2 of 7.8, 13.1 and 15.2 for Herr Ridge, Powers Hill and Bushman Hill, respectively, were significantly different from each other (Table 2). Average Oak seedlings per m^2 at Bushman Hill (2.1) were significantly different from those at Herr Ridge (0.2) and Powers Hill (0.3). Average Other Tree seedlings per m^2 at Bushman Hill (6.4) and Powers Hill (6.0) were significantly different from those at Herr Ridge (1.2). There were no significant differences in the number of Oak seedlings at Herr Ridge and Powers Hill and the number of Other Tree group seedlings at Bushman Hill and Powers Hill. The only other significant treatment effect was with canopy opening for Other Tree. Average number of Other Tree seedlings per m^2 in the closed canopy (1.8), small canopy (2.7) and large canopy (2.3) treatments were significantly different from each other. This response was mainly due to the number of ash seedlings.

There was an average of 2.1 shrub germinates and 3.7 tree germinates per m^2 in the first growing season after treatment. There were seven shrub species that produced a least one germinate. Redbud (*Cercis canadensis* L.) and grape (*Vitis* spp.) had the most abundant shrub species germinates with 0.2 and 1.7 per m^2, respectively. Overall, 11 tree species produced at least one germinate. Ash and yellow-poplar had the most abundant tree species germinates with an average of 0.6 and 2.9 per m^2, respectively. Number of ash, grape, redbud, yellow-poplar, Total Tree, Total Shrub and Total germinates were significantly affected by woodlot, canopy treatment and litter treatment but not by fencing. Except for ash and yellow-poplar, all of these species groups also had significant canopy x litter interaction effects.

Ash was the most abundant before-treatment tree species in the understory of the woodlots. The number of germinates was greatest for the intact litter condition and for the small canopy opening condition (Table 3). In contrast, yellow-poplar seedlings were not present in the before treatment woodlot conditions. However, as the size of canopy opening increased and particularly when the litter was removed, the number of yellow-poplar germinates increased substantially (Table 3). Number of germinates for redbud and grape were differentially influenced by canopy and litter treatment. Litter retained or removed had little effect on germination when the canopy was closed (Table 3). However when openings were made in the canopy, the number of germinates was greater when the litter was removed.

SUMMARY AND CONCLUSIONS

Canopy opening and fencing to exclude deer treatments have been applied in previously unmanaged, unevenaged mixed oak woodlots at GNMP. Abundance, composition and structure of understory herbaceous and woody vegetation was inventoried before treatment and in the first growing season after treatment. Herbaceous vegetation in the first growing season after treatment was responsive to size of opening in the canopy and fencing. Broadleaf plants and vines were more responsive to treatments than were the grasses. Broadleaf plants and vines also had substantial after treatment increases in coverage with increased canopy removal. In addition, broadleaf plants had greater coverage where white-tailed deer were excluded.

Richness of shrub and tree species in the before and after treatment inventories indicated that the conditions were favorable for establishing natural regeneration. In addition, canopy opening and litter conditions could be adjusted to

Table 3. Average[1] number of germinates in the first growing season after treatment for ash, yellow-poplar, grape, redbud, all tree, all shrub and total, by canopy opening and litter treatment.

Species	Closed Canopy Litter Removed No	Closed Canopy Litter Removed Yes	Small Canopy Litter Removed No	Small Canopy Litter Removed Yes	Large Canopy Litter Removed No	Large Canopy Litter Removed Yes
	(----------Number of Germinates per Square Meter----------)					
Ash	0.4	0.3	1.3	0.8	0.7	0.3
Yellow-poplar	.3	.3	1.5	5.0	3.2	7.2
Grape	.1	.4	.4	4.4	.8	3.8
Redbud	.1	.0	.2	.4	.1	.6
Total Tree	.8	.6	3.1	5.8	4.1	7.6
Total Shrub	.6	.4	.9	5.0	1.1	4.6
Total	1.4	1.0	3.9	10.8	5.3	12.2

[1] Based on 18 plots per canopy - litter condition.

influence the abundance and species mixture of new germinates from natural seed sources. However, the initial size and apparent height growth of the seedlings were of major concern. There was some increase in the relative number of seedlings >25 cm tall in the first growing season after treatment but the increase was small even where the seedlings were protected from white-tailed deer. Before treatment overstory interference on the seedlings' opportunity to grow, and frequent foraging by white-tailed deer may have resulted in seedlings with low energy reserves and poor vigor. The slow height growth of the tree seedlings combined with the rapid response in coverage by the herbaceous vegetation raises a concern about the potential application of large canopy openings to regenerate these woodlots.

Abundance, composition and structure of herbaceous vegetation, shrub and tree seedlings in the second growing season after treatment have been inventoried in 1994. Additional germinates in the second growing season have been recorded. These data and additional data to be collected in the fourth growing season after treatment (1996) will be used to develop management recommendations for the unevenaged mixed oak woodlots at GNMP.

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HISTORY OF DEER POPULATION TRENDS AND FOREST CUTTING ON THE ALLEGHENY NATIONAL FOREST

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Abstract: The forests of the Allegheny Plateau section of northwestern Pennsylvania have been severely impacted for more than 70 years by selective browsing by white-tailed deer (*Odocoileus virginianus*). Historical and ecological interactions of deer and the forest ecosystem in this region from pre-settlement times to the present are presented based on data from the Allegheny National Forest area. The data suggest that deer impacts on forested ecosystems can be controlled through a combination of increased, sustained deer harvests and increased forage production through timber harvesting.

INTRODUCTION

As the second-growth forests on the Allegheny Plateau began to mature in the mid-1960's, foresters attempted to regenerate new stands by clearcutting, anticipating that species composition of the new stands would be similar to the second-growth forest that originated near the turn of the century. However, these harvest cuts resulted in a complete failure of woody species to regenerate or in understocked stands that lacked the hardwood species diversity of the second-growth forest.

Concern about the inability to obtain regeneration success led to the establishment of a regeneration research project conducted at the Northeastern Forest Experiment Station's Forestry Sciences Laboratory at Warren, Pennsylvania. Grisez and Peace (1973) showed that nearly half of all regeneration cuts in the 1960's failed to regenerate to woody species. The half that did regenerate successfully had well-established advance regeneration prior to cutting. Deer browsing was suspected as a primary cause of most regeneration failures. A series of deer exclosure studies was installed to test this hypothesis. Exclosures placed in recently failed clearcuts demonstrated that in 87 percent of the stands sampled, regeneration was satisfactory inside the fences but not outside (Marquis and Brenneman 1981). This finding supported the hypothesis that deer browsing was a direct factor in causing regeneration failures in the 1960's. But there were other indirect effects of deer browsing and other ecological factors that changed the entire dynamics of the regeneration process.

Four major components of the ecosystem had changed dramatically since the turn of the century, leading to the complex situation for regeneration that exists today. First, deer were nearly extinct in Pennsylvania at the turn of the century but at record levels in the 1960's (Horsley 1992). Second, species composition of the overstory and understory was different in second-growth forests than in the original forest. Much of the original forest contained overstories of large white pine (*Pinus strobus*), hemlock (*Tsuga canadensis*), sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), and chestnut (*Castanea dentata*), and small quantities of black cherry (*Prunus serotina*), white ash (*Fraxinus americana*), oaks (*Quercus* spp.), and red maple (*Acer rubrum*) (Marquis 1975; Whitney 1990). The understories in the original forest contained many well-established advanced seedlings which were absent in the 1960's (Marquis 1973). The second-growth forest contained higher proportions of shade-intolerant species such as black cherry, white ash, and yellow-poplar (*Liriodendron tulipifera*) in the main crown canopy, with very shade-tolerant species such as American beech and sugar maple relegated to the lower canopy unless left as residual trees from the original forest (Marquis 1981). Species of intermediate tolerance such as red maple, basswood (*Tilia americana*), and cucumber (*Magnolia acuminata*) were in a third layer just below the main crown level.

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Third, the second-growth forest contained a species-altered and less diverse understory of shrubs and herbaceous plants because of decades of preferential browsing by an exceptionally large deer herd (Leopold 1943; Whitney 1984). Many of the shrub species that were abundant in the original forest, for example, greenbrier (*Smilax rotundifolia*), hobblebush (*Viburnum alnifolium*), wild raisin (*Viburnum cassinoides*), and other viburnums, were nearly lacking from the second-growth forest (Leopold 1943; Whitney 1984). Instead, the understory is dominated by species such as striped maple (*Acer pennsylvanicum*), root suckers of American beech, or by fern and grass. Fourth, seedbed conditions in the 1960's were different due to long periods without fire, years of heavy browsing, and a lack of dead and down course woody debris that had been present at the turn of the century (Mladenoff and Sterns 1993).

A look at long-term trends in deer densities and forage production suggests a strong relationship between the two (Whitney 1990; Marquis 1975). In this paper I have made the assumption that timber harvest can be used as a surrogate for forage production. This assumption is made with the knowledge that not all silvicultural treatments create the same quantity and quality of forage, and that forage production can vary within treatments. With these assumptions in mind, I present an historical overview of events from pre-settlement times to the present that shows how deer populations and forage availability have combined to create a deer impact on all forest communities on the Allegheny Plateau in Northwestern Pennsylvania.

Actual data for estimates of the deer population taken in Warren, Forest, McKean, and Elk Counties, that encompass the Allegheny National Forest, are presented. These data were used rather than statewide averages because actual timber harvesting figures are available for the Allegheny since 1923. Comparing deer population and forage production levels reveal some important relationships that are not apparent when looking at the two separately.

The 4-county area encompassing the Allegheny National Forest is a part of the larger Allegheny Plateau region. Elevations on the Plateau range from about 1,500 to 2,500 feet (Hough 1959). The climate is generally cool and moist with precipitation well distributed throughout the summer months (Whitney 1990). Nearly all of the soils in the area are podzols derived from acid sandstone and shale. The forests of the area are primarily second-growth cherry-maple mixed hardwood forests and are listed as SAF type 28 (Marquis 1975).

Human influence on ecological processes in the Allegheny region has been divided into 6 distinct periods[2]. I add a 7th period to reflect changes in forest management from 1980 to the present. Each period is discussed separately and summarized to show trends.

ECOLOGICAL PERIODS INFLUENCED BY HUMAN ACTIVITY

Pre-settlement Period: Pre-1800

Native Americans occupied northwestern Pennsylvania for many centuries prior to the arrival of the European settlers. The Allegheny River, Clarion River, and Conewango Creek served as major travelways and the rich river valleys provided sites for major Indian villages.

Vegetation conditions. Pre-settlement forests were a mosaic of forest stands in various stages of successional development. The largest share of the forest was mature or overmature hemlock-beech and beech-sugar maple (Illick 1923; Marquis 1975). There were scattered meadows and rough mountaintops void of forest cover but also vast stretches of unbroken timber (Whitney 1990). Areas of early successional ecosystems were created by natural disturbances such as hurricanes and tornadoes. A large area of blowdown was reported in what is now Alleghany State Park, just north of the present Allegheny National Forest, by a surveyor for the Holland Land Company in June 1805 (Gordon 1940). Understory conditions probably were variable due to human activity and periodic windstorms.

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[2]Bennett, A. 1969. History of Allegheny Plateau Forest. Unpublished talk at Regeneration Conference on Allegheny Plateau Hardwoods, Bradford, PA.

Native Americans periodically burned forest understories near their villages to increase berry production and enhance hunting opportunities (Marquis 1973; McCabe and McCabe 1984). Shrub species such as the viburnums likely were present during pre-settlement times as they were noted in early surveys of the area in the late 1700's.

Forage availability. The amount and quality of forage used by white-tailed deer were highly variable, spatially and temporally, due to periodicity of windstorms and the abandonment of Indian villages as resources were depleted (Marquis 1973). Agricultural crops grown in or near these villages provided quality forage as did early successional ecosystems, for example, areas burned by Native Americans and those created by tornadoes.

Predation. Predators of deer included wolves (*Canis lupus*), mountain lions (*Felis concolor*), bobcats (*Felis rufus*), and bears (*Ursus americanus*). It is believed that these predators accounted for half of the annual deer mortality (McCabe and McCabe 1984). The remaining mortality was caused by hunting by Native Americans, disease, starvation and weather events.

Weather. Due to cyclic patterns of weather and the length of the pre-settlement period, the entire range of weather conditions was experienced.

Deer densities. Estimates cited from the literature of deer densities during pre-settlement times are for the entire range of white-tailed deer, which extends beyond northwestern Pennsylvania. Findings from numerous archeological middens show that deer were an important source of protein in the diets of Native Americans (McCabe and McCabe 1984). Using population models developed by Severinghaus and Maguire, estimates of deer densities were made using sex ratios and age-class structures of the population harvested by Native Americans (Severinghaus and Maguire 1955). These estimates in the range of 20 deer/square mile for the region, agrees with those of Seton based on suitable habitat within the range of eastern white-tailed deer(Seton 1909).

It is likely that deer populations in the 4-county area of this report differed from place to place and year to year. Populations likely were higher near Native American villages where agricultural crops were available and firewood cuttings and burned areas produced browse. Also, populations probably were higher in and around areas of early successional forest created by periodic weather events. Most of the four-county area was inaccessible and heavily forested resulting in low deer browse production during this period so the best estimate of average deer density is 10 to 15 per square mile.

Pioneer Period: 1800-1860

The first record of Europeans in northwestern Pennsylvania was an English trading post established in 1750 at Buckaloons along the Allegheny River near Warren, Pennsylvania[3]. The first permanent European settlers traveled from Susquehanna County to Warren County in 1795. Settlement of the area progressed slowly at first but then increased rapidly in the latter part of the 19th century. Warren County, for example, had a population of only 26 in 1810 (Marquis 1973). The last half of this period saw rapid growth in human populations.

Vegetation conditions. Overall, vegetation changed little from pre-settlement times. The impact of humans on vegetation was restricted to the major drainages as most of the Plateau remained much as it was for centuries prior to settlement. A slight increase in agricultural land and timber harvesting resulted in modest increases in early successional vegetation along the drainages.

Forage availability. During the first 20 years of this period, little timber cutting took place. The few European settlers there were cut small quantities of wood for cabins, barns, and firewood, and cleared land for gardens. Forage availability during this time likely was similar to that of pre-settlement times.

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[3]Stotz, L. 1973. A country supposed uninhabitable, Senecas and Settlers on the Allegheny National Forest. In: Allegheny Nat. For. Spec. 50th Anniversary Pub. Article #3. Schuler, N. (ed.)

After 1820, water-powered mills were common and there was some commercial timber cutting along the major drainages where logs could be floated easily to the mill. During the last half of this period, forage production increased as timber cutting accelerated to meet the demands of an expanding population.

Predation. There was a gradual, continuing decline of major deer predators during this period, culminating in the near extirpation of some natural species toward the latter portion. Predators such as the gray wolf, wolverine, mountain lion, and lynx were not extirpated, but their numbers declined steadily as human populations increased (Merritt 1987). Deer populations also were reduced by Native Americans and European settlers.

Weather. The only major documented event was a large tornado that occurred in what is now the Tionesta Scenic and Research Natural Area in 1808 (Bjorkbom and Larson 1977). Winters likely were much as they are today.

Deer Density. During the first 20 years of this period, deer densities likely were similar to those of pre-settlement times as human populations increased little and timber harvesting remained about the same and were restricted to lands along the major drainages. As the period progressed, human populations and timber harvesting increased while natural predators decreased, resulting in a probable increase in deer populations. Speculative estimates of deer densities for this period are 10 to 15 per square mile early in the period and 25 to 35 per square mile toward the end of the period.

Steampower Period: 1860-1890

The 1860's saw the beginning of industrialization of the Allegheny Plateau. Steam replaced water as the major source of power. Steam-powered sawmills, railroads, log loaders, and other logging equipment became common during this period (Marquis 1975). This equipment was capable of processing huge quantities of timber. Tanneries were common during this time and used large quantities of hemlock bark for tannin.

Vegetation conditions. Major changes in vegetation began to take place. White pine was the major species being harvested by 1875,and was almost gone by 1892. Pine and hemlock were replaced with shade-intolerant, early successional hardwood species. Once the pine was depleted, hemlock became the major species harvested, for its bark for the tanning industry and for sawlogs cut into construction lumber. Most of the pine and hemlock harvests was small patchy cuts that were heaviest along the major drainages. Most of the forest remained intact.

Forage availability. Forage production resulting from the harvest cuts along the major drainages increased greatly. Encroachment by white settlers increased with better access to the region afforded by railroads, and more land was cleared for agriculture. But forage on the plateau top remained essentially unchanged.

Predation. By the end of this period, all of the major natural predators were extirpated. Wolverines, mountain lions, and lynx were gone by the late 1800's (Merritt 1987), and the last gray wolf was reported in 1892 in Clearfield County, southeast of the Allegheny region (Merritt 1987). Humans continued to play a large role in the annual mortality as there were no laws to protect deer..

Weather. There are no detailed records for this period. A large tornado was recorded in 1870 in the Tionesta Scenic and Research Natural Area (Bjorkbom and Larson 1977).

Deer Density. Although forage was increasing, especially near the major drainages, so too was human harvest of the deer herd as year-round hunting prevented the herd from becoming excessive. Knowing what we know about forage availability and the effects of hunting, I estimate the deer population was 20 to 25 deer/square mile during this period.

Chemical Wood Clearcutting Period: 1890-1930

Until this time, steam locomotives were used primarily for transportation. During this period specially designed narrow gauge locomotives were developed to traverse the most rugged terrain, and were located in nearly every

drainage making nearly all of the Plateau accessible. Chemical wood plants utilized wood of all species, quality, and size to produce charcoal, acetic acid, wood alcohol, and other distillation products (Marquis 1975). The combination of markets utilizing trees of all species and quality, and the heightened access afforded by logging railroads resulted in an extremely high degree of utilization of the wood resource. Most of the area was clearcut down to stems 1 or 2 inches in diameter, resulting in a species conversion from true northern hardwoods to the second-growth cherry-maple forests that cover the area today. Along major drainages, stands of pine, chestnut, and oak, were converted to chestnut-oak-red maple.

Vegetation conditions. Nearly the entire area was in early successional stage vegetation consisting of a mixture of species of various tolerance to shade. Many shrub species such as hobblebush were abundant. By the late 1920's, about half of the cut areas had grown into sapling or small poletimber trees.

Forage availability. Vast quantities of forage were available to deer throughout this period. The midpoint of the period provided the most forage. By the late 1920's, many stands were growing out of the reach of deer.

Predation. All natural predators were absent during this period. From 1890 to 1900, deer were hunted for venison and hides and to supply meat for logging camps. By 1900, deer were all but extirpated from the entire state. The Pennsylvania Game Commission was established in 1896 primarily to establish laws to protect what few deer remained (Winecoff 1930). Salt licks and the use of dogs for hunting deer were prohibited after 1898. The first game refuge, known as State Gamelands today, was established in 1905. A "bucks only" law was enacted in 1907 and remained in effect for the next 16 years. A hunting license law was enacted in 1913 to raise money to restock deer from other states. During the period, 700 deer were purchased and restocked in Pennsylvania.

Weather. No unusual or extreme weather events are recorded for this period.

Deer density. Deer densities went from near extinction at the turn of the century to near record highs at the end of this period. Laws to protect deer, a lack of natural predators, and vast forage production resulting from chemical wood cuts led to an irruption of the deer population. Densities increased from nearly zero to more than 40 deer/square mile by the end of this period.

Figure 1 shows the pattern of timber harvest and deer density on the Allegheny National Forest from 1900 to 1992. Forage production was greatest during the chemical wood harvest and sawtimber-modern eras. Circles mark actual data points listed in Table 1, and are connected by arbitrary straight lines. Timber harvest data are actual annual figures for the Allegheny National Forest for 1941-92. The crashes in deer densities in the early 1940's and again in the late 1970's occurred after successive severe winters and a sharp decrease in forage production (timber cut).

Second-Growth Poletimber Period: 1930-1950

By the early 1930's, trees on the chemical wood clearcuts had grown into the large sapling-small pole size classes. Much of the land in the 4-county area was under protection and management by the USDA Forest Service. Although the Allegheny National Forest was established in 1923, little management other than fire protection took place until the 1950's.

Vegetation conditions. Much of the shrub layer of vegetation began to decline in number during the early part of this period and many species were absent by 1950. Fern, grass, striped maple seedlings, and small root suckers of American beech made up a large portion of the understory toward the end of this period. The overstory consisted of dense sapling and poletimber stands composed of shade-intolerant species such as black cherry, yellow-poplar, and white ash, with red maple poles growing in the intermediate canopy. Sugar maple and American beech saplings were found beneath the main canopy. By 1942, nearly all of the hemlock advance regeneration had been eliminated from the Tionesta Natural Area by deer browsing (Hough 1965). Most of the hemlock in the second-growth forest was left as residual trees from the original forest. Stems of sugar maple and American beech in the same crown layer as the shade intolerants also were residuals from the original forest and represented an older age class.

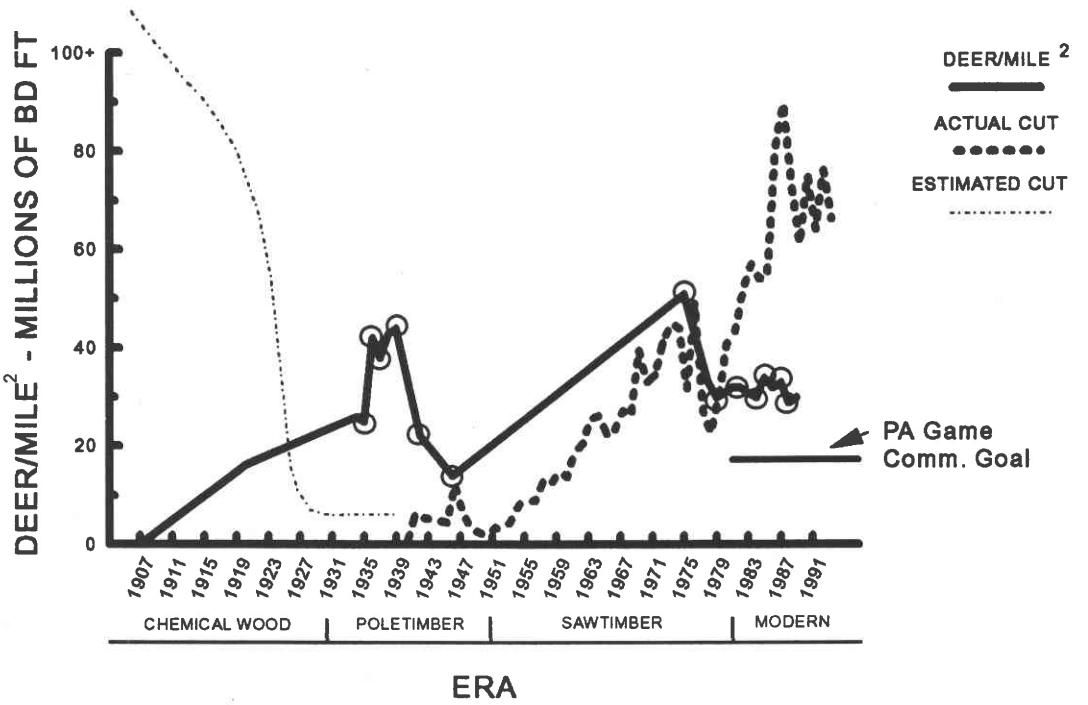

Figure 1. Deer population and timber cutting trends on the Allegheny National Forest.

Table 1. Data sources reported in Figure 1.

Source	Date	Deer/square mile
McCain	1935	23
McCain	1936	27
McCain	1937	34
McCain	1938	23
Hough	1936	29
Hough	1938	42
Hough	1946	14
Jordan	1974	15[a]
Jordan	1975	43
Jordan	1976	35
Jordan	1977	29
PA Game Comm.	1980 - 1992	30
deCalesta	1992	31

[a] Jordan's 1974 estimate was reduced by 30 percent due to consistent error found in the sampling technique used.

Forage availability. Due to the young age class of most of the forest, there was little timber cutting during this period (Fig. 1). Some selection cutting removed sawtimber left during the chemical wood cuts and some stands were thinned for pulpwood in the late 1940's. Little browse or mast was being produced during the period.

Predation. The natural predators that became extinct before the turn of the century remained absent during this period. Doe seasons were introduced periodically to reduce the deer herd which had irrupted before this time and remained high. Concerns for orchards and agricultural crops led to public pressure to control the deer population. The first concern in the forestry community was for plantation failures presumably caused by heavy deer browsing. The deer herd had become so large that hunting mortality no longer kept pace with recruitment.

Weather. Weather in the form of severe winters coupled with declining forage availability caused massive starvation of deer in the late 1930's and early 1940's (Fig. 1).

Deer density. This period witnessed the first attempts to quantify deer density. Deer drives were conducted using Civilian Conservation Corps crews; actual counts of deer on known areas were made in four consecutive years. In total, 8,124 acres of the Allegheny National Forest were driven, resulting in estimates of deer density for 1935 through 1938 of 23, 27, 34, and 23 per square mile, respectively (McCain 1939).

Estimates of carrying capacity during the mid-1930's indicated that the forest could support no more than 18 deer/square mile, and that many areas of the forest were overstocked by as much as 220 percent (Clepper 1931). As a result of too many deer, forage growing out of their reach, and successive severe winters in the late 1930's and early 1940's, mass starvation reduced population levels to a 20th century low (Fig 1). In one drainage alone, more than 200 deer were found starved to death[4].

Second-growth Sawtimber Period: 1950-1980

By 1950, some of the stands that regenerated around the turn of the century, though not yet mature, were yielding sawlog-size black cherry, white ash, and yellow-poplar. The timber industry, which had left the area during the 1930's, was small in 1950 but beginning to grow. Stands were managed using the single-tree selection system until 1960. A change to even-age systems of management was initiated in 1960; clearcuts were not to exceed 1 percent per year of all operable timber stands, and no single cut was to exceed 50 acres[5].

A new deer management era in Pennsylvania was initiated during the 1950's. Deer and habitat management were placed on a scientific basis, and food plots were established on the Allegheny National Forest by the Pennsylvania Game Commission under cooperative agreement. Sex ratios and age structure of the harvested deer were used to assess the condition of the herd and antlerless seasons became an annual occurrence.

Vegetation conditions. By the early 1950's, many of the second-growth stands that regenerated from cuttings between 1890 and 1910 had grown to sawtimber size. Individual trees were selectively harvested for sawtimber. Gaps created by these single-tree selection cuts generally were filled by beech, striped maple, or ferns. Understories remained fairly sparse throughout the forest. With a switch to even-age management in 1960, some stands regenerated to early successional stages of woody vegetation consisting primarily of black cherry and red maple. About half of these regeneration cuts regenerated to savannah stands of herbaceous cover along with isolated black cherry and red maple trees (Marquis 1973).

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[4]Curnutt, W.C. 1973. More than 1000 acres a year, the "new look" in reforestation on Allegheny National Forest. In: National Forest Special 50th Anniversary Pub. Warren, PA. Schuler, N. (ed.). Article #8

[5]Stotz, L. 1973. Crosscut'n doublebit, timber sales and timber utilization on the Allegheny National Forest. In: National Forest Special 50th Anniversary Pub. Warren, PA. Schuler, N. (ed.). Article #7

Forage availability. As a result of a steady increase in timber harvesting, forage production increased throughout the period as a whole (Fig. 1). An exception occurred during the early 1970's when there was a drop in timber harvesting due to a lack of stands with adequate advanced regeneration. Techniques for establishing adequate advanced seedlings were being developed at the Forestry Sciences Laboratory at Warren and not yet available. The sharp increase in harvest from 1976 through 1980 initially was the result of a large tornado salvage in 1976-78. By 1980, the use of guidelines for establishing advance reproduction using an herbicide-shelterwood technique made more stands eligible for final harvest. Both the quantity and quality of forage produced by regeneration and intermediate cuts was much less than that produced by cuttings at the turn of the century. Forage produced during this period tended to be of a single species such as black cherry or fern, both of which are low in nutrition and in preference by deer. Species of higher nutritional quality such as sugar maple, white ash, and yellow-poplar were preferentially browsed and eliminated or greatly reduced in abundance by white-tailed deer.

Predation. Hunting and highway accidents involving deer were the major causes of mortality during this period.

Weather. Severe winter weather played a major role in reducing deer numbers during this period. The winters of 1976 and 1977 were long and cold and characterized by heavy snowfall. A series of tornadoes occurred on the Allegheny National Forest during the summer of 1976.

Deer density. From 1950 through 1976, deer populations increased steadily as forage availability increased due to increased final harvest cuts (Fig 1.). In the mid-1970's pre-fawn estimates were made on 2154 acres of the Allegheny National Forest during 1974 through 1977 (Jordan 1977). These estimates, made using the belt-transect pellet-group method, were 50, 43, 35, and 29 deer/square mile, respectively. Three additional National Forest areas totaling 48 square miles that were sampled in 1974 had population estimates of 28, 35, and 41 deer/square mile (Jordan 1977). The deer population declined drastically during the late 1970's due to 2 consecutive severe winters and a sharp decrease in the annual cut on the Forest (Fig. 1).

Modern Period: 1980-Present

In response to an increasing deer herd the Pennsylvania Game Commission adopted a system of issuing doe permits in 1979 that was based on the overwinter carrying capacity of each county. Carrying capacity was determined by a formula involving amount of woodland in the seedling-sapling, poletimber, and sawtimber stage. This resulted in an increase in antlerless permits. A saturation point of 500,000 to 550,000 antlerless licenses annually was not enough to reduce populations to a goal of 18 deer/square mile on the Forest (Pa. Game Comm. 1991). To increase harvests, a bonus deer program was initiated in 1988 to allow hunters to harvest more that one deer. During this period, the Allegheny National Forest was increasing its annual timber harvest substantially due to the maturing of the forest. Annual cuts in the 1980's and early 1990's were more than double those of the 1970's.

Studies by Forest Service scientists at Warren showed that deer also could influence other portions of the ecosystem--songbirds, wildflowers, and other herbaceous plants (deCalesta 1992). Public awareness of the impact of deer on the entire ecosystem has been heightened as a result of these studies.

Vegetation conditions. Salvage from the 1985 tornadoes coincided with increased harvest levels when the Allegheny National Forest Land and Resource Management Plan was implemented in 1986 (USDA For. Serv. 1986). This increased the proportion of the land area in early successional stages, usually in communities dominated by one or several tree species. Partial cutting also increased and was supplemented by a series of insect and disease infestations, including invasion by the gypsy moth (*Porthetria dispar*), elm spanworm (*Ennomos subsignarius*), pear thrips (*Taeniothrips inconsequens*), beech bark disease complex, and a sugar maple decline complex. In understories exposed to increased light as a result of these forces, fern, grass, striped maple, and root suckers of beech often increased in dominance. Toward the end of the period there was increasing anecdotal evidence of increases in the regeneration of birch and of the limited recovery of at least one shrub species (hobblebush).

Forage availability. Although the quantity of forage has increased since the 1970's, quality has remained about the same. Regeneration cuts are restricted to stands with adequate advanced regeneration, generally black cherry. As a result, harvest cuts tend to regenerate to pure black cherry stands. In areas where there are insufficient advance seedlings, the understories generally are occupied by species not preferred by deer, for example fern, grass, striped maple, and beech. Forage availability increased substantially following a number of large tornadoes that struck the region in May 1985.

Predation. As mentioned previously, the Pennsylvania Game Commission attempted to reduce deer populations through increased antlerless license allocation and bonus tags. Road kills continue to be a major source of mortality in the state. In 1992, more deer were killed by motor vehicles on Pennsylvania highways (43,000) than were harvested through hunting in Vermont, Massachusetts Connecticut, and Rhode Island combined (38,520) (David S. deCalesta, pers. commun., 1994).

Weather. Winters during the early 1980's were more severe than normal; these were followed by 8 winters that were much milder and more snow free than normal. The winter of 1994 was the most severe snowfall for the period. As mentioned, tornadoes swept the region in 1985.

Deer density. Deer densities for the Allegheny plateau have remained fairly constant throughout the period. Estimates by the Pennsylvania Game Commission for the 4-county area remained at about 30 deer/square mile for the entire period (Palmer,Wm. Pa. Game Comm. pers.commun.1994). In 1992, data from 11 sites on the Allegheny National Forest obtained by the transect/pellet method showed a range across the forest of 19 to 49 deer/square mile, averaging 29 deer/square mile, which agrees with the Pennsylvania Game Commission's 1990 estimate (Pers. Comm. D.deCalesta 1994).

CONCLUSION

Human activity in northwestern Pennsylvania has shaped and influenced factors that affect species composition, age, and structure of plant communities from pre-settlement time to the present. Pre-settlement forests were characterized by vast expanses of unbroken mature forests intermixed with early successional stages created by natural disturbances such as tornados, hurricanes, and natural fires, and by Native Americans who cleared land for villages and agriculture, and burned the woods repeatedly for berry production and to facilitate hunting and travel.

Since the arrival of European settlers in the region around 1800, man, has manipulated the forests of northwestern Pennsylvania changing their character at an accelerated pace. In particular, human activity has altered the conditions under which forests of the region regenerate. Timber harvesting has increased forage availability for deer which, in turn, have increased from near extinction at the turn of the century to record levels during the mid-1970's (Fig. 1). Laws to protect deer, seasons and bag limits, and a lack of natural predators also have contributed to the overpopulation of deer. With deer densities well above ecological carrying capacity for more than 70 years, all forest communities have been affected as animal-plant, and plant-plant interactions have been influenced. The result is a less diverse ecosystem.

Through the first 6 historical periods reviewed in this paper, there was a tight link between forage availability, as represented by timber harvesting and deer density. The experience of the modern period suggests that this link is not unbreakable. Although current deer densities are well above goal levels, the herd has been stabilized at about 30 deer/square mile throughout the modern period due to Pennsylvania Game Commission's program of antlerless deer harvest and despite a dramatic increase in forage availability.

The experience of the modern period also allows us to focus on the challenge of managing deer and forested ecosystems on a landscape scale. We can anticipate cyclic fluctuations in forage availability as forest and ecosystem managers attempt to achieve a balance in age classes. At the same time, both the number of hunters have declined steadily since peaking in 1982 and the number of new hunters as reflected by the number of people taking hunter-

training has also experienced a steady drop in numbers (deCalesta and Whitmer 1992). Careful coordination of management of both forage and deer density will be required if the region's forests are to recover from decades of overbrowsing during this century. The best short-term solution is to maintain high forage production through sustainable harvest levels while increasing hunting pressure to reduce density to 18 deer/square mile in the Allegheny region.

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EFFECTS OF TWO-AGE MANAGEMENT AND CLEARCUTTING ON SONGBIRD DENSITY AND REPRODUCTIVE SUCCESS

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Abstract: We examined density and reproductive success of passerine species on 7 uncut forest stands and on 12 stands harvested 10-14 years ago on the Monongahela National Forest of West Virginia (6 clearcut stands and 6 stands harvested using 2-age management). In 2-age management, stands resemble a shelterwood cut with 10-30 overstory trees/acre left uncut. Uncut periphery areas adjacent to treatments were used to examine edge effects. All stands and periphery areas were censused 5 times, May-July 1993 and 1994, with the variable-circular plot technique and were thoroughly searched for nests. Two-age treatment stands had higher densities (650 birds/40 ha) than all other treatment and periphery stands ($F = 19.08$, $P = 0.0001$). Clearcut treatment stands averaged 522 birds/40 ha, no-harvest (341 birds/40 ha). Uncut periphery stands of two-age and clearcut treatments did not differ significantly from no-harvest stands. Intensive nest searching of all plots and periphery areas yielded complete nest histories of 90 nests representing 23 species. Nest success, calculated using the Mayfield method, was as follows: two-age treatment stands 49.8%, two-age periphery 61.2%, clearcut treatment 53.9%, clearcut periphery 48.2%, and no-harvest stands 59.6%.

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AN ANALYSIS OF PHENOTYPIC SELECTION IN NATURAL STANDS OF NORTHERN RED OAK (*QUERCUS RUBRA* L.)

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Abstract: Comparison of growth and stem quality parameters of 19-year-old progeny from superior and comparison trees indicates that rigorous phenotypic selection of trees in natural stands may not be an efficient method of parent tree selection for *Quercus rubra* L. Total tree height, dbh, number of branches in the butt log, fork height, and number of mainstem crooks of progeny from 11 phenotypic select trees and their co-occurring comparison trees were analyzed. Pooled data from 4 test sites indicated no difference in dbh or number of mainstem crooks among select and comparison families. However, comparison families were significantly ($p \leq 0.05$) taller, had significantly less branches in the butt log, and higher fork heights than select families. Similar patterns of variation were also found within test sites. The large degree of within-stand variation in growth and stem quality traits among families indicates that selecting several phenotypically above-average candidate trees may be more effective than rigorously selecting a smaller number of phenotypically superior *Q. rubra* trees.

INTRODUCTION

Mass selection of superior phenotypes is the most widely used method of obtaining material for the initial stages of tree breeding programs (Zobel and Talbert 1984). Regional *Quercus rubra* L. improvement programs have recently been initiated in both the central and southeastern U.S. (Cox and Schlarbaum 1991, Robison and Overton 1987). These programs have used mass selection to provide material for the establishment of seed orchards and test plantings. Results of relatively young provenance/progeny tests of *Q. rubra* have shown a large degree of intra-stand and provenance growth variation (Farmer and others 1981, Houston 1987). This suggests that gains through mass selection are possible however, gains from plus-tree selections have proved inconsistent (Robison 1988). Previous tests have focused primarily on growth characteristics which, for some hardwood species, have relatively low narrow-sense heritabilities (Zobel and Talbert 1984). The use of phenotypic selections in natural stands of *Q. rubra* remains appealing in light of potential stem quality gains, certain aspects of which may have high narrow-sense heritabilities (Zobel and Talbert 1984). Little information on potential stem quality gains is available for *Q. rubra* as previous analyses of provenance/progeny tests were completed in relatively young plantings (Kriebel 1993).

This paper addresses the question of whether mass selection criteria need be extremely rigorous to provide gains in the initial stages of *Q. rubra* tree improvement programs. Specifically, we were interested in comparing the performance of open-pollinated progeny from some of the best phenotypes in the region to the performance of open-pollinated progeny from several above-average phenotypes or "comparison" trees from the same stands as the select phenotypes. Results of our analysis may aid in designing effective selection strategies for *Q. rubra*.

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METHODS

Experimental Materials

We studied 19-year growth and stem quality data from 43 open-pollinated *Quercus rubra* L. families from a provenance/progeny test of 226 *Q. rubra* families established in 1973 by the Tennessee Valley Authority (TVA) (Farmer 1980). Mother trees were located throughout the Tennessee and lower Ohio Valleys between Long. 81°58'-89°00' W, Lat. 34°15' - 38°50' N, and 152 - 1402 m elevation. Acorns were collected in the fall of 1971 from dominant and co-dominant trees with good size and form. Twenty-two of the mother trees were selections made in the 1960's by the TVA. These mother trees were the best phenotypes in their stand, based on form and size, as well as being representative of the best phenotypes in the region (K. Taft, TVA, personal communication). Fifty-nine additional mother trees were located in stands with the select trees and could be considered "comparison" trees in the mass selection process (Zobel and Talbert 1984). The remaining 145 mother trees were above-average phenotypes located in stands where intensive selection had not been conducted by TVA personnel (D. Scanlon, TVA, personal communication).

Establishment and Measurement of Provenance/Progeny Tests

Acorns were planted in 2 replicates at a TVA nursery near Norris, TN (Lat. 36° N, Long. 84° W), in the fall of 1971. Seedlings were lifted in the spring of 1973 and graded for size. The 16 largest seedlings of each family were planted at test site I (Fig. 1), and the remaining ungraded seedlings were distributed to 11 other test sites. A randomized complete block design was selected for each test site. However, the number of blocks, trees per plot, and families differed by test site. By 1991, only 4 plantations remained intact for analysis (Fig. 1, Table 1). Total number of families, family plot size, number of replicate blocks, and number of select and comparison families included in our study are shown for each test site in Table 1.

Table 1. Description of *Quercus rubra* provenance/progeny test sites.

Test Site	Number of Families			Blocks	Plot Size (# of trees)
	Total	Select	Comparison		
I LBL, KY	226	10	30	8	2
II Parke Co., IN	134	8	17	3	2
III Ames, TN	80	3	2	6	4
IV Coshocton,OH	176	10	30	7	4

All test sites were evaluated during the dormant season of 1991-1992. Growth and stem quality measurements used in this analysis included total tree height (THT); diameter at 1.3 m height (DBH); number of branches (BR#), live or dead, in the butt log (0.32-5.3 m height); height to lowest fork (FKHT, fork defined as the occurrence of 2 crown branches resulting in a mainstem deviation $\geq 10°$); and number of crooks below FKHT (CR#, crook defined as mainstem deviation $\geq 10°$). Heights were measured with a clinometer (Suunto) positioned 15.2 m from each tree. A diameter tape was used for DBH determination, and a height pole was used to designate the butt log. A protractor was used to establish angle deviation for fork and crook identification.

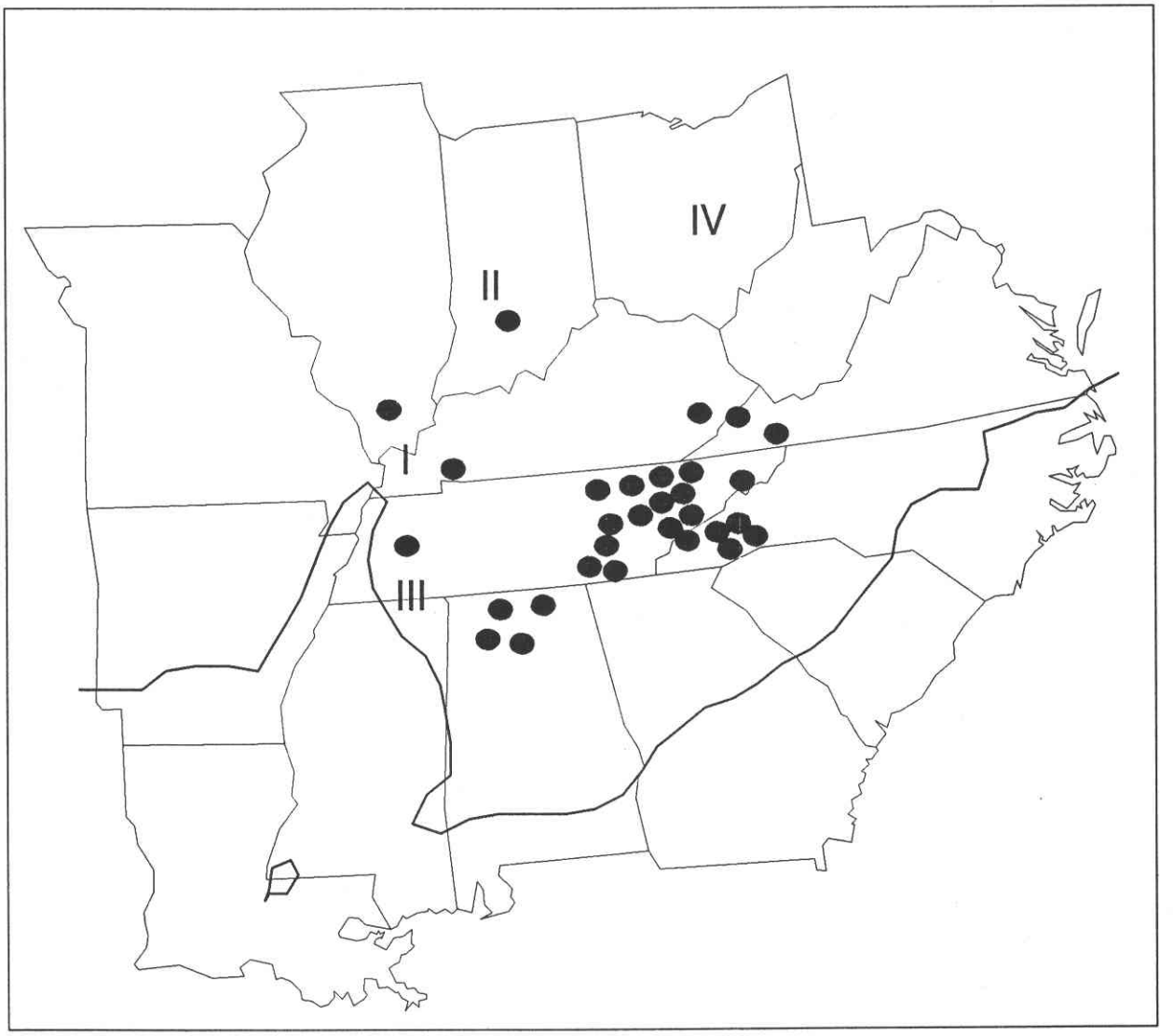

Figure 1. Stand location of mother trees (solid circles) and test site location (Roman numerals) of *Quercus rubra* L. genetic tests. Heavy lines indicate the natural range of *Q. rubra* L.

Data Analysis

Analyses were conducted on families from stands containing one select tree and at least one comparison tree. In the TVA test, several select trees were represented by a limited number of seedlings, and a small number of stands contained more than one select tree. Families of these trees were deleted from the analyses as were associated comparison tree families. The 145 families from stands lacking select trees also were deleted from the analyses. Measurements from families of the remaining 11 select trees and their 32 associated comparison trees formed the data set for our study.

Fork heights were converted to percent of total tree height (FKPER) and normalized using an arc sine transformation (Steel and Torrie 1980). All other variables were subjected to Wilk-Shapiro/rankit plots to check for normality. Data were subjected to a series of analyses and were analyzed untransformed. Pooled select and pooled comparison families were compared over all test sites as well as within test sites. Within-stand comparisons (i.e. analyses of select families and their associated comparison families) were also completed within test sites and for data pooled over all test sites. Two sample t-tests were used to detect differences at $p \leq 0.05$.

RESULTS AND DISCUSSION

Growth Variables

No significant differences ($p > 0.05$) in mean THT were found among pooled select and pooled comparison families within any of the test sites. Mean DBH of select versus comparison families was significantly different only at test site I (Fig. 2). Mean DBH for select families (12.05 cm) was 6.8% higher than comparison families (11.22 cm). None of the other test sites analyzed showed any differences among select and comparison families for DBH. While test site I was the location where the 16 largest seedlings of each family were planted, the importance of this in regards to a DBH difference among groups is not clear.

No significant differences ($p > 0.05$) were observed in analyses of data pooled over all test sites for mean DBH among select and comparison families, but mean THT's of select families were significantly less than those of comparison families (Fig. 3). However, the difference was relatively small, approximately 4 percent.

Stem Quality Variables

There were 12 possible variable/test site analyses evaluating pooled select versus pooled comparison family data within each test site. Of these 12 analyses, we found only 4 instances of significant differences ($p \leq 0.05$) for the stem quality variables (BR#, CR#, and FKPER) and only 2 instances where select families outperformed comparison families. Mean CR# was significantly less for select families (0.02) at test site I than for comparison families (0.12) but at test site II the opposite was true (Fig. 4). Differences for mean FKPER were also found at test sites I and II (Fig. 5). Only at test site II did select families perform better than comparison families, averaging 64.2 and 53.4 percent FKPER, respectively. No differences in select versus comparison families for mean BR# were found within test sites.

Data pooled over all test sites indicated no significant differences for mean CR# among select and comparison families. However, significant differences among groups in both BR# and FKPER were found ($p \leq 0.05$) (Fig. 6). Comparison families had the smaller mean BR# (7.14), compared to select families (7.99), representing a 10 percent difference. Comparison families also outperformed select families in FKPER having a mean of 70.7 compared to 66.0 for select families, representing a 7 percent difference.

Within-Stand Comparisons

Analysis of data pooled over all test sites found only 16 of the 51 within-stand comparisons of growth and stem quality variables significant ($p \leq 0.05$). Results of these analyses were expressed in terms of percent differences in means between select and comparison families (Table 2). Of the 16 significant differences, only 6 were positively associated with select families, while comparison families outperformed select families in 10 analyses.

The results of within-stand comparisons among select and comparison families within test sites were similar to the within-stand analyses that were pooled over test sites. Six significant differences ($p \leq 0.05$) were found out of 50 analyses performed on growth and stem quality variables for select versus comparison families in test site I. Only 2 of these significant results were positively associated with select families. Similar patterns occurred in within-stand analyses at each of the other test sites.

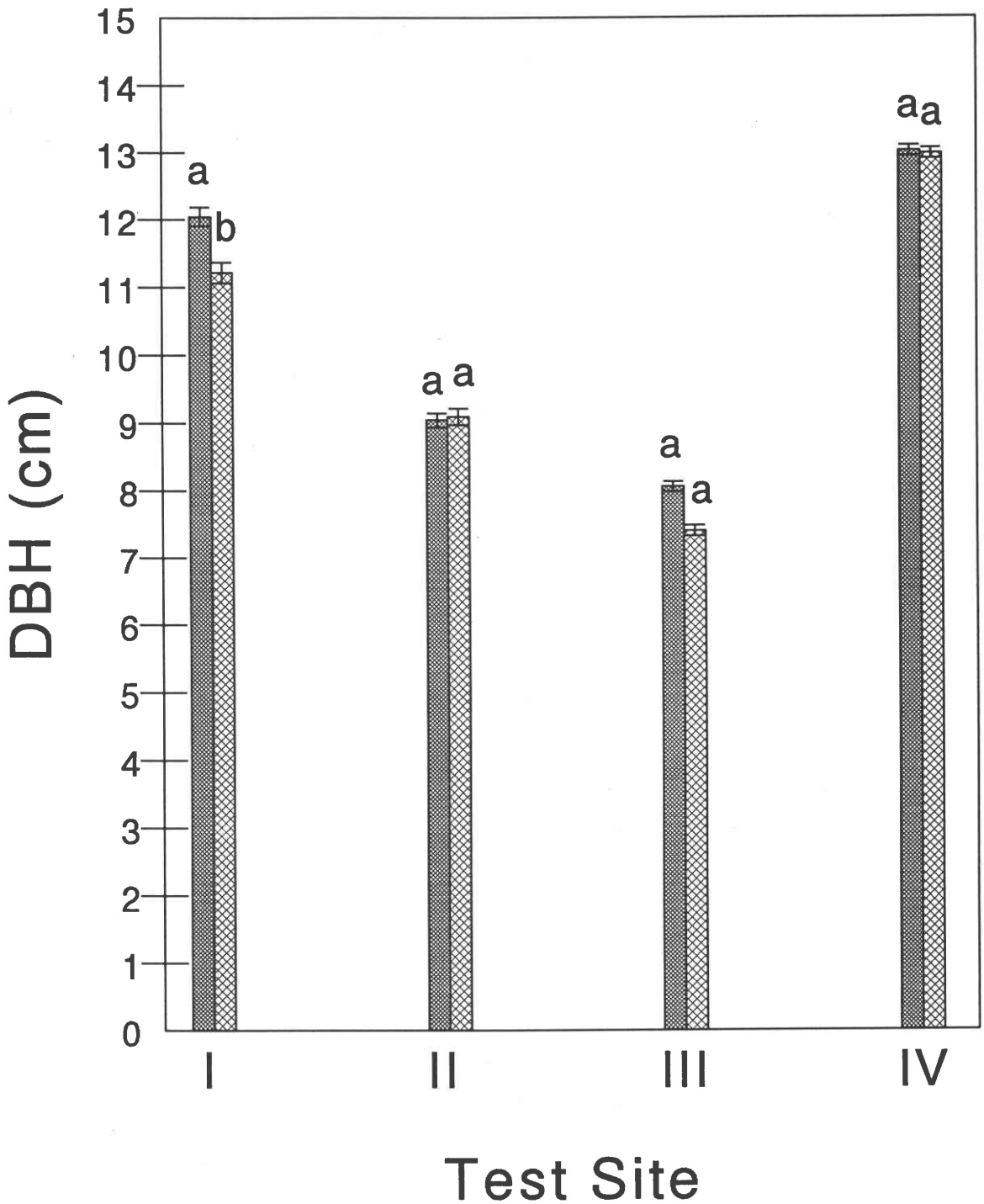

Figure 2. Contrasts of mean DBH among select (shaded) and comparison (hatched) families for each test site. Bars represent means with standard errors for each group. Significant differences ($p \leq 0.05$) are indicated by different letters.

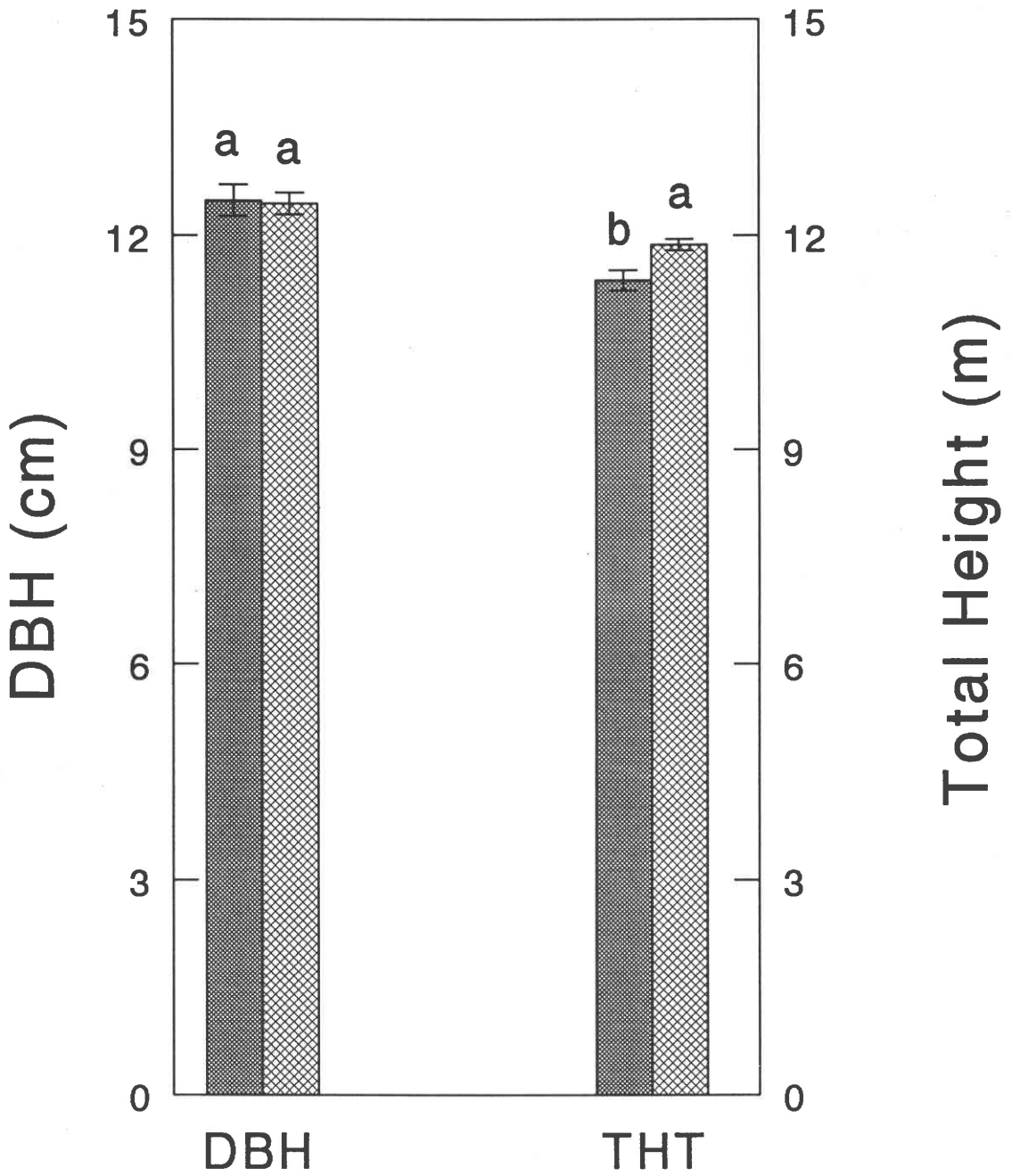

Figure 3. Comparison of mean DBH and mean THT among 11 select (shaded) and 32 comparison (hatched) families. Bars represent means with standard errors for each group pooled over all test sites. Significant differences ($p \leq 0.05$) are indicated by different letters.

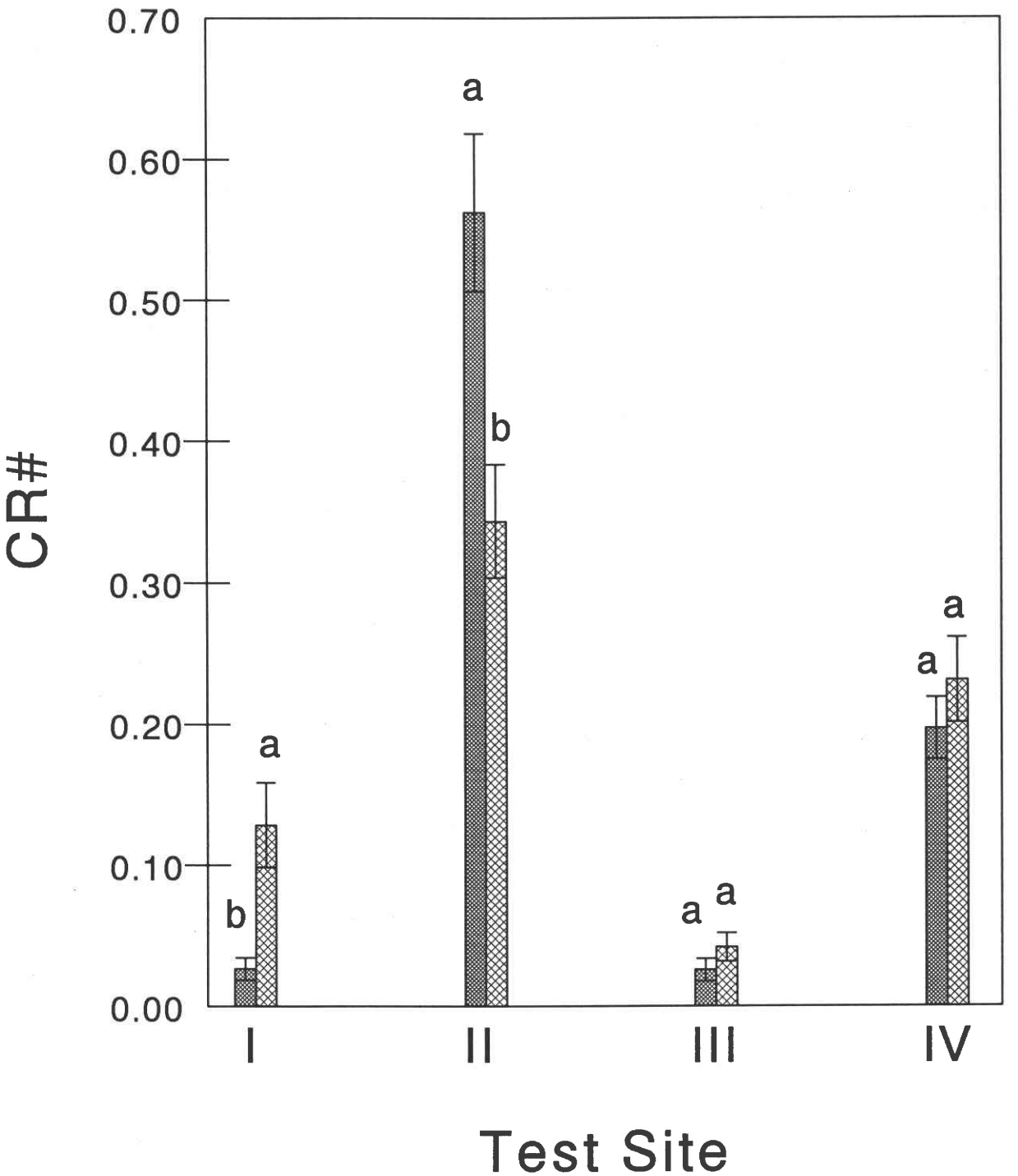

Figure 4. Contrasts of CR# (producing ≥ 10° deviation of mainstem) among select (shaded) and comparison (hatched) families for each test site. Bars represent means with standard errors for each group. Significant differences ($p \leq 0.05$) are indicated by different letters.

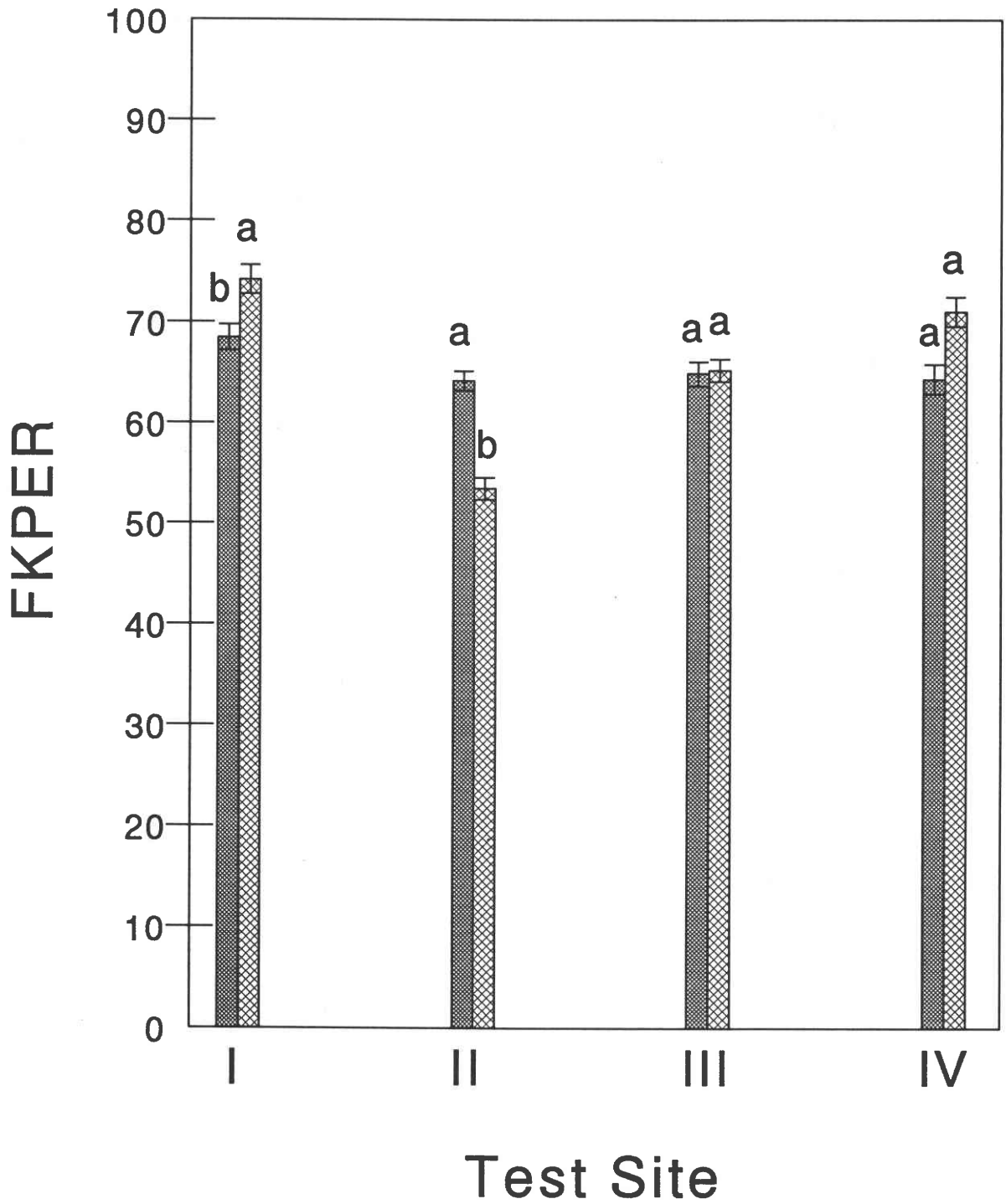

Figure 5. Contrast of FKPER (fork height, expressed as percent of total tree height) among select (shaded) and comparison (hatched) families for each test site. Bars represent means with standard errors for each group. Significant differences ($p \leq 0.05$) are indicated by different letters.

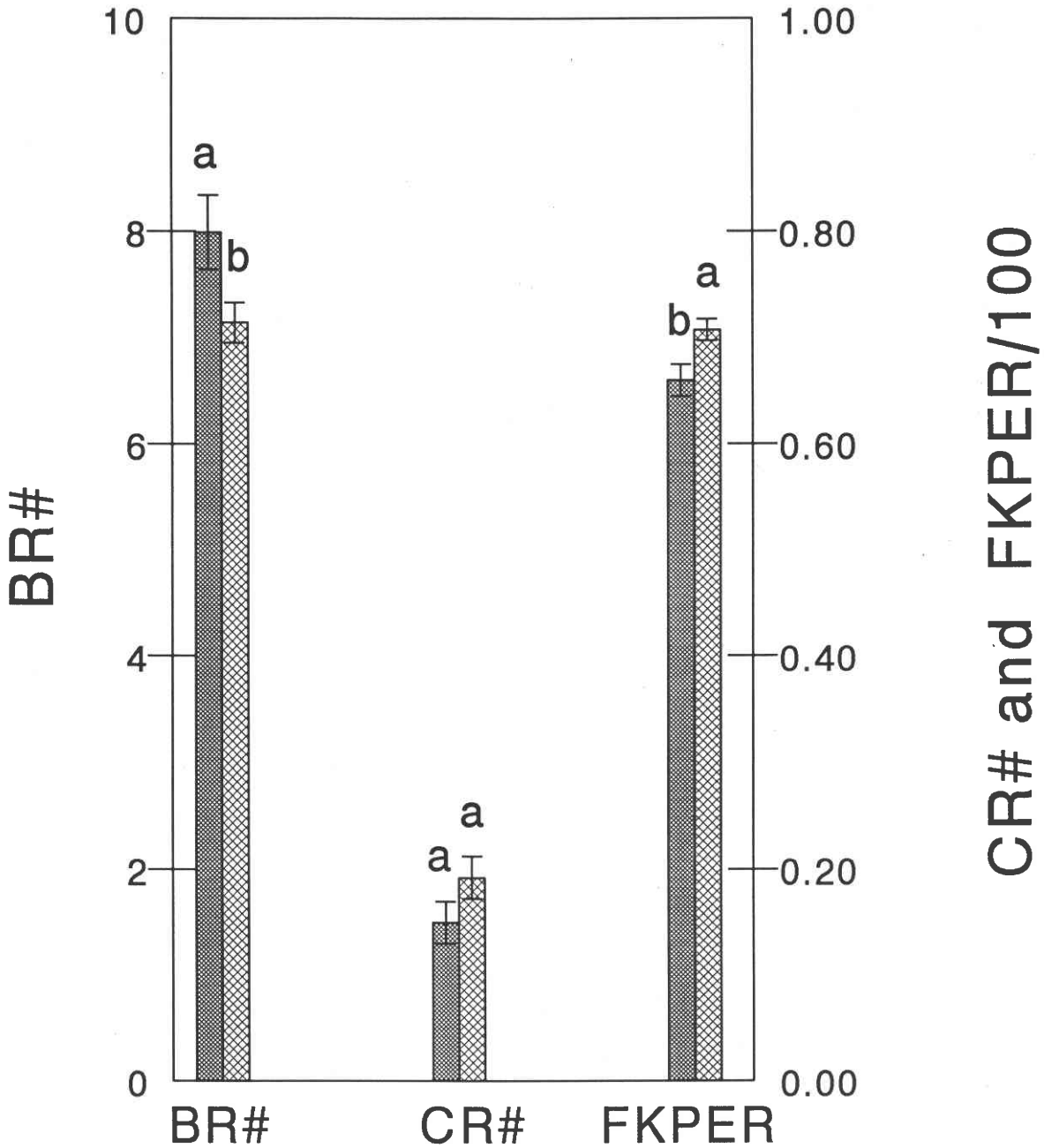

Figure 6. Comparison of BR#, CR#, and FKPER, among select (shaded) and comparison (hatched) families pooled over all test sites. Bars represent means with standard errors for each group. Significant differences ($p \leq 0.05$) are indicated by different letters.

Table 2. Percent differences in performance between select and comparison *Quercus rubra* families pooled over all test sites.

Select Family ID# (# progeny)	Number of Comparison Families (# progeny)	Percent Difference of Select Versus Comparison Families				
		DBH	THT	BR#	CR#	FKPER
2412 (22)	4 (82)	9.3	3.5	**-24.6**[1]	-47.5	10.7
2419 (30)	3 (45)	3.5	-4.5	32.0	48.3	65.6
2421 (6)	4 (60)	**19.2**	-5.2	-21.0	----	-40.0
2427 (31)	1 (17)	----	----	**-71.7**	27.1	-18.9
2429 (24)	2 (43)	-0.4	-4.2	1.4	-35.0	-14.0
2451 (42)	2 (38)	**-22.1**	**-22.1**	**48.2**	-69.8	-7.5
2452 (18)	2 (28)	-4.8	-8.8	**46.1**	-92.6	**-28.5**
2457 (32)	4 (93)	-9.9	**-11.4**	-7.1	42.9	3.3
2458 (37)	2 (58)	**-19.5**	**-16.3**	22.0	-146.7	**15.8**
2462 (31)	6 (163)	**15.2**	**8.3**	-0.4	-14.8	-22.1
2471 (24)	2 (76)	**-17.9**	**-10.4**	20.4	---	11.6
(297)	32 (703)	-2.7[2]	-7.1	4.1	-32.0	-2.2

[1] Bold values represent a significant difference at the 95% probability level using a two sample t-test.
[2] Arithmetic means.

CONCLUSIONS

Our analyses indicate that selecting one tree per stand using rigorous phenotypic selection was no more effective in providing initial improvements in growth and stem quality of *Q. rubra* than selecting a number of good phenotypes in a stand. These results are similar to those obtained for *Q. alba* (Schlarbaum 1993). Similar results and recommendations have also been obtained for other species (e.g. Pitcher 1982). The relatively large differences among select and comparison families within stands, indicating large genetic variances, suggest that several of the best phenotypes in a stand should be used to obtain seed. Progeny testing should be used to establish rankings for candidate mother trees for future collections. However, our study focused on families from relatively large and well formed trees in each stand. We can not speculate on the relative gains possible from selection across the entire range of phenotypes.

Our analyses of select versus comparison progenies indicate low genotype-phenotype correlations in the parental trees, and thus provide little encouragement for rigorous phenotypic selection in natural stands. This may not be surprising given the nature of development of many oak dominated stands. Canopy tree recruitment comes predominately from advanced regeneration subjected to disturbance (Sander 1972, Watt 1979). Oak advanced regeneration developing in non-disturbed stands does so slowly. Many biotic and anthropogenic factors may

influence phenotype during this time. Disturbance which ultimately leads to rapid development of oak cohorts may also alter phenotype.

Selection of *Q. rubra* trees for tree improvement might successfully utilize a mother tree or candidate tree approach. This involves selecting a number of good phenotypes, relying on progeny testing to indicate superior or elite mother trees. Selecting a number of good individuals from a stand eliminates detailed grading procedures, but ensures that the best phenotypes are represented in collections. This approach would increase the number of initial selections and make identification of each selected tree easier compared to a more rigorous regional grading system. Widening the options for individual tree selection may be especially advantageous in years with little mast production. While the nature of *Q. rubra* may necessitate this approach, it may be possible to accelerate gains through juvenile trait selection (Bailey and Steiner 1989, La Farge and Lewis 1987). However, the time interval necessary to determine reliable estimates of family rankings for growth parameters in this species is still under scrutiny (Houston 1987, Kriebel and others 1988).

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POLLINATION BIOLOGY OF NORTHERN RED AND BLACK OAK

Robert A. Cecich and William W. Haenchen[1]

Abstract: Pistillate flower abortion in northern red oak and black oak was evaluated in relation to pollination and fertilization. The presence, position, and characteristics of the pollen grains, pollen tubes, and ovules were determined with bright field and fluorescence microscopy. Flower survival counts were made weekly, from late April to mid-September. Both species have rudimentary ovules and small locules during the first growing season. Pollen tubes cease growth during the first growing season in mid-May at the level of the distal end of the perianth and juncture of the three stigmas. In the next growing season, meiosis in the mature ovules may trigger the advance of the pollen tubes into the locules. Fertilization occurred in mid-June for northern red oak and late June in black oak.

INTRODUCTION

Bumper crops of acorns are considered to be sporadic or episodic. Sork and others (1993) found that the size of a given acorn crop was determined by both flower abundance and survival of the flowers to maturation. Most of the potential seed crop is lost when the pistillate flowers abort before fertilization (Cecich 1991). As members of the subgenus Erythrobalanus, northern red oak (*Quercus rubra* L.) and black oak (*Q. velutina* Lam.) require two growing seasons between pollination and seed maturation; only pollination occurs in the first season. However, it is during the 6-8 weeks after pollination in that first season that most of the abortion occurs. In northern red and white oak (*Q. alba* L.), up to 100 percent of the flowers may abort during that 6-8 week period (Cecich 1991). The possible role of insects, especially the treehoppers (Membracidae), as factors in oak flower abortion has been previously addressed (Cecich 1993).

Because of the time that most of the flower abortion occurs, the pollination process should be examined. The success of the acorn crop can be related to pollen dispersal. Emergence of the catkins and shedding of pollen increase with rising temperatures (Romashov 1957). Rainy weather, often associated with decreased temperature, can reduce pollen dispersal. Sharp and Chisman (1961) concluded from field observations that pollen dispersal occurred when relative humidity dropped and remained below 45 percent for several hours, but they did not mention the success of the acorn crop. Sharp and Sprague (1967) found no correlation with relative humidity and acorn yields in their field studies and concluded that temperature was a primary factor in determining acorn crops. Using growth chambers, Wolgast (1972) and Wolgast and Stout (1977) demonstrated experimentally that relative humidity at the time of pollen shed and stigma receptivity can limit the size of an acorn crop in *Q. ilicifolia* Wangenh. No acorns matured when relative humidity exceeded 61 percent, but about half the flowers matured into acorns when relative humidity was lower.

Past observations of *Quercus* pollen tube growth are conflicting and incomplete. Benson (1894) first observed pollen tubes in *Q. robur* L., a member of the Lepidobalanus (white oak) subgenus, in the locules, just before fertilization. She did not see them in the stigma or style. Jovanovic and Tucovic (1975) observed that pollen germination in *Q. robur* was completed within 24 hours, but fertilization occurred 6-7 weeks later, and speculated that pollen tube growth did not proceed until the ovule had completed development. In contrast, Allard (1932) noted that "When pollen reaches the stigma of members of the white oak group, the growth of the pollen tube containing the male cells follows an uninterrupted advance into the tissues of the style until the ovules are fertilized." Allard (1932) also found that in the red oak group the pollen tubes ceased growth at the base of the style until the following spring when fertilization occurred. Unfortunately, no details of pollen tube growth were documented in any of these reports. Therefore, in this study, we provide the first details of the pollination biology of a Missouri population of northern red oak and black

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oak during the first and second growing seasons. Pollen tube growth is examined in relation to the time of pollen shed, fertilization, and flower survival.

METHODS

The trees in this study were located along a forest road in a closed stand on the Thomas S. Baskett Wildlife Research and Education Area, Ashland, MO (38°N, 92°W); operated by the University of Missouri in cooperation with the Missouri Department of Conservation. Sampling was done from a hydraulic lift truck equipped with a 40-foot boom and reaching a height of 50 feet.

The number of pistillate flowers, their survival and anatomical development, and the time of pollen shed were determined from 1990 to 1994. Data sets for 12 black oaks (BO) and 4 northern red oaks (NRO) from the growing seasons of 1990-1991, 1991-1992, and 1992-1993 were selected to demonstrate the variation in flowering and survival. Pistillate flowers, from the upper and middle crown regions, were counted on the current year's growth of five 2nd-year (one-year-old) branches per tree. Pistillate flowers are found in the axils of current year's leaves. The average number of flowers per 2nd-year branch ranged from about 3-63, depending on the tree and the year. These same trees and branches were used to count the 2nd-year flowers that matured into acorns. For microscopic examination, ten pistillate flowers were collected weekly from one BO and one NRO, during mid-April to Mid-September, from 1991 to 1994. In the spring of 1993 and 1994, two additional black oaks were sampled. Estimates of the dates of pollen shed were done on the same trees as the flower counts, but during the years of 1991-1994.

Flowers were fixed in FAA (formalin : acetic acid : alcohol), embedded in Paraplast-Plus, cut on a rotary microtome, and sections, both longitudinal and transverse, were attached to slides with Haupt's adhesive. For observation by bright field microscopy, sections were cut 10-µm-thick, stained in safranin-fast green, and coverslips mounted in Permount. For fluorescence microscopy, sections were cut 20-µm-thick and stained with 0.005% aniline blue in 0.15M K_2HPO_4 at pH 8.2 for 10-30 minutes to locate callose, a 1,4 ß-glucan polysaccharide that fluoresces a bright yellow-green, in the pollen tube wall (Currier 1957). Coverslips were mounted in the staining solution. Sections were examined with a Zeiss Universal microscope, illuminated by an HBO 200 W mercury lamp, in conjunction with the exciter filters UG-1 and BG-38 and the barrier filters 44, 47, and 53. Photomicrographs were taken with a Wild-Heerbrugg MKa4 Photoautomat camera using Kodak Ektachrome 400 HC film.

RESULTS

Mean survival curves of NRO and BO pistillate flowers, from 1990 to 1993, are shown in Figures 1-3. In NRO, most of the flower loss occurred in the first 6-8 weeks after pollination. However, additional loss was recorded in conjunction with the overwintering period of the pollinated flower. Since observations were not made between September and April of the following year, we could not determine the timing or cause of this loss. Thus, the loss that occurred overwinter (between September and April) is expressed as the increment between April and May in those Figures.

Staminate flowers that produce the pollen are found on inflorescences or catkins. Catkins are found in buds that only produce catkins ("male buds") or in "mixed buds" (buds that contain vegetative shoots and catkins, or buds that contain vegetative shoots, pistillate flowers and catkins). Depending on the year, pollen of NRO is usually shed during the week of April 20 at the central Missouri site, and about one week later in BO (Table 1). Pistillate or female flowers are found in the axils of newly emerging leaves of the terminal bud or its subtending lateral buds. In NRO and BO, the pistillate flowers (Figure 4) tend to be distributed among the leaves of the central and basipetal portion of a branch. Based upon our weekly sampling schedule, the stigmas of the pistillate flowers (Davis 1966) seemed to be receptive for no more than one week. Our assessment of receptivity was based on the

flexibility and bright red coloration of the stigmas. Once the stigmas turned brownish in color and became rigid, we felt that receptivity had passed. Oak stigmas are classified as dry (Heslop-Harrison and Shivanna 1977), meaning that they do not produce visible secretions on the stigmatic tip.

Table 1. Annual variation in the beginning date of pollen shed. The duration of pollen shed for a species in the stand seemed to be less than one week.

YEAR	RED OAK	BLACK OAK
1991	0*	1 May
1992	24 April	1 May
1993	1 May	7 May
1994	22 April*	29 April

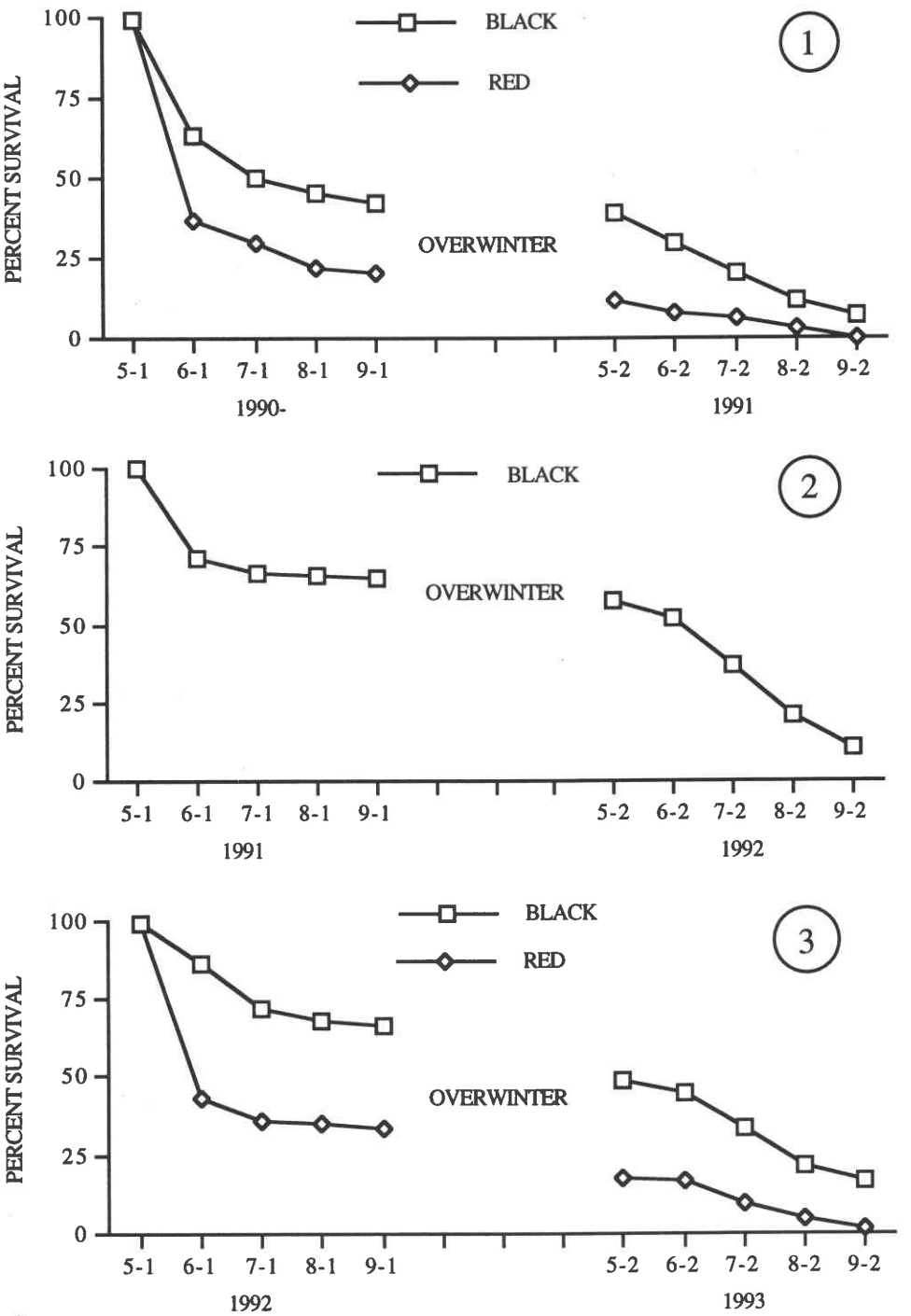
*No pistillate flowers in 1991 and few to none in 1994.

Pollen grains covered all surface tissues of the flowers, but it was only those that landed on the stigma surfaces that affected pollination (Figure 5). Heslop-Harrison and Heslop-Harrison (1985) have reviewed pollen-stigma interactions that specifically promote the germination and penetration of the pollen tube into the stigmas. Germination of oak pollen has been shown to occur within 24 hours of landing on the stigma (Jovanovic and Tucovic 1975). Our weekly collections did not permit verification of that timing. Pollen tubes were so numerous in some styles that accurate counts were not possible.

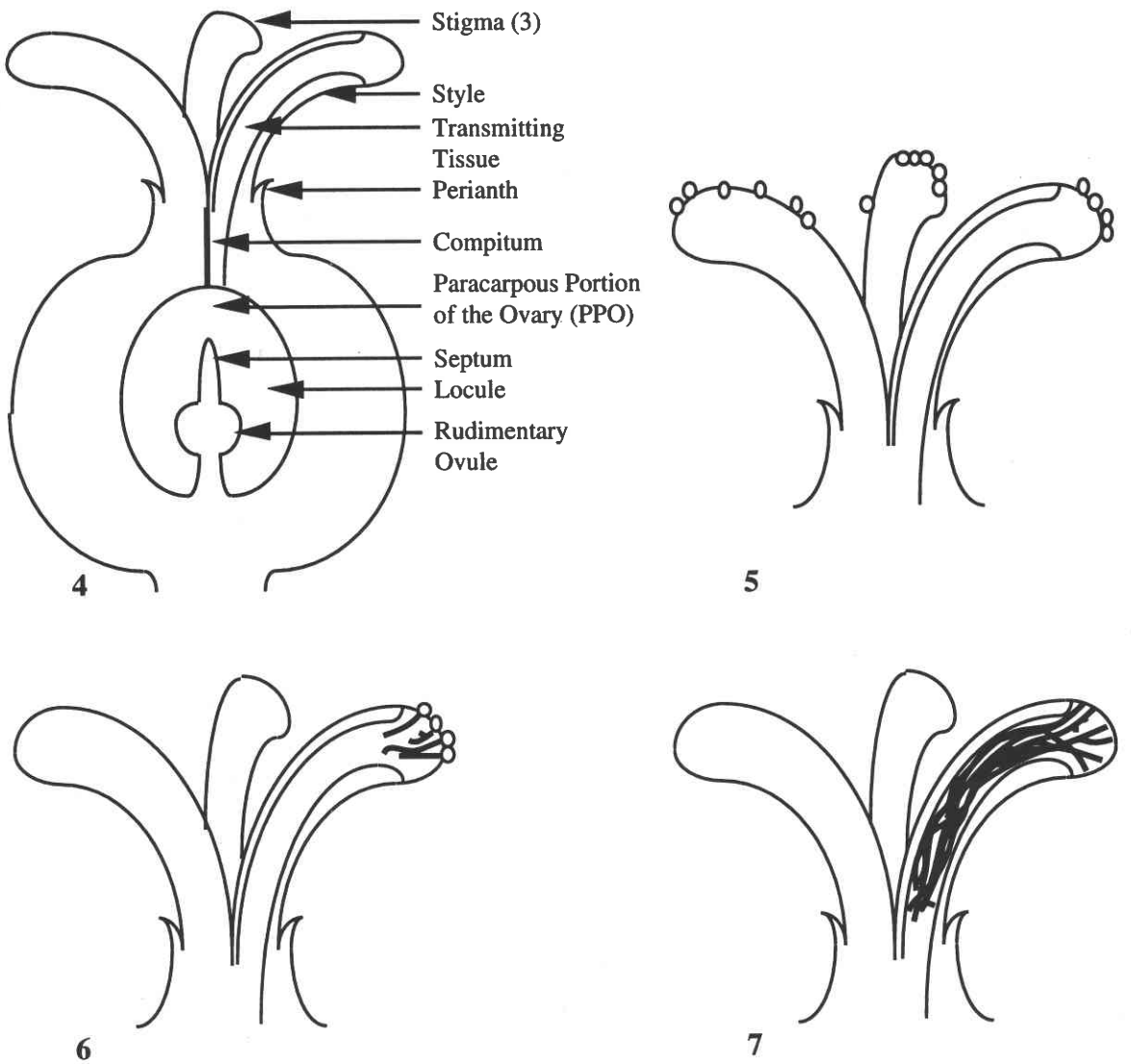
Both NRO and BO had the same pattern of pollen tube growth after the grains landed on the stigmas. The pollen tubes penetrated the stigmatic epidermis and synthesized a callose plug within the pollen tube, effectively isolating the contents of the tube from the now-empty pollen grain (Figure 6). One common feature of oak pollen tubes was the occurrence of several callose plugs within a single pollen tube. Callose plugs were generally amorphous masses. The pollen grains fell off the stigma within the week. Some pollen tubes produced numerous branches upon penetrating the stigma epidermis into the transmitting tissue of the style. Pollen tubes grew through the "central core" of the style in the parenchymatous, transmitting tissue toward the ovules (Figure 7).

The pollen tubes of both species penetrated downward, through the transmitting tissue until they reached the level of the perianth, just above the juncture of the three stigma/styles, on or about May 15. Thus, both species ceased penetrating on the same date, even though NRO began pollen tube growth at least one week before BO. No further pollen tube growth occurred during the first growing season. The ovary tissues of NRO and BO pistillate flowers are not well-developed in the first growing season. The three locules are small and contain a few hairs that surround the rudimentary ovules. From mid-May until the following spring, there were no observable changes in the structure of the pistillate flower or the position of the pollen tubes.

During the second growing season, pollen tubes of NRO resumed elongation on or about the May 21 collections. By that date, the ovary tissues had greatly enlarged. The ovules became organized into a nucellus that was surrounded by inner and outer micropyles. The locules enlarged and the number of hairs in the locules increased. In the next specimen collection (about May 28), pollen tubes were seen in the compitum (Carr and Carr 1961), the narrow "cylinder" of empty space that separates the three styles within the flower (Figure 8). After leaving the basipetal end of the compitum, the pollen tubes entered the paracarpous portion of the ovary (PPO) (Carr and Carr 1961). The central axis of the ovary is formed by the fusion of three septae. Each septum gives rise to a pair of rudimentary ovules. The septae are incompletely developed toward the top of the locules, creating a free space (the PPO) that connects each locule to the other two. Thus, as pollen tubes enter the compitum and PPO, they can fertilize any of the six ovules. Pollen tubes were randomly oriented and interwoven in the PPO.



Figures 1-3. Survival of pistillate flowers in the first growing season and developing acorns in the second growing season. Fig. 1. 1990-1991 growing seasons. Fig. 2. 1991-1992 growing season. There were no flowers on the red oak in 1991. Fig. 3. 1992-1993 growing season. For purposes of plotting data, we have used an estimated value for the first day of each month, not the weekly data.



Figures 4-7. Line drawings of the pistillate flower and pollen tube growth. Fig. 4. A median-longitudinal section of an oak pistillate flower with pertinent parts labelled. Fig. 5. Pollen grains on the three stigmas. Fig. 6. Pollen tubes grow through the stigma epidermis and produce a callose plug. Branches may form on the tubes. Fig. 7. Pollen tubes cease growth in mid-May at the level of the perianth.

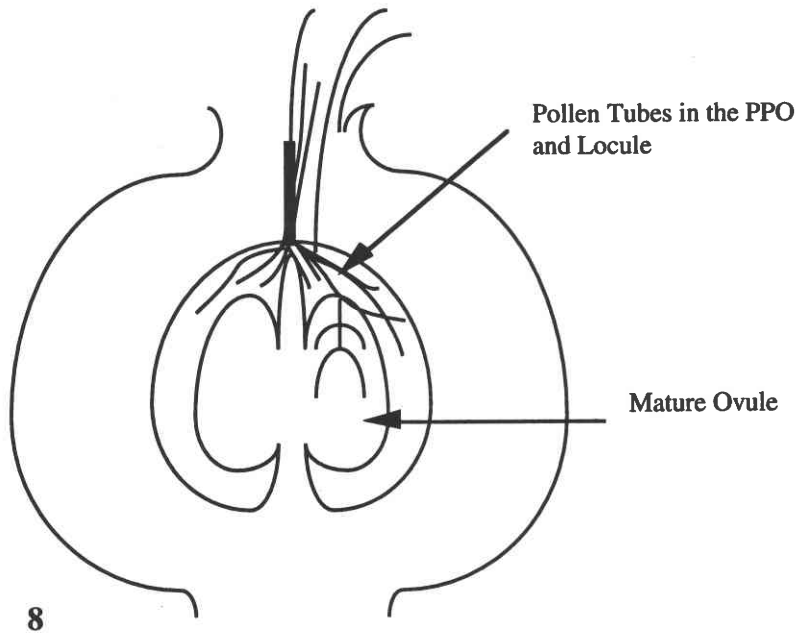


Figure 8. When pollen tubes resume growth, they leave the transmitting tissue and enter the compitum and paracarpous portion of the ovary (PPO), where they can randomly enter any of the three locules and fertilize any of the six ovules and their eggs. See Fig. 4 for additional labelling of flower parts.

Pollen tubes of NRO resumed growth in the May 21 collections and were observed in the locules, below the level of the PPO, in the June 4 and June 11 collections, as the megaspore mother cells were undergoing meiosis. The free nuclear endosperm stage was observed on June 18, indicating that fertilization had occurred after June 11. The BO pollen tubes were first seen resuming growth as they entered the compitum on June 4, two weeks after NRO. However, a free nuclear endosperm was not seen until July 2, indicating that fertilization occurred about two weeks after it did in NRO.

DISCUSSION

Our observations suggest several possible mechanisms for the control of pollen tube growth. Since both species ceased elongation on May 15, an external or environmental factor may be involved. This relationship has been demonstrated in a number of annual plants and grasses (Lyndon 1992). Pollen of *Q. pubescens* W. requires a higher temperature for germination (38° C) than pollen of*Q. robur* (20° C). Low and erratic fertility of *Q. pubescens* in cold habitats was attributed to the prevention of pollen germination (Jicinska and Koncalava 1978). However, the control over pollen tube elongation may be sporophytic, as manifested physiologically through the incomplete development of the ovules. For instance, the pollen tubes may stop penetrating the transmitting tissue because the rudimentary ovules may not be providing the correct signal, perhaps a hormonal gradient, for the pollen tubes to continue growing. This may be the type of feedback mechanism between pollen tubes and ovules that Owens (1992) cites. In addition, pollen tube growth may have stopped on May 15 because some portion of the ovary was producing an inhibitory substance that prevented the pollen tubes from elongating past the stylar juncture, and that this product was removed or neutralized with the approach of megasporogenesis. It may only be when the ovules reach a critical or threshhold stage of development, for instance in the second growing

season, that the proper stimulus for continued pollen tube growth is produced. This is especially evident in these two red oak group species that have only rudimentary ovules in the first growing season. It was observed in this study that the pollen tubes entered the compitum and PPO just as the megaspore mother mother cells were undergoing meiosis and not before. Thus, the putative "signal" for pollen tube growth resumption could have been produced shortly before meiosis was visible.

We have found no evidence of late spring frosts at our study site. Sharp (1958) concluded that low temperatures in the spring did not affect flowering unless there was a freeze sufficient to damage shoots and leaves. Goodrum and others (1971) also concluded that the influence of low temperatures on flowering, setting of fruit and subsequent acorn yield was inconclusive. However, low air temperatures were associated with a delay in the development, but not the number, of pistillate inflorescences of *Q. robur* near Moscow (Minina 1954).

The role of various pollen sources must also be considered as factors in the success of an acorn crop. For instance, NRO is the first oak species to flower at our relatively dry ridgetop site. The fast abortion rate of the pistillate flowers in May (Fig. 1-3) could reflect a lack of available outcrossed NRO pollen or a high degree of self pollination. Oaks are relatively self incompatible (Cottam, Tucker, and Santamour 1982). Since there are few NRO trees in our stand, the chance for cross pollination is reduced. But, there is an increased opportunity for a greater proportion of self pollinated flowers. Schwarzmann and Gerhold (1991) reported that northern red oak is a highly outcrossed species with a low level of self fertilization. Their sudy was done with acorns, the product of successful fertilizations. Our results, to a large extent, deal with flowers that have a high likelihood of aborting. Thus, a comparison may not be completely valid. The black oaks in our study have a slower rate of flower abortion during the first 6-8 weeks, compared to NRO. There are many more trees in the stand, allowing a greater likelihood of cross pollination. Any selfing would occur in less proportion and lead to a higher survival rate of those flowers.

One speculated role for callose plugs is as an incompatibility reaction, either from interspecific pollen or from self pollination (Dumas and Knox 1983). Thus, the callose plugs in these oak pollen tubes could indicate that many of the pollen tubes are from selfing. However, we have occasionally observed callose plugs in pollen tubes that had reached the locules and were apparently ready to fertilize the ovules in the second growing season. We expect that a self-pollinated flower would have aborted early in the first growing season. An alternate hypothesis is that callose plugs are a regulator of turgor pressure in the pollen tube (Dumas and Knox 1983). Callose plugs are formed sequentially as the tubes grow, perhaps allowing turgor pressure to be maintained in the cytoplasm at the tip of the pollen tube and facilitating penetration. Controlled pollination experiments are planned to examine the relationship between various types of crosses and the occurrence of callose plugs in oaks.

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AGE TRENDS IN GENETIC CONTROL OF *JUGLANS NIGRA* L. HEIGHT GROWTH

George Rink and F. H. Kung[1]

Abstract: Age-related trends in narrow-sense and family heritabilities for black walnut height and dbh from a southern Illinois open-pollinated progeny test are evaluated through age 20 years. Narrow-sense heritability for height tends to be relatively stable between ages 10 and 20 at 0.55 - 0.65 with similar patterns and values for family heritabilities for both height and dbh. Individual phenotypic age-age correlation coefficients increased from approximately 0.1 at age 1 to 0.8 at age 8 and remained above this level through age 20.

INTRODUCTION

Genetic gain, the quantitative estimate of the amount of improvement per generation resulting from selection, is the product of heritability and selection intensity. Thus, precise estimates of heritability are needed for calculating gain. Although such estimates are available for juvenile black walnut (Rink 1984, Rink and Clausen 1989), precise estimates of heritability for growth traits from mature trees are not available. Since it is known that heritabilities differ with tree age, changes in these genetic parameters need to be evaluated over the course of a generational rotation. Also, relationships between early and later estimates of heritabilities need to be explored along with phenotypic age-age correlations. The objective of this study is to evaluate heritabilities for black walnut tree height and dbh and age-related growth trends.

METHODS

Open-pollinated seeds collected in 1971 from 54 trees were used to produce 1-0 seedlings for the Pleasant Valley progeny test in southern Illinois. Parent trees were selected from within a 250-mile radius south of the plantation location (Fig. 1), the area most frequently cited as the optimum provenance for obtaining a growth stimulus from geographic seed source selection for artificial regeneration in southern Illinois (Bresnan et al., in press). This included an area bounded by southern Illinois, western Kentucky, western Tennessee, and southeast Kansas. However, most of the seed collections were from west Kentucky and west Tennessee. In most cases seed was collected from one tree per stand. Although trees of better than average form were sought at the time of collection, no rigid minimum selection criteria were applied. Open-pollinated progeny of these trees were planted in 5-tree row plots at a spacing of 1.8 m within rows and 3.7 m between rows, in a randomized complete block design with 10 blocks.

The test was established in early spring 1973 on a Haymond silt loam on the floodplain of Sexton Creek, Alexander County, Illinois. The Haymond loam is characterized as a coarse-silty, mixed, nonacid, mesic Typic Udifluvent classified as suitable for walnut culture (site index 50 at age 70) (Losche et al. 1980). The site was an abandoned fescue pasture on which 1.2 m strips were machine sprayed with a simazine, dalapon, 2,4-D mix before planting, spot sprayed in spring 1974 and 1975, and mowed annually. In addition, the test was rogued (thinned) after the 10th growing season to remove 3 of the smallest trees per 5-tree plot and again after the 14th growing season to retain the most vigorous, best formed tree per family-row plot. The trees were measured for total height 14 times and dbh

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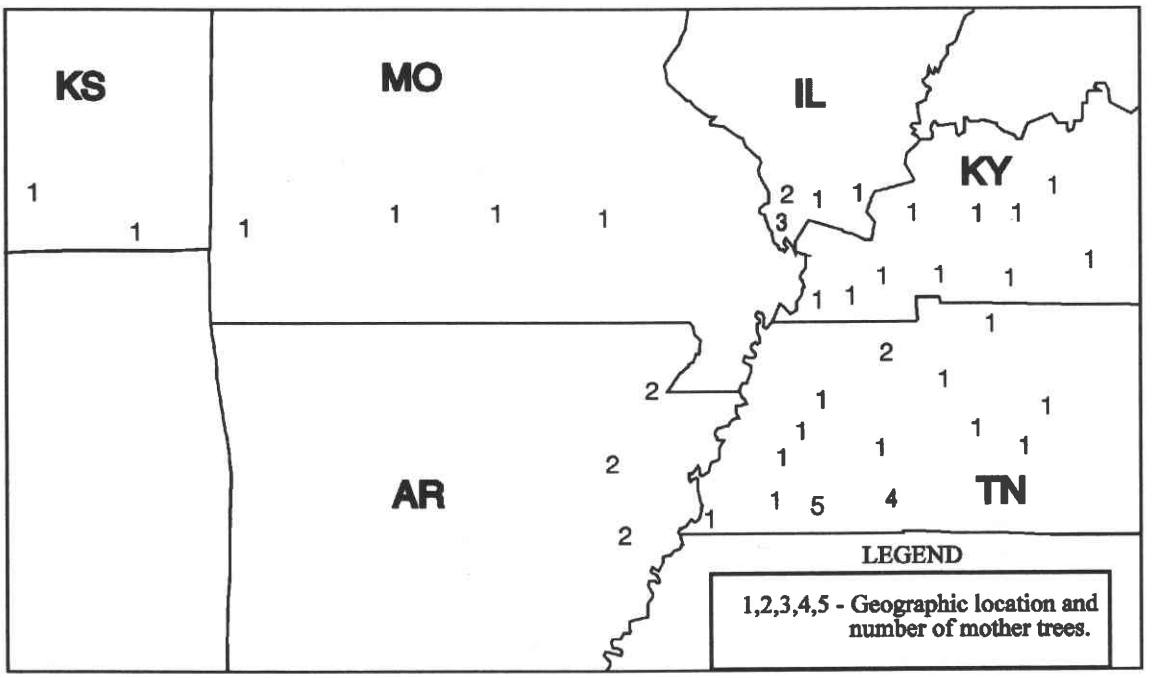

Figure 1. Location of mother trees from which seed were collected for Pleasant Valley open-pollinated progeny test.

11 times during the 20-year period through the 1992 growing season. Statistical analyses were applied to data sets which included only the trees present at age 20 (i.e., the earlier data sets were censored to include only one tree per family-row-plot retained after the last thinning).

Annual measurements were subjected to univariate analysis of variance on an individual tree basis using the following model:

Source of Variation	Mean Square	Expected Mean Square
Blocks (b)	MS_b	$\sigma_e^2 + 0.099\,\sigma_f^2 + 47.277\,\sigma_b^2$
Family (f)	MS_f	$\sigma_e^2 + 9.076\,\sigma_f^2$
Error (e)	MS_e	σ_e^2

Variance components were generated from the Type I sums of squares in the General Linear Model of the Statistical Analysis System (SAS Institute 1982). The Type I method was used in order to obtain an unbiased estimate of the family variance component; by this method all ambiguous variance is assigned to the first variance component entered into the estimation procedure (in this case the block term). Resulting variance components were used to generate narrow-sense and family heritabilities. Pearson correlation analysis was also used to develop phenotypic age-age correlations on an individual tree basis, but only correlations of height at age 20 with height at earlier ages are presented.

RESULTS AND DISCUSSION

Mean cumulative height of black walnut trees in the Pleasant Valley progeny test was contrasted for the ten tallest and ten shortest families (Fig. 2). The group of 10 families which were tallest and shortest at outplanting remained tallest and shortest, respectively, at age 20; the cumulative growth curves never cross. Analyses of variance of the three family groups (10 tallest families, average families, and 10 shortest families) indicated significant differences in height every year beyond the first year, and Duncan's Multiple Range tests consistently separated the three groups of mean heights after age 8. Growth of the 10 tallest families exceeded the plantation average by 8 percent every year beyond age 2. The average height at age 20 was 9 meters.

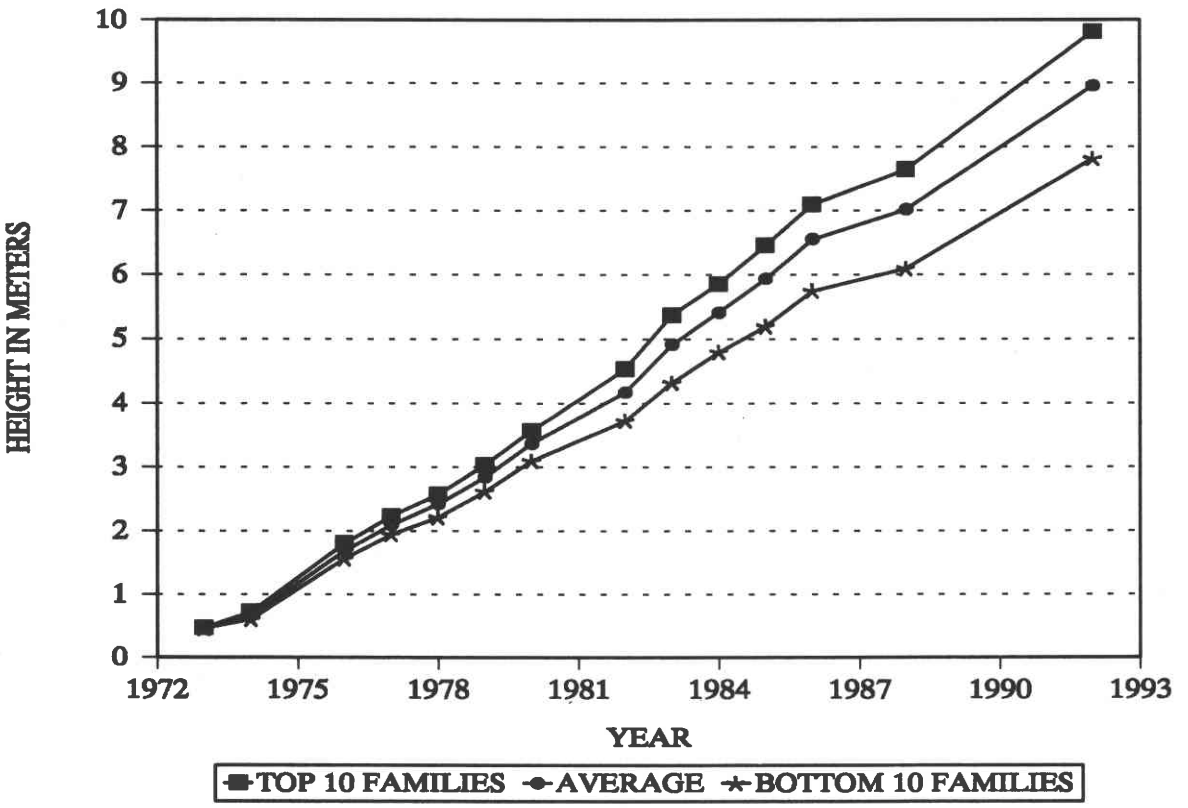

Figure 2. Average cumulative black walnut tree height in Pleasant Valley progeny test, from establishment in in the winter of 1973 through the 20th growing season, including averages of the ten fastest and ten slowest growing families.

The distribution of mean annual height increment was less uniform during this period (Fig. 3). The low height increment during the 1986-1988 period corresponded to a 3-year drought in which growing season precipitation was below average. Similarly, the 1982-1983 high mean annual height growth corresponded to above average precipitation during the 1983 growing season. Differences between the family groups (10 tallest, average and 10 shortest families) were significant during these dry and very wet periods, although significant height increment differences were inconsistent in other years. Patterns of cumulative dbh and mean annual dbh increment were very similar to the patterns displayed for tree height.

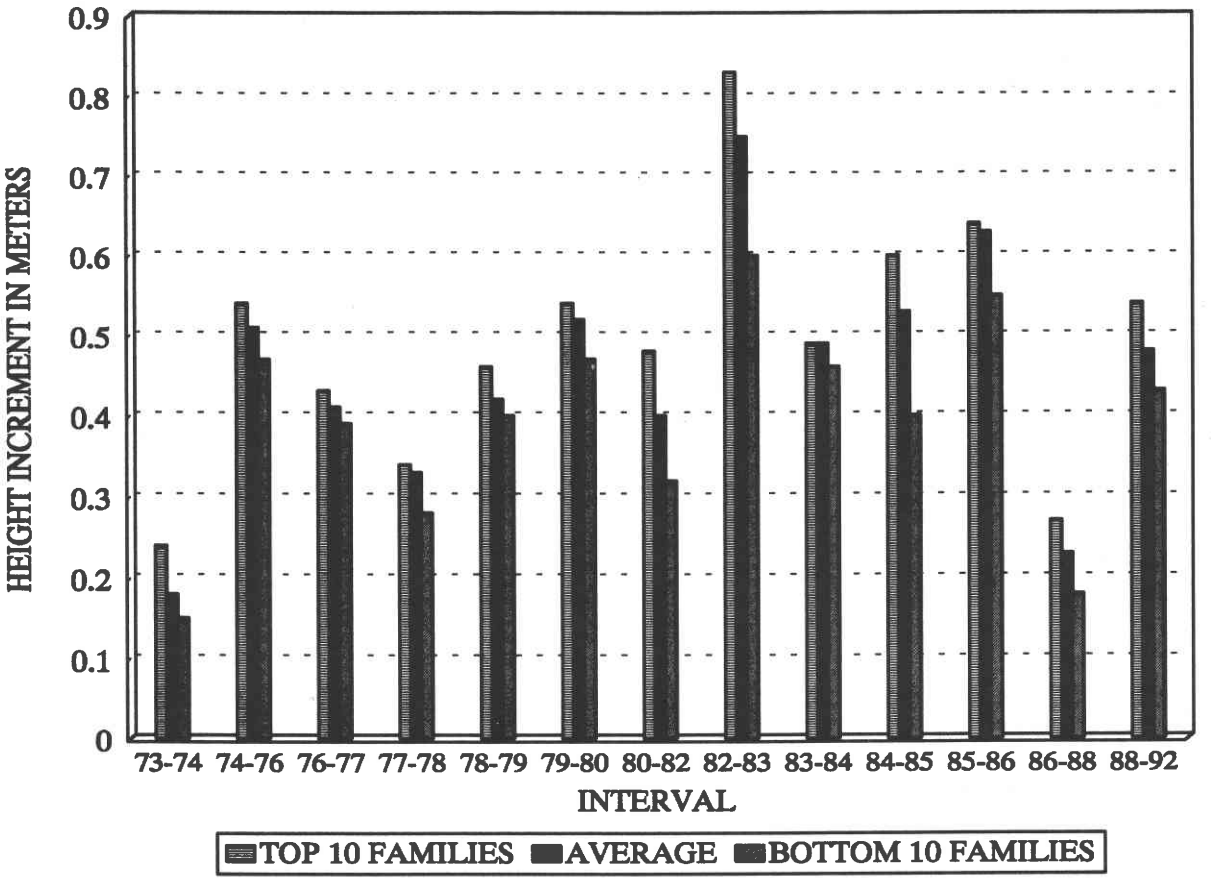

Figure 3. Mean annual height increment in Pleasant Valley black walnut progeny test, including averages of the ten fastest and ten slowest families.

Results from analyses of variance of height and diameter data indicated significance for both block and family effects in all 14 measurement years. Such results are common in experiments with large numbers of families and blocks and reflect an efficient experimental design. Due to missing plots a trivial coefficient (0.099) for the family variance component in the block effect was indicated; this reflected only minor confounding of the block effect.

The absolute magnitude of the family variance components consistently increased throughout the 20-year period of this progeny test (Table 1). However, on a percentage basis, in relation to total variance, the family variance component was at a maximum the year after establishment and decreased consistently during this 20-year period (Table 1). By the end of the 20th growing season, the family component accounted for 2.7 percent of total variance. The trend for family variance components was consistent with that reported by Rink and Clausen (1989) for this experiment through age 13 years and prior to thinning. Block variances, both absolute and as a percentage of total, increased over time, accounting for approximately 80 percent of total variance by the end of the 20th growing season (Table 1). Such a trend of rising block variance components may reflect intensified site occupancy over time by growing root systems of a site-sensitive tree species such as black walnut.

Table 1. Mean height (m) and dbh (cm), variance components, narrow-sense and family heritabilities[1], and respective standard errors for black walnut trees in Pleasant Valley progeny test.

				Variance Components				Heritability[1]			
Yr	Age	Mean	Std	Block	Family	Error	Total	Ind.	SE	Family	SE
						HEIGHT					
73	1	0.48	0.13	0.0002	0.0041	0.0127	0.0170	0.97	0.251	0.74	0.063
74	2	0.66	0.20	0.0032	0.0044	0.0344	0.0420	0.46	0.154	0.54	0.039
76	4	1.69	0.39	0.0280	0.0142	0.1100	0.1523	0.46	0.136	0.54	0.034
77	5	2.10	0.50	0.0812	0.0146	0.1608	0.2566	0.33	0.100	0.45	0.025
78	6	2.43	0.65	0.1696	0.0320	0.2318	0.4333	0.48	0.104	0.56	0.026
79	7	2.86	0.79	0.3057	0.0394	0.3074	0.6525	0.45	0.089	0.54	0.022
80	8	3.38	0.94	0.5353	0.0499	0.3561	0.9413	0.49	0.075	0.56	0.019
82	10	4.19	1.35	1.2477	0.0981	0.6147	1.9605	0.55	0.066	0.59	0.017
83	11	4.93	1.60	1.8299	0.1475	0.7600	2.7374	0.65	0.066	0.64	0.017
84	12	5.41	1.66	2.0336	0.1429	0.7715	2.9481	0.63	0.061	0.63	0.015
85	13	5.94	1.96	3.0129	0.1772	0.9337	4.1238	0.64	0.053	0.63	0.013
86	14	6.57	2.14	3.7270	0.1899	1.0293	4.9463	0.62	0.048	0.63	0.012
88	16	7.03	2.34	4.3719	0.2118	1.3177	5.9014	0.55	0.047	0.59	0.012
92	20	8.98	3.19	8.8437	0.2981	1.9031	11.0449	0.54	0.036	0.59	0.009
						DBH					
77	5	1.73	0.84	0.1431	0.0588	0.5168	0.7187	0.41	0.127	0.51	0.032
78	6	2.33	1.14	0.4594	0.1077	0.7895	1.3566	0.48	0.113	0.55	0.028
79	7	3.14	1.53	1.0079	0.1490	1.2835	2.4405	0.42	0.094	0.51	0.023
80	8	4.17	1.90	1.8367	0.2508	1.6958	3.7833	0.52	0.091	0.57	0.023
82	10	5.65	2.38	3.2980	0.3481	2.3499	5.9960	0.52	0.080	0.57	0.020
83	11	6.69	2.68	4.3814	0.4602	2.7617	7.6034	0.57	0.079	0.60	0.020
84	12	7.67	2.87	5.1486	0.4903	3.1269	8.7658	0.54	0.075	0.59	0.019
85	13	8.51	3.17	6.6255	0.5264	3.5360	10.6880	0.52	0.067	0.57	0.017
86	14	9.34	3.38	7.9461	0.5681	3.7849	12.2992	0.52	0.063	0.58	0.016
88	16	10.53	3.81	9.7916	0.6471	5.0398	15.4785	0.46	0.061	0.54	0.015
92	20	13.46	4.89	17.3099	1.0297	7.2604	25.6001	0.50	0.056	0.56	0.014

[1] Narrow-sense individual tree heritability = $\frac{4\sigma_f^2}{\sigma_e^2+\sigma_f^2}$

Family heritability = $\frac{\sigma_f^2}{\frac{\sigma_e^2}{9.076}+\sigma_f^2}$

Between ages 1 and 5 years, narrow-sense individual tree heritabilities for height declined from 0.97 to 0.33. This was followed by steady increases to a maximum of 0.65 at age 11, remaining relatively stable until age 20 (Table 1, Fig. 4). Family heritabilities for height were slightly higher than for individual tree heritabilities, but the overall trend over time was nearly identical. Heritabilities for diameter at breast height were also similar in magnitude and had similar age trends (Table 1, Fig. 5).

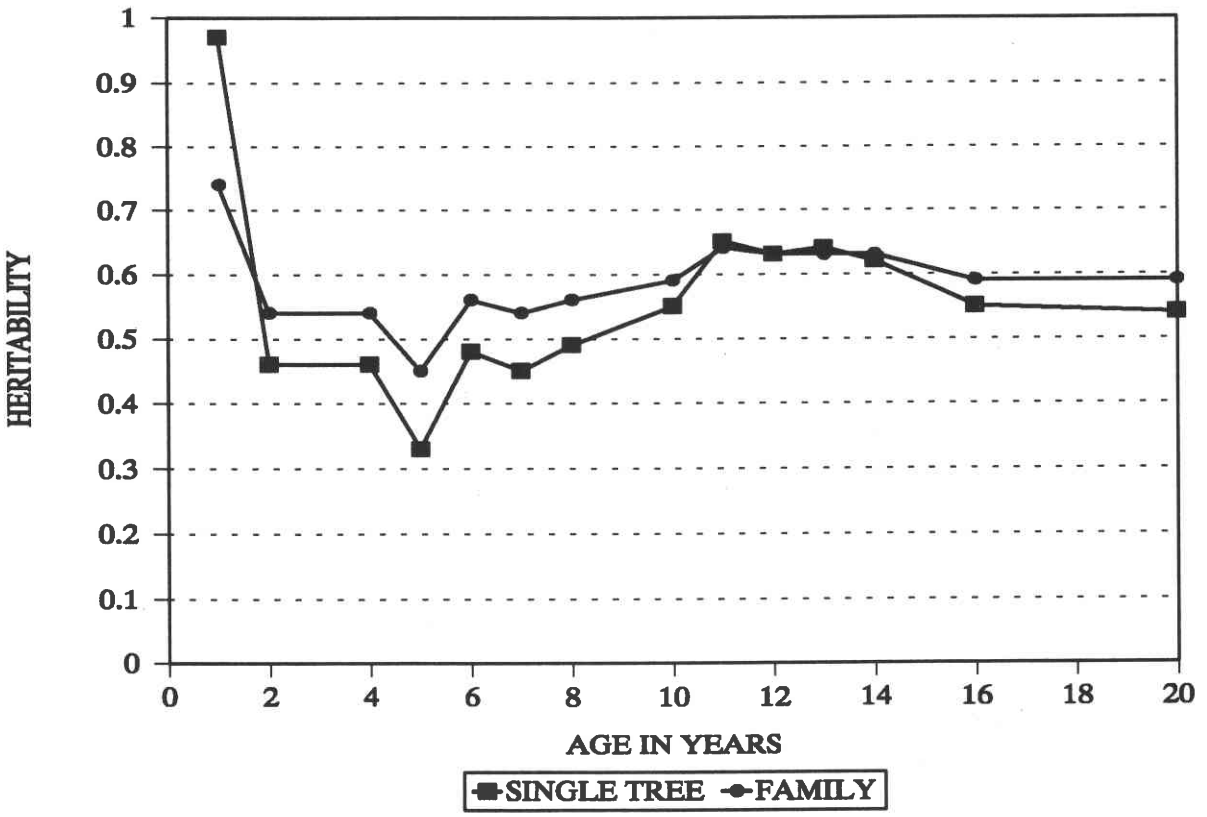

Figure 4. Narrow-sense (single-tree) and family heritability values for height at various ages of the Pleasant Valley black walnut progeny test.

The age-related pattern of heritability for black walnut tree height is similar to that described for North American conifers. Franklin (1979) proposed a hypothetical model with three genetic phases developed on the basis of heritability and additive genetic variance. The model consisted of a juvenile genotypic phase with a peak in heritability in the early to middle part of the phase, a mature genotypic phase with another peak in heritability occurring early in the phase, and a codominance-suppression phase characterized by rapidly declining heritability. The transition between juvenile and mature genotypic phases is characterized by a low in heritability (e.g., in Douglas-fir (*Pseudotsuga menziesii* (Mirb.) France) at age 16, $h^2 < 0.005$) and a mature genotypic phase which extends from age 15 to age 40 (Namkoong et al. 1972).

Although the absolute values for black walnut differ from values for coniferous species, the pattern of heritabilities is similar. Peak heritability was found at age 1 ($h^2 = 0.97$) with a low estimate at age 5 ($h^2 = 0.33$), corresponding to a juvenile phase. Heritabilities gradually increase to a high at age 11 ($h^2 = 0.65$) and diminish slightly to 0.54 by age 20. Although the black walnut heritabilities in this study tend to be substantially higher than those reported for North American conifers, there is a resemblance in the distribution of heritabilities with that of the conifers. Thus, the low

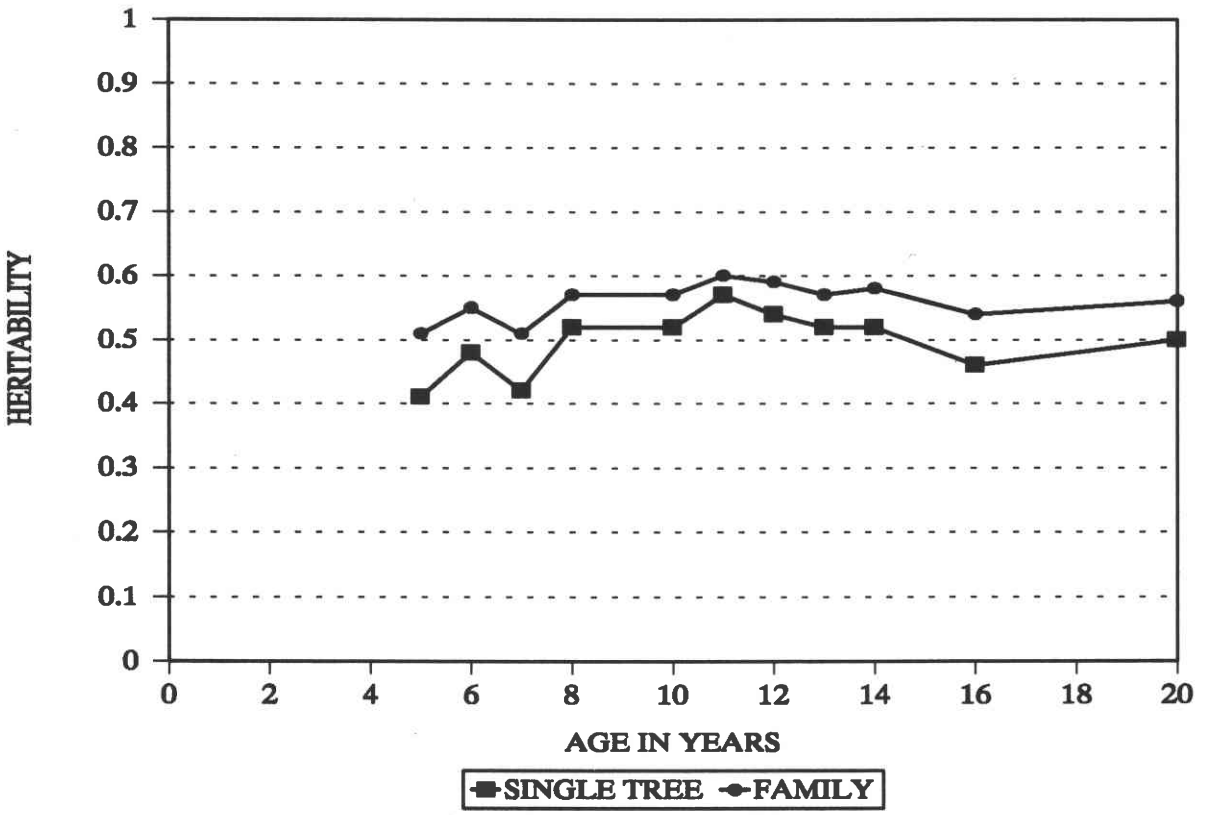

Figure 5. Narrow-sense (single-tree) and family heritability values for dbh at various ages of the black walnut progeny test.

heritability at age 5 may correspond to black walnut crown closure at this age and spacing and apparently occurs earlier than the age 15 in Douglas-fir (Franklin 1979).

It is possible that individual tree heritabilities presented here are overestimated due to potential confounding with a seed source effect (i.e., non-random mating within local seed sources) and a genotype x environment interaction, neither of which could be estimated from these data. Although age 13 genotype x environment interaction estimates presented elsewhere were very low (Rink and Clausen 1989) and would not have a great impact, the provenance effect potentially could have a substantial influence on these heritability estimates if trees represented in this study were in fact from more than one provenance. Thus, Squillace (1974) reported that overestimates of heritability result when data from unrelated families (e.g., such as from distant localities) are used for calculations. Strictly speaking, heritabilities should be estimated using data from open-pollinated families representative of a random-mating population in genetic equilibrium.

The curve of phenotypic age-age correlations between individual tree height in 1992 (at age 20) and individual tree height in earlier years is presented in Figure 6. By 1980 (age 8), the curve of age-age correlations appears to have attained a somewhat horizontal plateau; at this age, the correlation between 1992 height and 1980 height is 0.809, and the individual tree narrow-sense heritability is estimated at 0.49.

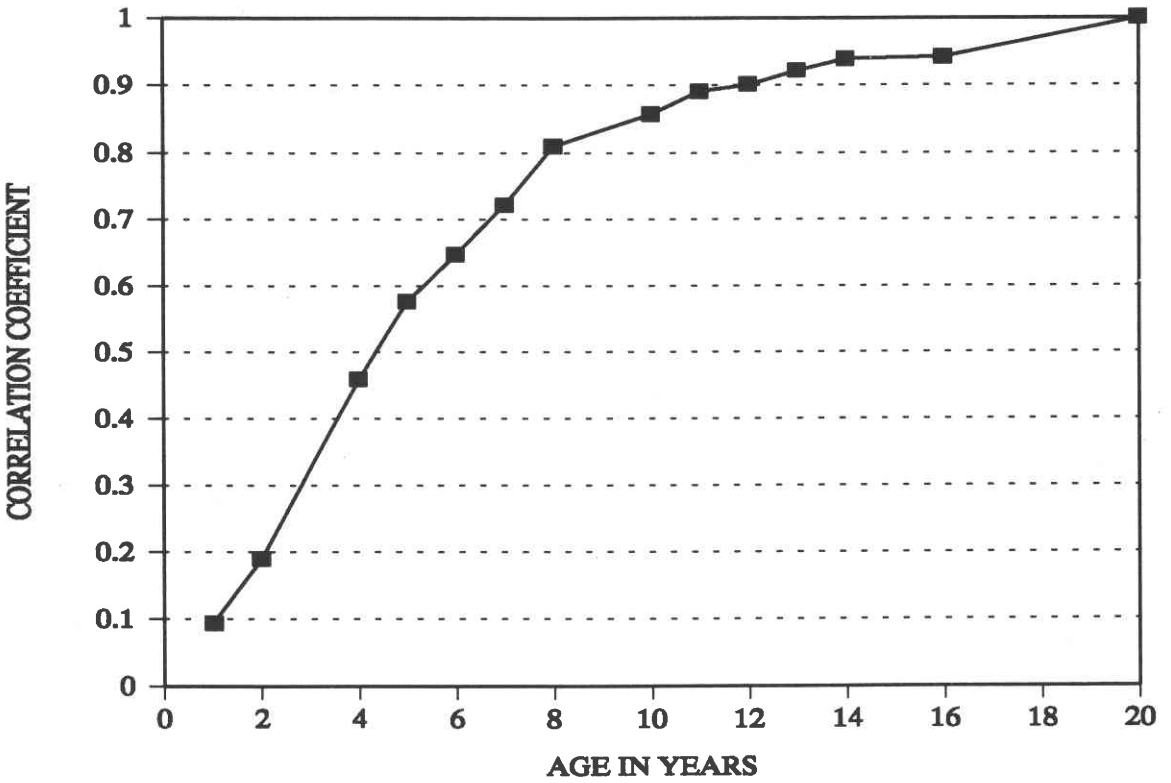

Figure 6. Age-age correlations of height of black walnut trees in 1992 (i.e., age 20) with height in other measurement years, on an individual tree basis.

Although the distribution of these heritability and phenotypic correlation values over time is based on a progeny test on only one site in southern Illinois, it would appear that selection of black walnut for improved height growth could be carried out with a high degree of confidence beyond age 8. Phenotypic correlations appear to have attained a plateau and seem to remain relatively stable. These results support the Indiana Division of Forestry (Coggeshall and Pennington 1982) effort in establishing close-spaced, short-term black walnut progeny tests for the purpose of juvenile testing and selection. Such approaches accelerate progeny testing and increase gain per unit time.

ACKNOWLEDGMENTS

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EFFECTS OF HAYSCENTED FERN DENSITY AND LIGHT ON WHITE ASH SEEDLING GROWTH

Tracy E. Hippensteel[1] and Todd W. Bowersox[2]

Abstract: Communities of hayscented ferns [*Dennstaedtia punctilobula* (Michx.) Moore] are present in many forested areas of Pennsylvania. These ferns can reduce the number and height growth of desirable tree seedlings. A study was conducted to determine the effects of fern frond density on the stem growth and leaf development of bare-root planted white ash (*Fraxinus americana* L.) seedlings. Treatments of 0, 25, 50, and 100% frond density created conditions of 0, 151, 271, and 448 g/m^2 of ovendry frond biomass in August. Amount and quality of sunlight passing through the fronds was measured. Shoot growth of the white ash seedlings was greatest in the 0% frond density (21 cm) and lowest in the 100% frond density (12 cm). A study using simulated fern fronds to alter sunlight conditions where root competition was not a factor was also conducted. White ash seedling stem growth was 23 to 45 cm for frond densities ranging from 80 to 0%. Reduced seedling shoot growth and changes in seedling leaf tissues associated with increased frond density were, in-part, due to changes in sunlight quality and quantity.

INTRODUCTION

In Pennsylvania and in other areas of the Northeast, the presence of hayscented fern [*Dennstaedtia punctilobula* (Michx.) Moore] often results in seedling establishment failure in hardwood stands. Hayscented fern can form a dense, nearly monospecific, ground cover beneath hardwood overstories following reproduction cuts. Dense ground covers of hayscented fern can reduce the number of desirable seedlings by 50-90% and inhibit seedling height growth by 40-60% (Horsley, 1983). When seedling growth is reduced, the seedlings are exposed to deer browsing for a longer period of time, seriously arrested advance regeneration development. Horsley (1993) recently has shown that reduced availability of light was the key factor preventing establishment and growth of black cherry (*Prunus serotina* Ehrh.) in Allegheny hardwood stands. We conducted two field studies to determine the effects of light altered by fern foliage on growth and development of planted white ash (*Fraxinus americana* L.) seedlings.

LITERATURE REVIEW

Competition for Light

Competition for light occurs when one species because of more rapid growth, taller stature, or established presence casts shade on another species, thereby limiting its growth. Competition for light differs in several ways from competition for other resources needed for plant growth. Basically, this difference arises from the nature of the supply of the resources. With water or nutrients there is a reservoir from which all competitors draw. It may be a virtually limitless supply or it may be a fluctuating or inadequate supply, such as occurs when soil moisture alternates between field capacity and wilting point. The situation for light is quite different; there is no reserve. Instead, light energy is instantaneously available and it must be instantaneously intercepted by leaves or it will be lost as a source of energy for photosynthesis. Light cannot accumulate in the environment for later use.

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A vital relationship in competition for light is one of physical position, the position of the foliage relative to the foliage of its competitors. Allocation of carbohydrate to new photosynthetic surface area will result in an exponential increase in growth. Increased growth may allow the development of a plant canopy that will shade a competitor, thereby reducing its growth and ability to exploit available resources. Allocation to below ground structures, although necessary for water and nutrient uptake, increases respiratory demand and decreases carbohydrates available for canopy development.

As one plant overtops another, not only is the quantity of light changed but also the quality. Plant canopies act not only to weaken but to filter sunlight. For plants growing under a vegetative canopy, there are three spectral changes which are of potential physiological significance. First, the quantity of photosynthetically active radiation is drastically reduced; second, a marked reduction in the quantity of radiation in the blue waveband; and third is the strong depletion of the red waveband and relatively weak reduction of the far-red waveband. These spectral changes occur as a result of the selective spectral absorption of leaves (Federer and Tanner, 1966). Light arriving at a green leaf is absorbed selectively, the remainder being scattered (reflected or transmitted). The blue and red (450-500 nm and 650-700 nm, respectively) wavebands are absorbed strongly, the green waveband is less strongly absorbed and the far-red is almost entirely reflected or transmitted (Holmes, 1981). The ratio of red to far-red radiation is expressed by R:FR which is the ratio of the photon fluence rate in 10 nm wide wavebands centered at 660 nm and 730 nm respectively (Holmes, 1981). Red:far-red during daylight, irrespective of time of year and weather conditions, remains remarkably constant, averaging 1.15 + 0.02 (Smith, 1982).

The first person to make accurate measurements of the radiation spectrum on a forest floor was Knuchel (1914) who used a spectrophotometer. Unfortunately his measurements did not extend beyond 650 nm.

Seybold (1936) and Engle (1937) made measurements with glass filters which extended to 720 nm, but the wave bands were not sharply defined. These measurements clearly showed that the light in a forest is relatively rich in far red and near infrared. Only recently have suitable interference filters become available which make satisfactory field measurements possible.

Dirmhirn (1964) made measurements (between 400 nm and 1100 nm) under a group of walnut trees using a monochromator with interference filters. Federer and Tanner (1966) made measurements with a similar instrument under different stands in a wavelength range of 400-740 nm. Goodfellow and Barkham (1974) used an ISCO Spectroradiometer to measure quantity of radiation in the range 380-1100 nm in a beech forest. More recently, the LI-COR Spectroradiometer has become available and is capable of a similar range of readings (300 nm-1100 nm).

As early as 1936, Meischke drew attention to the fact that germination of many seeds is inhibited by far-red light, the wavelength band which is very strongly transmitted by green leaves. He concluded that under natural conditions this inhibition might occur under dense vegetation and showed in an experiment that light filtered through a leaf could inhibit seed germination (Stoutjesdijk,1972).

Smith (1982) found that low R:FR light led to increased petiole length, reduced leaf area, increased stem dry weight, reduced branching, and changes in chlorophyll content. Thus there is strong evidence that plants are able to detect and react to both light quantity and R:FR levels associated with vegetation shade.

A typical spectral photon distribution for daylight between 400 nm and 800 nm at a station at 53°N latitude, under a clear midsummer sky and near midday, is shown in Figure 1 (Holmes and Smith, 1977a). There is some decrease in fluence rate at the blue end, but between 450 nm and 850 nm the photon fluence rate is remarkably uniform, being interrupted only by O_2 and H_2O absorption troughs.

Under a vegetative canopy, Figure 1 (Holmes and Smith, 1977b), highest transmission occurs beyond 700 nm which is maintained to 900 nm or 1100 nm. The transmission curve for the visable region shows a slight minimum being found between 425 nm and 475 nm and at 675 nm, with a maximum at 550 nm. This is explained by the absorption and reflection spectra for deciduous leaves. Within the visible spectrum a certain amount of radiation will be

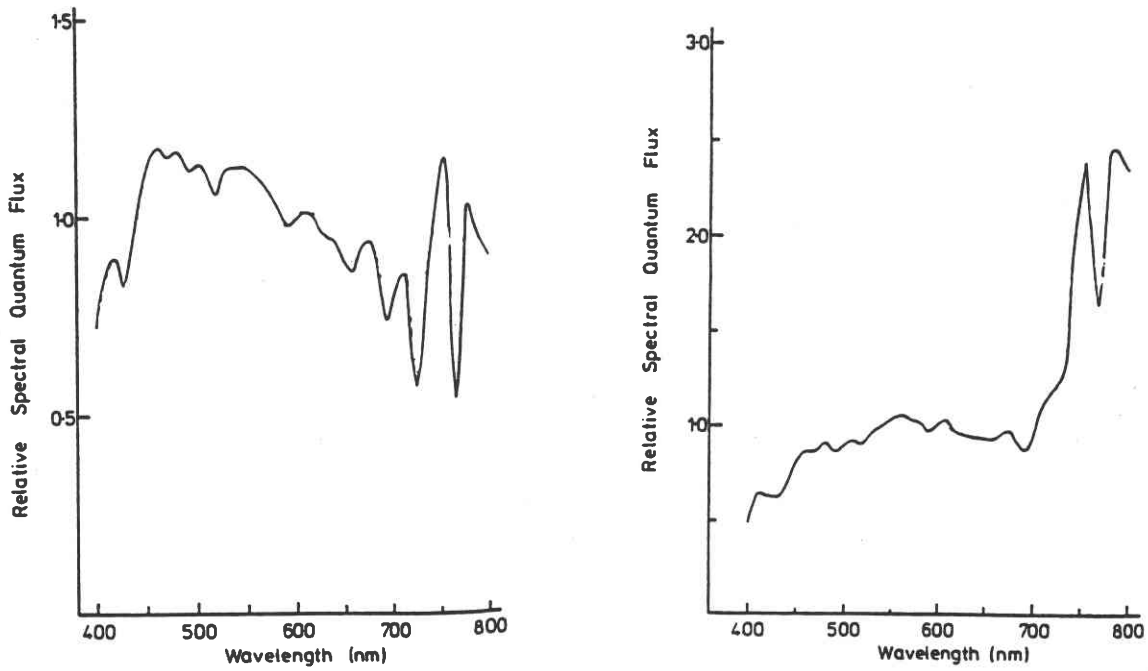

Figure 1. Light quality spectrum under clear skies, left (Holmes and Smith, 1977a); and light quality spectrum under a wheat canopy, right (Holmes and Smith, 1977b).

absorbed in all wavebands, but at 440 nm and 670 nm, the peaks of the photosynthetic action spectra, absorption is strongest. Other wavebands will be subjected to greater amount of transmission and reflection by the canopy. At 550 nm absorption is at a minimum, and the amount of radiation reflected and transmitted will be high. Similarly, beyond 700 nm in the far red and near infrared wavebands, absorption is low and these wavelengths therefore figure prominently under vegetative canopies.

Hayscented Fern

Hayscented fern [*Dennstaedtia punctilobula* (Michx.) Moore] is a native perennial fern of North America with its range centered in the Appalachian region and extending into adjacent parts of eastern Canada and eastern United States (Cody et al., 1977).

It reproduces both sexually and asexually. Sexually it produces spores which are released in late summer. Several million spores may be produced by a single well developed frond (Cody et al., 1977). These minute spores can travel through the air, depending on wind and other conditions, several hundred feet from the mother plant. These spores, given the correct conditions, generally bare, moist mineral soil, in a protected microsite will germinate and eventually produce a new fern plant (Groninger, 1991). Fern spores are known to be capable of germination under a very wide range of light intensities (Page, 1979). Light quality may also be of significance. In the wild, the vast majority of forest fern spores must be able to germinate successfully under low light conditions. The highest temperatures which permit germination are 30-35°C (Page, 1979). Most species of fern succeed between 10-20°C in high humidity and shade (Page, 1979). Once mature, spore production takes place regularly from year to year, showing very little

fluctuation with annual variations in climate such as those that may produce good and bad seed years in Angiosperms (Page, 1979).

Asexually, ferns can reproduce by an extensive rhizome or root system which generally occurs in the mineral and organic soil layers generally 5-15 cm below the surface (Cody et al., 1977). The rhizome system produces frond buds which may remain dormant for one or two years and then develop into mature fern fronds. Two to five fronds is the common annual growth from each branch of the rhizome (Cody et al., 1977). The rhizome overwinters and produces new fronds in the spring and throughout the growing season. Where a single plant can spread its rhizomes and occupy and entire area, a frond density as high as 55,000 per hectare may occur (Page, 1979).

Horsley (1984), in a study comparing fern rhizome development in uncut and thinned Allegheny hardwood stands, found that rhizomes from thinned stands elongated more, and produced more nodes between forks than those from uncut stands. In the uncut stands 10-15% of the frond petioles bore rhizomes buds and these did not grow into new rhizomes unless the parent rhizome was injured. In the thinned stands, 55% of the frond petioles had rhizome buds and many of these grew into rhizomes the same year they were formed. This can possibly explain its rapid expansion in thinned or cleared areas.

There are three main stages in the annual development of the aerial part of the fern plant. The first stage is the fiddlehead stage which is characterized by a short straight rachis with a small curled top. This stage is visible from mid-April to mid-May. The second stage is the frond stage and is characterized by an increase in growth of the fiddlehead and the uncurling of the fern fronds to their full size. This stage occurs between mid-May and mid-September. The last stage is the dead frond stage. This stage is characterized by frond senescence, leaving the dried rachis which generally lies down on the soil and may persist for several years before degrading.

The only known method of totally eliminating hayscented fern is through the use of herbicides. The herbicides glyphosate (Roundup) and sulfometuron (Oust) are used in the northern hardwood forests of Pennsylvania to control hayscented fern (Groninger, 1991). Glyphosate must be applied to photosynthesizing foliage and has no soil residual activity. Sulfometuron, though, applied to exposed mineral soil may prevent the establishment of hayscented fern through spore germination (Groninger, 1991).

White Ash

Fraxinus americana (L.) grows most commonly on fertile soils with high nitrogen content and moderate to high calcium content. The absence of nitrogen reduces seedling dry weight by 38% (Schlesinger, 1990). Soil moisture is an important factor affecting local distribution. Natural regeneration from seeds will occur if the soil, humus, or leaf litter is wet in the spring. Best growth occurs in moderately well-drained soil. Tolerance to pH varies from 5.0 to 7.5 (Schlesinger, 1990). Seedlings are capable of withstanding protracted periods of soil drought and are intermediately tolerant to temporary flooding (Rink and Sambeek, 1987).

Young white ash exhibit strong apical dominance. Open-grown seedlings 2 m tall often have only two or three pairs of lateral branches. If the terminal bud is removed, apical dominance is altered and new branches develop from the uppermost pair of lateral buds. Generally one of these grows faster than the other and assumes dominance.

Seedlings develop best in 45% full sun or greater (Logan, 1973). Therefore, silvicultural systems that provide sunlight, such as shelterwood or clearcutting, have been recommended for white ash. When seedlings are young, they are shade tolerant but with age they become more intolerant. Seedlings are capable of survival under a canopy with less than 3% full sun, but they show very little growth at these light levels (Schlesinger, 1990). Under good conditions, they complete 90% of their height growth in 30 days and complete 100% in 60 days. Diameter growth continues until August (Schlesinger, 1990).

Despite its low shade tolerance, white ash is characteristic of intermediate as well as early stages of plant succession. Throughout its range it is a minor but constant component of both the understory and overstory of mature forests on

suitable soils. It owes its position in the final overstory to its ability to persist for a few years in moderately dense shade and to respond quickly to openings in the canopy.

White ash was chosen for the studies based on its behavior in a study by Kolb et al. (1990). They studied the influence of light intensity on weed-induced stresses of tree seedlings (white ash, red oak, and white pine). They found that white ash produced the greatest height growth in interference-free environments but also showed the greatest reduction in growth in a fern environment. These results suggest that white ash has a lower tolerance to herbaceous interference than either red oak or white pine. For this reason white ash was chosen as the study species.

METHODS

The study site was located in the Rothrock State Forest, Huntingdon County, PA, in the Ridge and Valley Province of central Pennsylvania. The area chosen had a southeastern aspect with soils derived from sandstones. The soils were mapped as a Hazelton-DeKalb association having moderately rapid permeability, low to moderate water availability, severe erosion hazard, and medium runoff (Soil Survey of Huntingdon County, PA). Soil pH averaged 4.6 for the site. A shelterwood cut was made in the late 1960's prior to clearcutting during the 1984 growing season. The site index prior to any cutting activity was approximately 70 for oak. Oak species dominated the stand. The site now consists of 80-100% fern and grass communities.

Four areas measuring 8 m by 8 m were selected which contained a dense fern population. The 8 m by 8 m areas were divided into 4 m by 4 m sections with each section representing a fern treatment.

Four areas measuring 2 m by 8 m were selected adjacent to the fern treatment areas and were herbicided with glyphosate (Roundup) at a rate of 1.1 ai/ha one year prior to study initiation. The 2 m by 8 m areas were divided into four 2 m by 2 m sections with each section representing a shade treatment. All areas were cleared of brush and logging debris in the spring of 1991.

On April 19, 1991 one year old bare root, white ash (*Fraxinus americana* L.) seedlings obtained from the Pennsylvania Bureau of Forestry's Penn Nursery at Potters Mills, PA, were planted randomly in the designated areas. White ash was selected for this study because it had a substantial response to ferns in an earlier study (Kolb et al., 1990). The seedlings were top clipped to 10 cm and root clipped to 15 cm before planting. Each fern treatment received 24 seedlings with either one or two seedlings per meter square, a total of 384 seedlings for the fern treatment areas. Each shade treatment received 5 seedlings in each replication for a total of 80 seedlings for the shade treatment areas.

Each treatment in the fern plots was randomly assigned one of four density treatments; 100% fern, 50% fern, 25% fern, and 0% fern. The treatments assigned 100% fern were left untouched with the ferns being allowed to grow to their natural density. The 50% fern treatments were marked off with parallel lines every 0.25 m. The 25% fern treatments were marked off by a combination of parallel and perpendicular lines every 0.25 m.

Shade frames were built over the shade plots with each replication receiving four different shade treatments: 0%, 30%, 60%, and 80% shade. White PVC pipe was used as a frame to support black polypropylene shade cloth 1.2 m above the ground. The shade cloth also extended halfway down the side of the frame to eliminate large amounts of sidelighting but to allow for air circulation under the frames. A four-foot-high wire mesh fence was erected around each area to exclude deer.

Four gypsum moisture blocks and two copper-constantan thermocouples were installed at random locations at 7 cm depth in each fern treatment areas. Two gypsum blocks and one thermocouple were installed at random locations in each of the shade treatments. Soil moisture readings from the gypsum blocks and soil temperature readings from the thermocouples were taken once a week beginning in June and continuing throughout the study. The readings were taken between 11:00 AM and 1:00 PM.

Seedling heights from root collar to top of stem and stem diameters were measured after all treatment preparations had been made. Seedling growth (shoot growth from site of initiation on original stem) was measured every six weeks until mid-September. Any herbaceous vegetation which grew in the shade plots during the study was removed.

The fern treatments were clipped after the fern fronds had grown to about 50% of full size (May 25, 1991), but before bud break in the seedlings. In areas designated 0% fern, all of the fern fronds were cut off at the soil surface. In treatments designated 25% fern, every other 0.25 m wide strip was cut in both directions leaving 0.25 m^2 per 1.0 m^2 patch of fern. In areas designated 50% fern every other 0.25 m wide strip was cut leaving alternating strips of 0.5 m^2 fern and 0.5 m^2 of non-fern. In treatments designated as 100% fern, no ferns were cut. Since the seedlings were located randomly in the plots, some occurred in the fern strips whereas others occurred in the cut strips. Two 1.0 m^2 samples of fern fronds were collected from each treatment in each replication for a total of 8 samples for each treatment. The dry weight of the ferns per 1.0 m^2 were measured for each treatment and are shown in Table 1. Several days after the cutting treatments were completed the fern fronds leaned over and filled in the non-fern spaces in a somewhat uniform manner with the fronds facing in all directions. The areas were re-clipped once a month to maintain the cut strips.

Table 1. Average fern biomass per square meter ± SE as compared to 100% fern treatment. Ferns were collected from two 1.0 m^2 areas in each treatment of each replication for a total of 8 samples for each treatment. The ferns were then oven dried, weighed, and averaged among treatments.

Treatment	Dry Weight (g)	% Biomass
100% FERN	448 ± 28.1 g/m^2	100%
50% FERN	271 ± 18.2 g/m^2	60%
25% FERN	151 ± 21.9 g/m^2	30%
0% FERN	0 g/m^2	0%

Light quantity readings were taken under all treatments on July 10, 1991 with a LI-COR 191SA Line Quantum Sensor, Lincoln, NE. Two light quantity readings were taken in each treatment of each replication for a total of 8 readings per treatment. The line sensor is 1 m long and approximately 5 cm in height. The sensor was placed under the ferns in the fern treatments perpendicular to the cut strips so that fern/no fern strips were uniformly distributed across the sensor. The sensor was placed in the center of the shade treatments for the readings.

Light quality readings were taken randomly under each fern treatment with a LI-COR 1800 Spectroradiometer Lincoln, NE. Light quality readings were taken on two consecutive days (September 7th and 8th, 1991) with two readings per treatment per replication for a total of eight measurements for each fern treatment and one reading was taken in each treatment of each replication for a total of 4 readings for each shade treatment. The sensor was located approximately 18 cm off the ground and was approximately 4 cm in diameter. Readings were taken in both fern and cut areas of the fern treatments to try to get an average. All light readings were taken within two hours of midday in order to eliminate variations in light intensity and quality due to rapidly changing zenith angle of the sun. Readings were taken in reference to the horizontal surface under clear-sky, calm weather conditions.

The study was conducted for a single growing season and seedlings were harvested at the soil surface on September 17, 1991. Shoot growth was measured at this time and the leaves from each plant were removed and leaf area

measured with a LI-COR Area Meter Lincoln, NE. The stem and leaves were then dried in a drying oven for 5 days and then weighed to obtain dry weights.

The data were treated as a randomized block experiment and analysis of variance (ANOVA) was run on the following parameters to test for significant treatment effects; shoot growth, dry weight of the stem, number of leaves, leaf area per plant, area per leaf, dry weight of the leaves, dry weight per leaf, and specific leaf area. Root length was not measured because the hardness of the soil due to lack of rain did not permit the extraction of the entire root system. Mean separations were performed on those parameters which showed significant treatment effects with Tukey's Method of Multiple Comparisons (0.05 was the level of alpha considered significant).

RESULTS AND DISCUSSION

Fern Treatments

Light Quantity. The range of light quantity readings, their averages and percentages are shown in Table 2. Light quantity readings taken in full sun were the basis for comparison. One would not expect this much variability in the ranges of light quantity as the Line Quantum Sensor is designed to reduce variability. The wide range in values for each treatment could be due to the variability of the fern populations and the nature of the treatments (fern/no fern strips). Another explanation is that in some areas the ferns were thicker (more fern fronds per unit area) than in other areas. Also the fern appeared more 'transparent' in some areas than in others, so more light may get through the more 'transparent' ferns.

Table 2. Light quantity ranges, averages, and percentages in the four fern treatments. The measurements were taken with a LI-COR Line Quantum Sensor under the ferns. Values are in *u*mol. The shade percentages were calculated relative to full sun (0% fern). The measurements were made within two hour of mid-day on July 10, 1991, with clear-sky conditions.

TREATMENT	RANGE (*u*mol)	AVERAGE	% SHADE	RANGE
0% FERN	1570-1985	1778	0%	-----
25% FERN	201-1413	545	70%	21-89%
50% FERN	136-781	277	84%	56-92%
100% FERN	5-162	53	97%	91-100%

Light Quality . Plants need light in the red and blue region (650-700 nm and 450-500 nm respectively) of the spectrum to carry out photosynthesis. The R:FR ratios (ratio of the photon fluence rate in 10 nm bands centered at 660 nm and 730 nm respectively) showed the reduction in red light as the fern densities increased. In the open the ratio was 1.13, under 25% fern the ratio was 0.84, under 50% fern was the ratio was 0.31, and under 100% fern the ratio was 0.20. The averaged light quality measurements are shown in Figure 2. The graph shows not only a reduction in light quantity but a large change in light quality in the wavelength range from 300-700 nm for the 100% fern and 50% fern treatments. This graph is similar to Figure 1 shown previously. So light penetrating a fern canopy is almost entirely far-red in quality, unusable by plants.

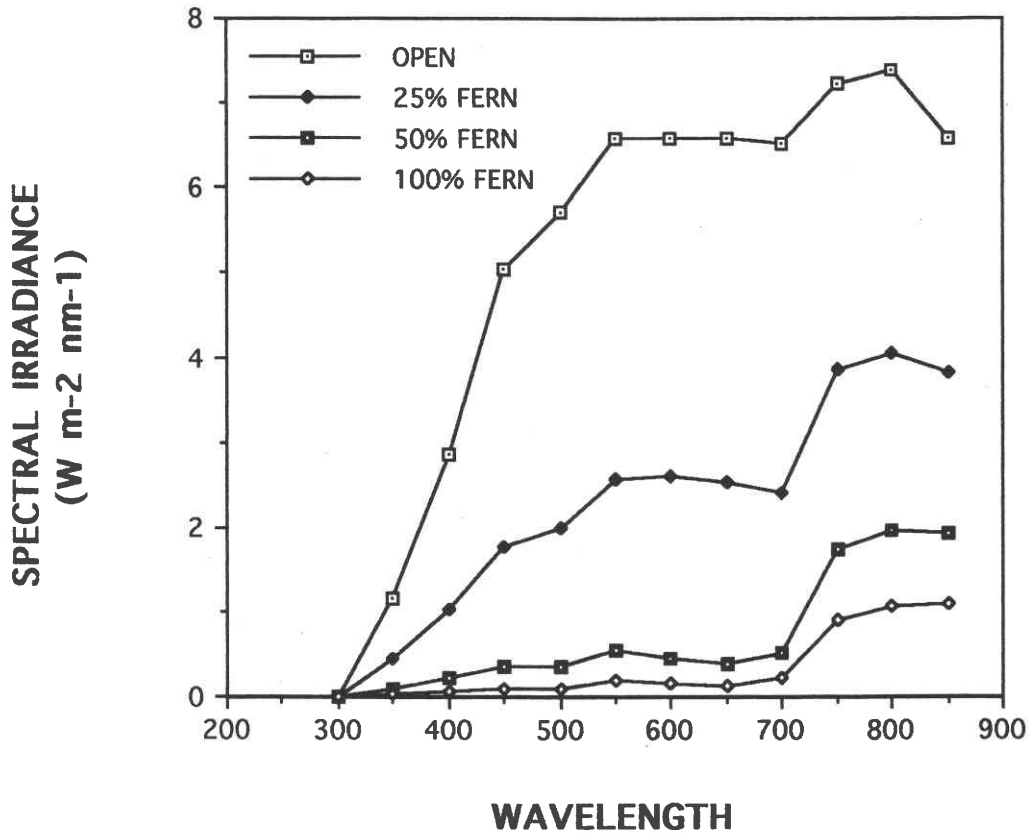

Figure 2. Average spectral irradiance values for the four fern treatments by wavelength. The measurements were made with a LI-COR Spectroradiometer. Each point is an average of 8 measurements.

Soil Moisture. Measurements taken with the gypsum blocks in all four treatments showed no significant difference between the treatments. Soil moisture fluctuated by month with rainfall events as recorded by The Agricultural Research Center at Rock Springs located approximately 12 km northwest of the study site. In general percent soil moisture increased after rainfall events as would be expected. Overall the soil moisture percentages seem much higher than would be expected in the field. The reason for this is unknown. Lower readings would be expected because of the low amounts of rainfall received in the 1991 growing season.

Soil Temperature. Soil temperature measurements showed that 0% fern treatments had significantly higher average soil temperatures at 7 cm depth than the 100% fern treatments. This difference averaged 2.5°C. The other treatments did not differ significantly. Kolb et al. (1989) found soil temperatures to be 4°C greater in fern free environments. In general, the 0% fern treatments had higher soil temperatures than the average air temperature for that particular day and 100% fern had lower average soil temperatures than the average air temperature for that day. The soil temperature difference between the 0% fern and 100% fern would be expected because there is less radiation reaching the soil surface beneath the ferns and so less soil heating.

Shoot Responses to the Treatments. Seedlings growing in the 0% fern treatment were significantly taller than all the other treatments averaging 20.6 cm of shoot growth (tallest stem measured from site of growth on initial stem) Table 3. Shoot growth of the seedlings growing in 25% fern, 50% fern, and 100% fern were not significantly different from each other. The average shoot growth was 12 cm in these treatments. So shoot growth was reduced

Table 3. Growth responses of white ash seedlings to different levels of hayscented fern cover. Means ± SE followed by the same letter are not significantly different at the P<0.05 level (n=96).

	TREATMENT			
RESPONSE	0% FERN	25% FERN	50% FERN	100%FERN
Shoot growth (cm)	20.6a ± 1.2	13.0b ± 0.60	11.3b ± 0.50	11.8b ± 0.60
Dry weight stem (g)	2.0a ± 0.27	0.65b ± 0.05	0.40b ± 0.03	0.33b ± 0.03
# Leaves/plant	16a ± 0.68	11b ± 1.08	8c ± 0.30	8c ± 0.33
Leaf area (cm^2)	503a ± 48.5	213b ± 13.3	156b ± 8.6	144b ± 9.3
Area/leaf (cm^2/leaf)	28.5a ± 1.5	19.9b ± 0.95	19.6b ± 0.89	18.8b ± 0.93
Dry weight leaves (g)	3.1a ± 0.30	1.0b ± 0.07	0.7b ± 0.04	0.5b ± 0.03
Dry weight/leaf (g/leaf)	0.18a ± 0.01	0.1b ± 0.005	0.08c ± 0.004	0.07c ± 0.003
Specific leaf area (cm^2/g)	167d ± 2.7	213c ± 3.9	241b ± 4.2	284a ± 6.1

40% in the presence of any amount of fern. Others have studied seedling height growth under fern and have also found that height growth is severely inhibited by the presence of ferns (Drew, 1988; Horsley and Marquis, 1983; Kolb et al., 1990).

Reasons for the reduced growth of seedlings under fern canopies have been proposed and studied by some authors. Allelopathic studies have been done on hayscented fern by Horsley (1977). He demonstrated that black cherry seedlings watered with nutrient solutions containing foliage extracts from hayscented fern reduced their growth. In future studies though, he dismisses the allelopathic effects of hayscented fern (Horsley, 1986).

Drew (1988) in his paper on the interference of seedling growth by hayscented fern and whorled aster suggests competition for soil nitrogen and forest microsite as the possible cause of reduced growth under the ferns and also suggests the reduced light environment under the ferns as a cause.

Stem (new growth taken from site of initiation on initial stem) dry weights showed results similar to shoot growth with the 0% fern treatment producing the most stem biomass; averaging 2.0 g (Table 3). The three fern treatments did not differ significantly; averaging 0.5 g per stem.

Leaf Responses to the Treatments. The seedlings growing free from above ground competition (0% fern) had larger leaves than the trees with above ground competition (Table 3). The 0% fern treatment averaged 28.5 cm^2/leaf and the 25% fern, 50% fern, and 100% fern did not differ and averaged 19.4 cm^2/leaf.

The largest dry weight per leaf averaged 0.18 g/leaf and was measured in the 0% fern treatment. This was significantly greater than all other treatments. The 25% fern treatment (0.10 g/leaf) differed significantly from 100% fern treatment (0.07 g/leaf), but did not differ from the 50% fern treatment (0.08 g/leaf). The 50% fern treatment dry weight per leaf did not differ from that of the 100% fern treatment.

Specific leaf area (SLA) is a measurement of area per unit weight and gives an indication of leaf expansion. Generally SLA increases in low light as a result of the formation of thinner leaves with a higher water content (Corre, 1983; Morgan and Smith, 1981; Nygren and Kellomaki, 1983). The results of this study produced similar results with all treatments being significantly different. The seedlings in the 0% fern treatment with the lowest SLA averaging 167 cm^2/g and seedlings in the 100% fern treatment with the highest SLA averaging 284 cm^2/g.

One reason for the general reduced shoot growth under the ferns was both a reduction in the amount and a change in quality of light passing through the ferns. The amount of light was reduced to 3-30% of full sun in the presence of 25-100% ferns. From Figure 2, it is clear that the R/FR ratio has changed significantly. Studies show that white ash is capable of survival but not growth under a canopy with less than 3% full sun, but they show very little growth under these circumstances (Schlesinger, 1990). In this study light varied from 100% sun to almost 0% sun and the seedlings survived, but showed very little growth. Only the above ground portion of the fern plant was manipulated, leaving the entire rhizome system intact. Perhaps the rhizomes have more of an influence on seedling growth than the fronds, since the rhizome system remained constant in all the treatments.

Shade Frame Treatments

Light Quantity. The light quantity readings taken under the shade treatments were consistent with the shade ratings of the nylon shade cloth used (30%, 60%, 80%), varying only 2-3% from the rating.

Light Quality. Light quality readings under the shade treatments (Figure 3) show the reduction in quantity but also show that the general shape of the spectra remained constant. Wavelength quantities were proportionately reduced, but all wavelengths remained in the spectrum. There was no reduction in the proportion of red light to far red light. The R:FR ratios under the shade treatments remained constant averaging 1.10 which is very similar to readings taken by Smith (1982) whose R:FR ratio in daylight averaged 1.15.

Soil Moisture. Measurements taken with the gypsum blocks in all four treatments showed no significant difference among them. Overall soil moisture remained fairly high throughout the study despite the low amounts of rainfall. Again soil moisture percentages seem very high for unknown reasons.

Soil Temperature. Soil temperature measurements showed that 0% shade treatments had significantly higher soil temperatures than all the other treatments. The 30%, 60%, and 80% shade treatments did not differ significantly. There was an average of 3°C difference between 0% shade and 30% shade, and average of 3.5° C between 0% shade and 60% shade, and and average of 4.5° C between 0% shade and 80% shade. In general, 0% shade treatment produced the highest soil temperatures which averaged above the average air temperature. Generally, the 30%, 60%, and 80% shade treatments averaged below the average air temperature.

Since the shade cloth produces an artificial environment some of its affects on the environment beneath it needs to be discussed. First there is reduced air circulation and so less evaporation of soil moisture. Second, there is less radiation reaching the soil surface and so cooler temperatures both beneath the cloth and in the soil. Rainfall amounts reaching beneath the shade cloth were probably different for all treatments since the shade cloth trends to intercept rainfall (different rainfall amounts were intercepted depending on the density of the shadecloth) and direct it off of the plots, running down the sides of the frames. Despite this redirection of rainfall, it seemed definitely damper under the shade cloth than in the open. This is probably due to the lack of air circulation and radiation under the cloth which would increase evaporation.

The shade cloth environments would compare to the different levels of shade cover in the following ways; first, under the ferns there would also be reduced air circulation, especially in the 100% fern. Second, the ferns also intercept radiation and rainfall much in the same manner as the shade cloths, with different levels of fern intercepting different amounts of each. So although different in composition and appearance, the shade cloth and the fern cover do produce similar results in the way of reducing air circulation, reducing radiation, and intercepting rainfall. The main difference being the selective spectral absorbance of the ferns.

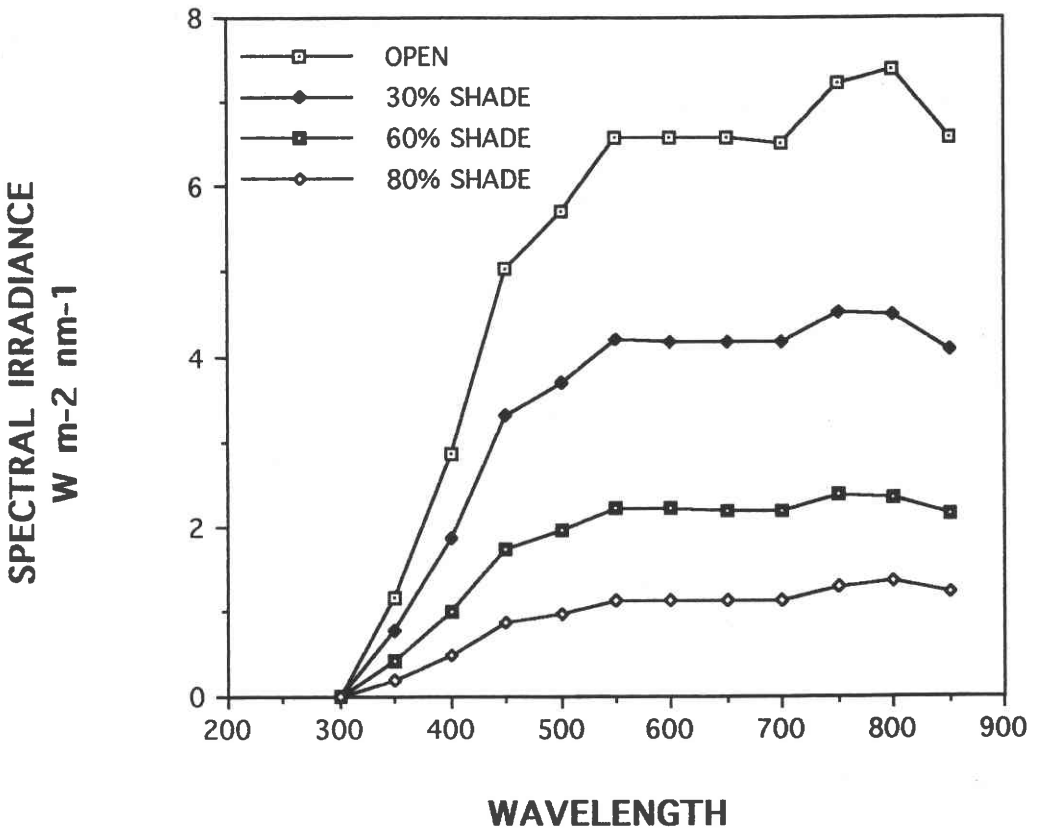

Figure 3. Average spectral irradiance values for the four shade treatments by wavelength. The measurements were made with a LI-COR Spectroradiometer. Each point is an average of 4 measurements.

Shoot Responses to the Treatments. Analysis of shoot growth (Table 4) shows that seedlings in the 30% shade treatment produced the best height growth averaging 44.7 cm in height (tallest stem). The 30% and 0% shade treatments did not differ significantly though and average growth between the two was 42.2 cm. The 0% shade treatment did not differ from the 60% shade treatment which averaged 34.4 cm in height. The 60% shade treatment did not differ from the 80% shade treatment averaging 26.0 cm. Schlesinger (1990) found that white ash seedlings grow best in 45% full sun.

The dry weight of the stems show that 0% shade, 30% shade, and 60% shade were not significantly different averaging 6.8 g. The 60% shade treatment did not differ from the 80% shade treatment and averaged 3.1 g.

Leaf Responses to the Treatments. The number of leaves produced did not differ significantly between 0% shade, 30% shade, and 60% shade and averaged 23 leaves per plant. The 60% shade treatment and the 80% shade treatment did not differ and averaged 19 leaves per plant (Table 4).

Total leaf area per plant exhibited a similar trend as the number of leaves per plant. Generally the more leaves a plant has, the greater the total leaf area. The 0%, 30%, and 60% shade treatments did not differ significantly and averaged 1331 cm^2. The 60% shade treatment and the 80% shade treatment did not differ and averaged 875 cm^2 (Table 4).

Table 4. Growth responses of white ash seedlings to varying shade treatments. Means ± SE followed by the same letter are not significantly different at the P<0.05 level (n=20).

	TREATMENT			
RESPONSE	0% SHADE	30% SHADE	60% SHADE	80%SHADE
Shoot growth (cm)	39.7 ab ± 3.8	44.7a ± 3.3	29.2cb ± 3.2	22.9c ± 3.4
Dry weight stem(g)	7.9a ± 1.5	8.5a ± 2.1	4.0ab ± 0.78	2.3b ± 0.44
# Leaves/plant	24ab ± 1.9	26a ± 2.1	21ab ± 1.7	18b ± 1.4
Leaf area (cm^2)	1468a ± 252	1498a ± 227	1037ab ± 142	713b ± 112
Area/leaf (cm^2/leaf)	55.4ab ± 6.0	59.4a ± 6.7	51.1ab ± 6.3	37.0b ± 3.9
Dry weight leaves (g)	8.9a ± 1.5	8.3a ± 1.7	5.2ab ± 0.83	2.9b ± 0.45
Dry weight/leaf (g/leaf)	0.34a ± 0.04	0.33a ± 0.05	0.23ab ± 0.02	0.15b ± 0.02
Specific leaf area (cm^2/g)	165a ± 11.0	215a ± 28.0	284a ± 89.0	252a ± 8.5

Area per leaf measurements show the seedlings in the 30% shade treatment to have the largest leaves with 59.4 cm^2/leaf average. The 30% shade treatment though did not differ from the 0% shade treatment or the 60% shade treatments. The 0% shade, 30% shade, and 60% shade treatments averaged 55.3 cm^2/leaf. The 80% shade treatment though did not differ significantly from 0% shade or 60% shade either averaging 47.8 cm^2/leaf overall.

Dry weight per leaf shows that 0% shade, 30% shade, and 60% shade did not differ averaging 0.30 g/leaf. Again, 60% shade does not differ from 80% shade averaging 0.19 g/leaf.

Specific leaf area (SLA) surprisingly did not differ significantly between treatments but the expected trend is shown in the results (Table 4). Plants growing in low light intensity have a higher SLA than plants growing in a higher light intensity (Corre, 1983; Nygren and Kellomaki, 1983). In this study 60% shade had the highest SLA averaging 252 cm^2/g and 0% shade had the lowest average of 165 cm^2/g. Why 80% shade treatment did not produce the highest SLA is unknown. One possible explanation is that 80% shade is past the threshold for shade tolerance of white ash and so shading decreased leaf area instead of increasing it. Huxley (1967) and Wassink and Stolwijk (1956) found similar results where leaf area increased with increased shading up to a certain point and then decreased.

Comparison of the two field studies results shows some interesting points that warrant further research. Overall, all of the shade treatments showed better growth than any of the fern treatments. The 0% fern treatments showed 100% lower growth than the 0% shade treatments despite the fact that they received equal amounts of sunlight. The only difference between the two treatments was the presence of active fern rhizomes in the 0% fern treatments. So obviously light is not the only factor reducing the growth of the seedlings, there is also some sort of soil effect (Drew, 1988). Comparison of soil moisture in both treatments, indicate a difference in percent soil moisture between 0% fern and 0% shade, with 0% shade having greater soil moisture than 0% fern. Perhaps the fern rhizomes are still utilizing soil resources without the fern fronds being present. Perhaps this difference in soil moisture is one explanation for the difference in growth between these two treatments.

It is also possible that the fern litter produced by herbiciding the shade plots enhanced the nutrient supply in the shade treatments, thereby influencing growth. Soil nutrient levels were not intensively studied here so no definite conclusion can be drawn.

CONCLUSIONS

It is obvious from the results of these studies that hayscented fern [*Dennstaedtia puntilobula* (Michx.) Moore] definitely influences the growth of tree seedlings. These studies indicate that hayscented fern produces not only an above-ground influence through changes in light quantity and quality, but a below-ground influence where the causal factors are unknown.

The fern cover study showed that a range of levels of fern cover; 25%, 50%, or 100% did not produce significantly different growth of white ash. Again only the above ground portion of the fern plant was manipulated leaving the entire rhizome system intact. Removing all of the fern fronds produced a significant increase in growth, probably due to increased light.

The shade frame study brought out some interesting points. First, all of the shade frame treatments showed better growth than all of the fern treatments. The seedlings growing in the 80% shade treatments showed better growth than the seedlings growing in the 0% fern treatments, despite the fact that the seedlings in the 80% shade treatments are receiving 80% less light than the 0% fern treatments. Light cannot be the only limiting factor. Second, the seedlings in the 0% shade treatment grew 100% taller than the 0% fern treatment although both were receiving the same amount of sunlight. There must be a below ground influence, whether it be soil moisture differences, nutrient differences, or another means. Although this study focused on light, it was shown that light has perhaps only a very small influence. Further studies are needed to determine what influence the rhizomes have on the seedlings and the soil environment in general. At this point it can be concluded that fern- free conditions are necessary to attain the maximum height growth of seedlings.

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THE INFLUENCE OF SHADE ON NORTHERN RED OAK SEEDLINGS

GROWTH AND CARBON BALANCE

Jennifer L. Jennings[1]

Abstract: One management problem of the Southern Appalachian mixed hardwood forest is the lack of development of northern red oak (*Quercus rubra* L.) seedlings on high quality sites. Regeneration of red oak is not the problem. Following the removal of a stand, a combination of new seedlings, advanced reproduction and stump sprouts ensure red oak's initial presence. However, seedlings and sprouts cannot compete with other hardwoods, specifically yellow-poplar (*Liriodendron tulipifera* L.). Advance reproduction is the exception-if present before harvest, it will usually be present in the new stand. A modified shelterwood cut, in which the understory is removed from below, has been shown to assure the development of this advance reproduction.

Light quantity is thought to be a major factor inhibiting the rapid development of northern red oak. Changes in light intensity will affect biomass partitioning and physiological parameters such as photosynthesis and respiration. To test this hypothesis, seedlings located in the Pisgah National Forest were shaded at 95%, 90%, 60% and 0% full sunlight. Height, diameter and photosynthesis readings were taken throughout the growing season. Seedlings were then destructively sampled and the oven dry weight was found for the leaves, stems and roots of all seedlings. Preliminary data suggest that seedlings shaded at 60% full sun have an increased growth response.

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CHARACTERISTICS OF A LONG-TERM FOREST SOIL PRODUCTIVITY RESEARCH SITE IN MISSOURI

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Abstract: Problems with soil quality and maintenance of soil productivity occur when management activities are improperly planned and carried out. To ensure that Forest Service management practices do not reduce long-term soil productivity (LTSP), a network of coordinated long-term experiments is being established across the United States. The first LTSP study in the Central Hardwood Region is being established in the Ozark Region of southeastern Missouri, in Shannon County. The study area contained mature upland oak-hickory forest with some oak-pine communities. Within the 17.4-ha (43- acre) study area are 27 plots that are approximately 0.4 ha (1 acre) each. The national study plan calls for three levels of organic matter removal (stem only, whole tree, and whole tree plus litter layer) and three levels of compaction (none, moderate, and severe). Logs were lifted from the uncompacted treatment (9 plots) using a skyline yarder system instead of entering plots with a skidder. The remaining plots were harvested conventionally. Pre-treatment data collected include census, height, diameter, and nutrient measurements of overstory, understory, and herbaceous plants; litter and forest floor weights as well as chemical composition; and chemical and physical (porosity and bulk density) soil properties. These data will be used to investigate the effects of treatments on growth, composition, and spatial distribution of woody and herbaceous vegetation, physical and chemical soil properties, and nutrient cycling.

INTRODUCTION

Developing forest management practices to ensure the long-term sustainability of forests is becoming a research and management priority. Interest in sustaining productivity of forest lands has steadily increased since the use of heavy equipment (Woodbury 1930) and whole tree harvesting (Hornbeck and Kropelin 1982). Public concern about the impacts of forest management activities has resulted in legislation (USDA Forest Service 1983) and policies (U.S. Code of Federal Regulations 1985) that mandate research and monitoring (USDA Forest Service 1987) of management systems to protect the productivity of the land.

Before these policies and legislation, there was the National Forest Management Act (NFMA) of 1976, which requires the research and monitoring of federal lands to safeguard the productivity of forest soils (Powers and others 1990). This requirement has led to a nationwide research and monitoring program to define the effects of management practices on long-term soil productivity through the establishment of coordinated, long-term experiments on major timber species, soils, and regions across the United States (Powers and others 1989). The objectives of the program are to: (1) quantify the effects of soil disturbance from management activities on long-term productivity; (2) validate soil monitoring standards developed in compliance with the NFMA of 1976; (3) learn more about the fundamental relationships between soil properties, long-term productivity, and forest management practices; and (4) evaluate the potential for mitigating the adverse effects of disturbance.

The soil properties believed to be primarily responsible in controlling forest productivity are soil porosity and soil organic matter content. These are also the soil properties greatly impacted by forest management activities. The experimental design used creates an artificially imposed factorial gradient in soil porosity and organic matter content to generate a productivity response as a function of these two soil properties rather than the impact of operational practices. The response relating changes in growth potential to changes in site organic matter and soil porosity will

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allow us to estimate the magnitude of damage to forest productivity that has been generated by disturbance. Except for treatments producing minimal soil disturbance, the remaining treatments sometimes greatly exceed the degree of disturbance caused by normal logging operations.

The national long-term soil productivity (LTSP) study plan calls for three levels of compaction (none, moderate, and severe) and three levels of organic matter removal (stem only, whole tree, and whole tree and litter layer). Most harvesting operations currently being conducted in the Central Hardwood Region fall into the lowest levels of compaction and biomass removal. The long rotation length (approximately 80 years), high installation cost, and commitment to maintain, measure, and report on study progress are not small matters. The present study (LTSP in Missouri) is being led by Forest Service Research, with cooperation from the Mark Twain National Forest, Missouri's Department of Conservation and Department of Natural Resources, Natural Resource Conservation Service, and the University of Missouri.

This paper describes pretreatment data collection and treatment installation; and it presents pretreatment data to characterize the first LTSP in the Central Hardwood Region.

METHODS

The Study Site

Site selection, plot criteria, experimental design, and treatments for the Missouri LTSP are based on information in the national LTSP study plan (Powers and others 1989). The study is located on the Carr Creek State Forest (Missouri Department of Conservation) in Shannon County. Mean annual precipitation there is 112 cm (44 inches), and mean annual temperature is 13.3°C (56°F). The site occupies the upper sideslopes of two northeastern ridges that extend from north to south in the southeastern Missouri Ozarks. It contained a well-stocked, mature, second-growth oak-hickory forest. Site index ranges from 74 to 80 based on black oak (*Quercus velutina* Lam.) at 50 years (Hahn 1991). The oak-hickory timber type is the major timber type in the Central Hardwood Region occurring over a variety of soils, relief, and stand conditions.

The sloping topography (20-28% slopes) has small shallow streams that contain exposed cobbles and stones. The area is underlain mainly by Ordovician dolomite, and areas of Cambrian dolomite and Precambrian igneous rocks are also present (Missouri Geological Survey 1979).

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). Water drains freely through the soils into subsurface channels.

Treatments

A preliminary soil survey was made to locate the study area on relatively uniform soil. Soil pits were dug and soils were described and analyzed with cooperation from the Missouri Department of Natural Resources, the Natural Resources Conservation Service, the Mark Twain National Forest, and the University of Missouri. Field examination included estimation of depth to restrictive layer, horizon thickness, texture, and gravel content. After estimating that the key soil properties did not vary significantly across the areas, we established boundaries for plots. Three replicates of nine treatment plots approximately 0.4 ha (1 ac.) in size were laid out in the summer of 1993. Four-meter-wide buffer strips were laid out around all plots. Treatments include a nine-plot factorial of three levels each of organic matter removal and soil compaction. Organic matter removal consists of: (1) stem only, (2) whole tree, and (3) whole aboveground biomass including the forest floor. Soil compaction levels are none, moderate, and severe. Severe compaction is defined as 80% of the difference between the hypothetical growth-limiting bulk density (Daddow and Warrington 1983) and the bulk density of the uncompacted soil. Moderate is midway between the extremes.

Inventory

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 (MOFEP) botany crew of the Missouri Department of Conservation. Overstory measurements were made on 0.2-ha (0.49-ac.) circular plots. All trees living and dead standing 11.5 cm (4.5 in.) dbh and greater were identified, and dbh was measured. All live saplings between 3.8 cm (1.5 in.) and 11.2 cm (4.4 in.) dbh were identified and measured on four circular 0.02-ha (0.05-ac.) plots located 17.25 m (56.4 ft.) from the center point in each cardinal direction (N, S, E, and W). Understory woody vegetation 1.3 cm but less than 3.8 cm dbh and greater than 1 m tall was measured on 0.004- ha (0.01-ac.) plots that were located within sapling plots. Dead and downed woody material greater than 5 cm dbh and 0.6 m long was inventoried (species, maximum diameter, length, and decomposition class) along each of four line intercepts 17.25 m (56.4 ft.) in length. Herbaceous vegetation was identified and counted on four 1-m^2 plots that were located 6.1 m from the sapling plot center along NE, SE, SW, and NW transects; a total of 16 herbaceous plots were sampled within each 0.2-ha plot.

Biomass and Nutrient Sampling

Biomass samples were collected for overstory canopy trees, understory saplings, ground vegetation, and leaf litter/humus layer. All samples were dried at 105°C for 72 hr or until a constant weight was achieved. Each sample was then ground to a fine powder in a Wiley Mill and analyzed for macronutrient and micronutrient content.

Leaf litter/humus samples were collected from eight 0.125-m^2 plots that were located 5 and 15 m from each plot center along N, S, E, and W transects. Leaf litter was separated from humus material, and all eight samples were combined for biomass and nutrient analysis. Woody and herbaceous ground cover was clipped separately on four 2.5-m^2 plots located at 5 m from plot center along the same cardinal transects. Plot centers for these samples correspond to each of the 27 treatment plots but not necessarily with all MOFEP inventory plot centers. Samples were collected before leaf drop in the fall of 1993.

A total of 54 trees were selected, felled, and weighed in the field during the spring of 1994. Twenty-six saplings (dbh < 10 cm) and 28 midstory and overstory trees (dbh > 10 cm) were selected for measuring total tree biomass. Individual trees were selected based on the following criteria: 1) one of the major species present in the overstory; i.e., black oak, scarlet oak, hickory spp., white oak, and shortleaf pine, 2) sapling with a dbh between 2.5 and 10 cm, at least one for each 2.5-cm size class, 3) midstory and overstory trees with a dbh between 10 and 50 cm, when possible, one for each 5-cm size class, and 4) trees that appeared vigorous and healthy with no outward signs of rot or limb damage. The dbh of trees sampled covered the range from 5 cm to 50 cm. Total height, dbh, basal diameter, and specific crown dimensions were measured on each tree. Trees with merchantable logs were sectioned in appropriate lengths: others were cut to 2-m lengths. Sawdust samples and wood disks were collected from the end of each log section up to the 10-cm diameter. Tree crowns were sectioned into <2 cm, 2 to 10 cm, and 10+ cm size classes. Fresh weights of all tree logs and crown portions were measured in the field. Some merchantable logs were too large to be weighed in the field (scale limit = 500 lbs). Thus, detailed log dimensions were taken to estimate volume: weight was later predicted from laboratory samples. Samples were also collected from each crown size class for further analysis. Fresh weights of all sawdust samples, wood disks, and crown subsamples were measured separately.

Soil Sampling

In addition to soil pits, soil samples were collected with a two-person power-driven coring device in a systematic design. Four intact cores (7.6 cm diam. x 40 cm long) of soil per plot were collected and partitioned according to soil horizon before determination of bulk density, pH, organic matter content, hydraulic conductivity, and macronutrient content (Soil Survey Staff 1984, Page and others 1982, Black 1965).

Timber Harvesting

Trees and biomass were removed according to protocols in the national LTSP study plan (Powers and others 1989) by means that minimize soil disturbance. Trees were harvested over a 4-month period in 1994, beginning in February and ending in May. On plots designated as uncompacted, merchantable trees were directionally felled and removed with a skyline cable logging system. Merchantable trees on remaining plots, 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. Remaining crowns, unmerchantable trees, dead and live snags, dead and down, leaf litter layer, and other debris were removed manually according to protocols. Depending on treatment, it was necessary to replace these materials (except the leaf/litter layer) after compaction was completed in some plots.

Tree crowns were retained on the stem only plots. On plots where the whole tree and leaf/litter layer were removed, the total aboveground biomass was removed. All understory vegetation was clipped and removed, and the forest floor was raked away to the mineral soil. The raking of the forest floor began in August and was completed by mid-November. Skidders and tractors were not permitted on uncompacted plots, but they were permitted on compacted plots.

After plots are compacted with a 14-ton vibrating sheep-foot roller, each 0.4-ha treatment plot will be regenerated with the appropriate timber species. Half of each plot will be kept weed free to permit target trees to grow freely. The other half will be allowed to develop naturally into a more complex community of trees and other vegetation. Net primary productivity in these two plant communities will provide direct measures of productivity as influenced by the degree of soil disturbance.

RESULTS AND DISCUSSION

The values for chemical properties, bulk density, and percent organic matter in Table 1 are typical for soils in the Clarksville series (Gott 1975). There is concern that the high gravel content in the soil could reduce the depth to which the soil can be effectively compacted. However, principal investigators of some LTSP studies have had problems in achieving a distinct difference between the moderate and severe levels of compaction in soils with low gravel content (Personal communication).

Table 1. Mean chemical properties and bulk density of soil in the Missouri LTSP study*.

Depth	Extractable bases				Extr.	Organic	pH	Bulk
	Ca	Mg	K	Na	Al	C	H_2O	density
cm	meq/100g					%		g/cm^3
0-10	4.6	0.8	0.3	Tr	0.8	5.4	5.5	1.54
10-20	0.5	0.4	0.2	Tr	1.1	0.5	5.3	1.58
20-30	0.6	0.5	0.1	Tr	1.2	0.1	4.8	1.70
30-40	1.1	2.3	0.2	Tr	3.1	0.1	4.9	1.71

*N and P analyses are not complete and are not presented.

Census Data

Overstory species composition by diameter class is shown in Figure 1. The red oak group consists primarily of black and scarlet (*Quercus coccinea* Muenchh.); the white oaks include white (*Quercus alba* L.) and post (*Quercus stellata* Wangenh.); the hickory group includes black (*Carya texana* Buckl.), mockernut (*Carya tomentosa* Nutt.) and pignut

(*Carya glabra* Mill.); the other category is composed of black walnut (*Juglans nigra* L.), blackgum (*Nyssa sylvatica* Marsh.) and dogwood (*Cornus stolonifera* L); and the predominant native pine species is shortleaf pine (*Pinus echinata* Mill.). Average height of co-dominants for each class were: 25.9 m for red oaks, 23.2 m for white oaks and hickory, and 23.8 m for pine. Red oaks, white oaks, hickory and pine made up 47.5, 26.1, 17.3 and 4.2 percent, respectively, of the overstory. Species composition of the sapling understory is shown in Figure 2. Hickory, white oaks, and dogwood accounted for 80 percent of all stems; the red oaks represented less than 3 percent. Seedling composition (less than 3.8 cm dbh) is shown in Figure 3. Total number of stems per hectare in this group was 1,905.

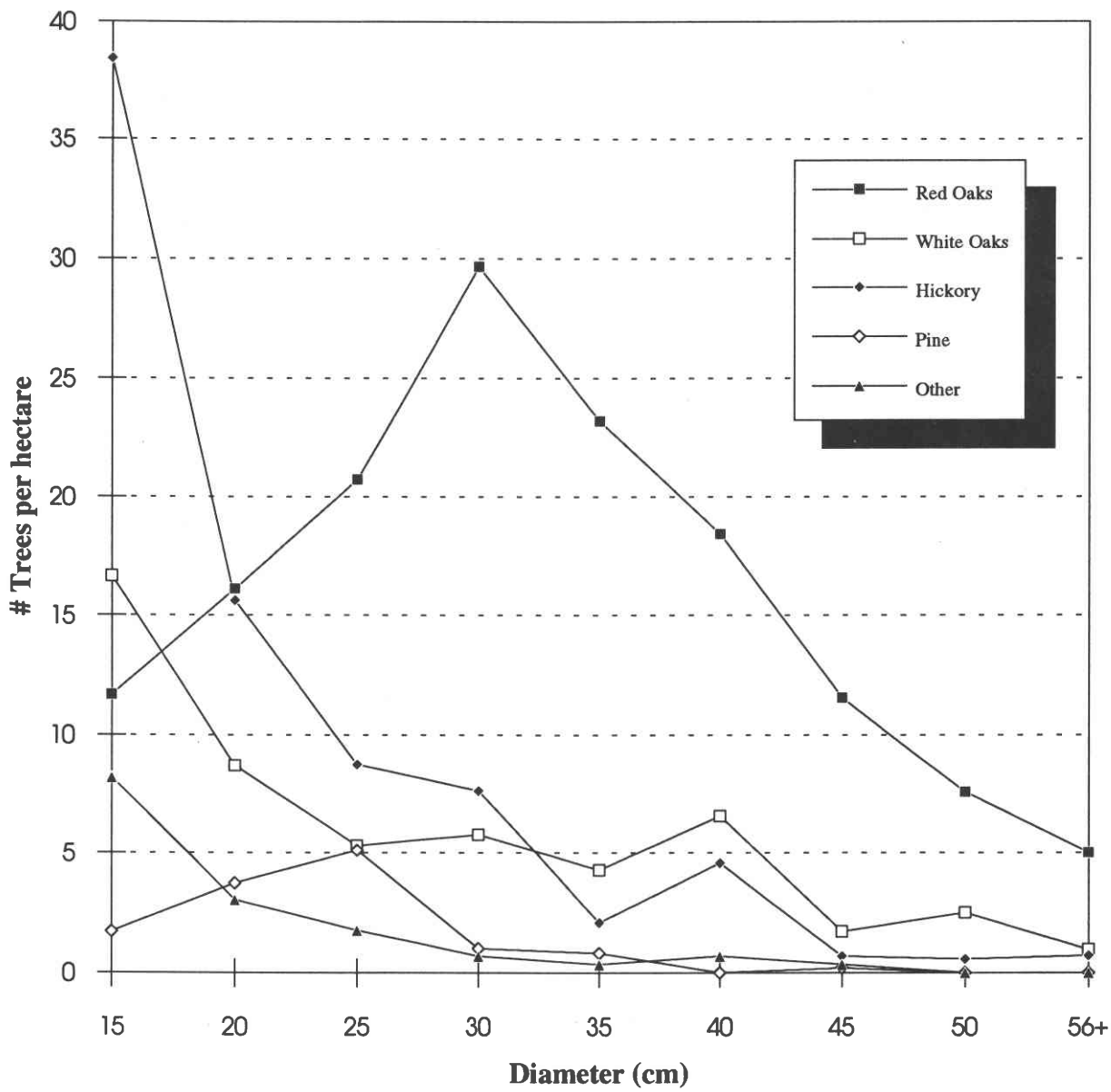

Figure 1. Overstory diameter frequency distribution by species for LTSP study site.

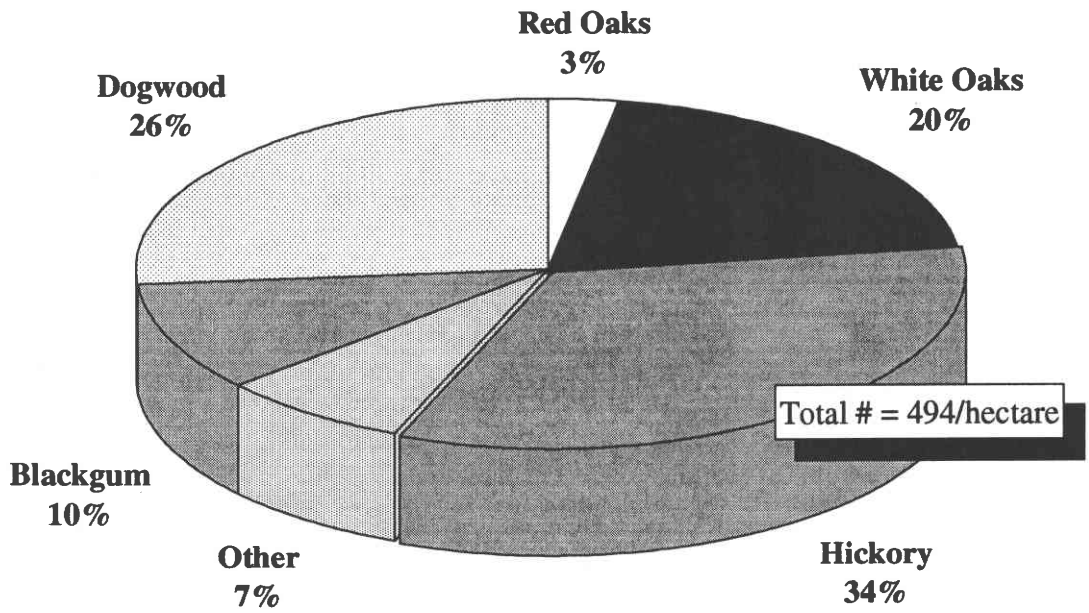

Figure 2. Sapling understory (3.8 to 12.5 cm dbh) species compostion.

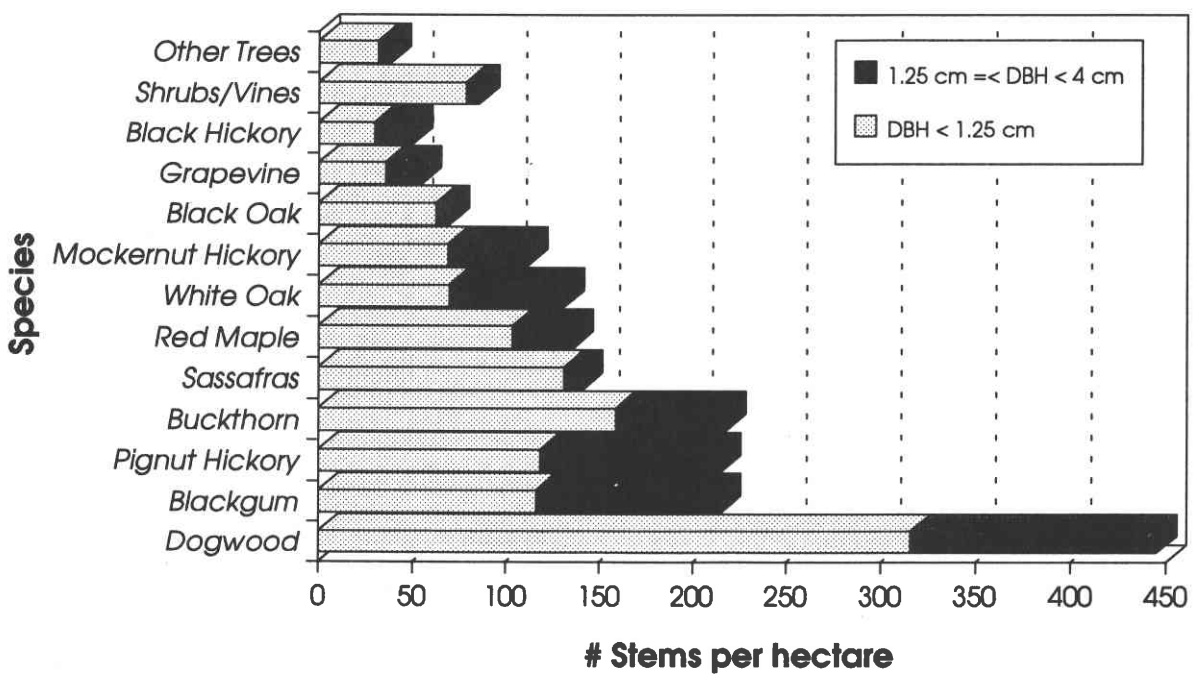

Figure 3. Number of stems per hectare of understory woody species at least 1 m tall.

The total number of dead standing trees is shown in Figure 4. The average number per hectare was 30. Most of the dead stems (74%) were either black or scarlet oak. Decay stage was recorded but is not presented here. The stages range from recently dead to highly decomposed but still standing.

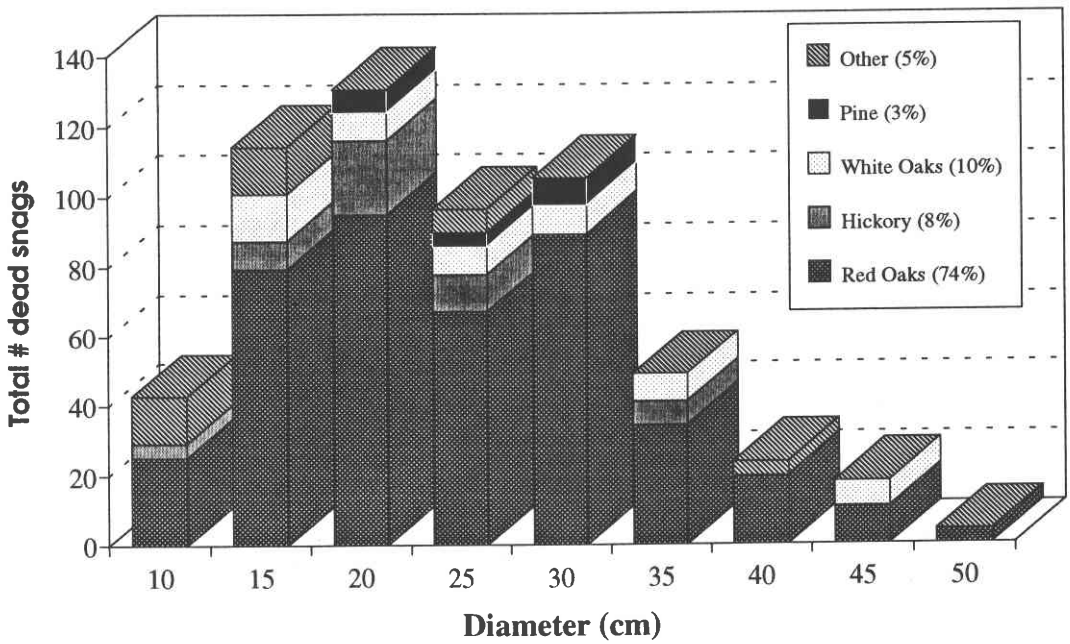

Figure 4. Total number and percent composition of dead standing trees on LTSP study site.

Biomass and Nutrient Data

Total dry weight biomass was calculated for all individual trees that were sampled. Dry weight was then plotted against dbh for each species, and biomass at each 5-cm diameter class was extrapolated from an eye-fitted curve. Combined with the number of trees per acre, total dry weight of the overstory (>4 cm) was estimated to be 175.5 tons per hectare (1,016 kg/ton). Figure 5 depicts the total amount of biomass estimated to be removed by each of the three organic matter treatments. Of the 175.5 tons of woody dry matter, red oaks comprised 63%, white oaks 15%, hickory 16.5%, pine 2%, and other species 3.5%.

Allometric regression equations of the form Y = a(DBH)**b were also developed and compared to these estimates. Additionally, the literature on biomass equations for the species found on our site was reviewed and used for comparison. Dry weight estimates for white oak and hickory by Clark and others (1985): black, scarlet and white oak and hickory equations by Wiant and others (1977); and black oak sawlog biomass equations by King and Schnell (1972) were compared to the dry weight estimates from our regression.

In general, estimates for dry matter for all species compared fell within one standard deviation of our estimates for size classes below 25 cm dbh, but estimates of dry matter above 25 cm ranged from 1.3 to 9.5 standard deviations from our data. One exception was the white oak estimates from Wiant and others (1977), in which all size classes were within 1.1 standard deviations of our data.

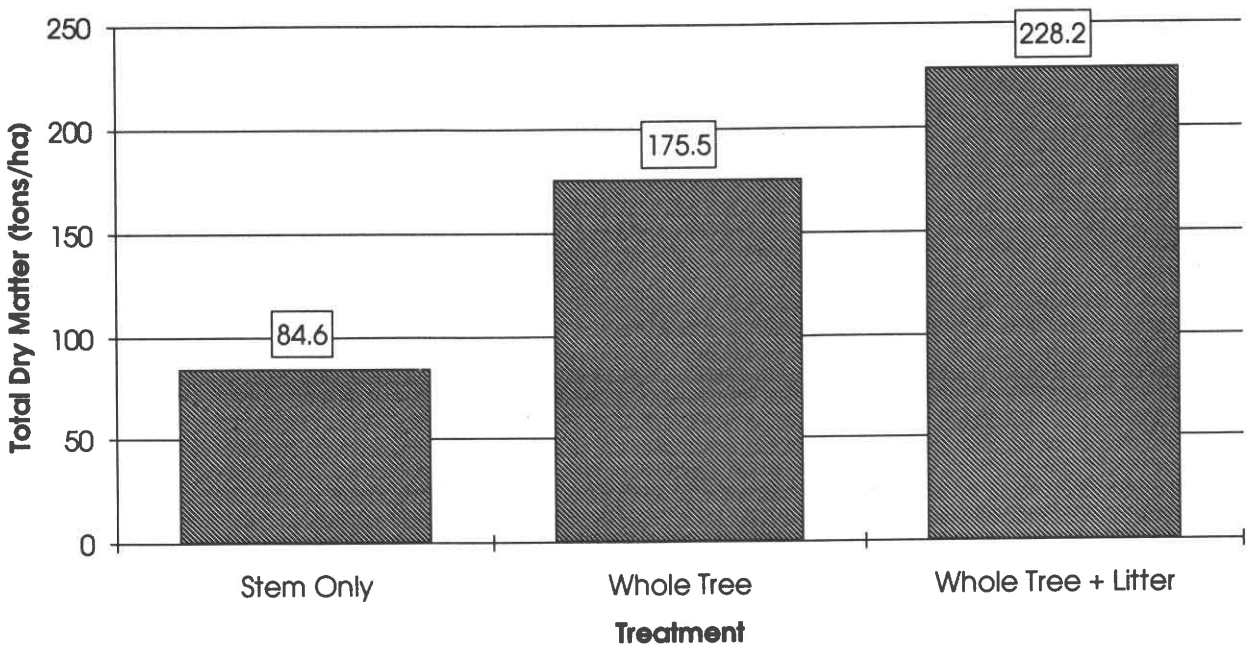

Figure 5. Total dry matter removed from plots relative to treatment.

Nutrient analysis was completed for all subsamples of ten individual trees, two of each dominant overstory species and dogwood. Crown, wood, and bark components were analyzed separately and multiplied by the proportion of the biomass they represented. Total macronutrient and micronutrient content is summarized in Table 2. The amount of nutrients removed in the biomass increased considerably when the leaf/litter layer was included.

The whole tree plus leaf/litter layer treatment removed 33% more N, 46% more P, 10% more K, 18% more Ca, and 35% more Mg than the whole tree treatmentl. Among micro-nutrients, considerably more Mn, Fe, and Al were removed in the whole tree plus leaf/litter treatment than in the whole tree treatment. Branches and unmerchantable tree parts also have large amounts of nutrients as shown when the whole tree treatment is compared to the merchantable stem only treatment. Removing the whole tree removed 3 times more N and P, 2 times more K, 3 times more Ca, and 2.6 times more Mg than did removing only the stem. Similar trends were apparent in micronutrients between whole tree and stem only treatments.

Table 2. Macronutrient and micronutrients removed from the site relative to organic matter treatment.

Nutrient	TREATMENTS Stem only	Whole tree	Whole tree + litter*
	kg/hectare		
Macro	195.26	540.04	810.80
N	7.41	25.96	48.30
P	108.65	255.61	284.90
Ca	774.24	2303.11	2819.22
Mg	19.97	52.59	81.20
Micro			
Mn	7.08	18.00	49.37
Fe	1.01	2.56	17.48
Zn	0.53	1.55	2.61
Al	2.10	7.61	81.12
Na	0.48	1.08	1.63
Cu	0.11	0.32	0.58
B	0.38	1.13	1.50

*Excluding snags and dead woody debris.

SUMMARY

This is a long term study that is in the latter part of the installation phase. It will be several years before answers about the relationships between management practices and long-term soil productivity are available. Data from this study will provide information for evaluating the effects of soil disturbance on long-term productivity of several Central Hardwood tree species and their associated flora and fauna communities. The results will help us better understand the joint role of soil porosity and site organic matter in their effect on site processes that control productivity. Combining information from the network of LTSP studies with other ecological information is expected to provide managers with more complete information with which to manipulate soil properties to either increase productivity or prevent reductions.

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A SUMMARY OF WATER YIELD EXPERIMENTS ON HARDWOOD FORESTED WATERSHEDS IN NORTHEASTERN UNITED STATES

J.W. Hornbeck[1], M.B. Adams[2], E.S. Corbett[3], E.S. Verry[4], and J.A. Lynch[5]

Abstract: This paper summarizes and compares long-term changes in annual water yield following cutting experiments at four locations in northeastern United States. Substantial increases in water yield of up to 350 mm yr^{-1} were obtained in the first year by clearfelling hardwood forest vegetation and controlling regrowth with herbicides. Commercial clearcutting of hardwoods with natural regrowth resulted in smaller initial increases in water yield of 110 to 250 mm yr^{-1}. This range in response was due to differences in precipitation and configuration of cuttings. Unless regrowth was controlled with herbicides, yield increases declined quickly after cutting, seldom persisting for more than 10 years. However, yield increases were readily extended over 20 years or more with intermediate cuttings and/or repeated control of regrowth with herbicides. Nearly all increases in water yield occur during summer and early autumn. Changes in species composition after forest cutting on several study watersheds eventually resulted in decreased water yields compared to those from uncut, control watersheds. Results are discussed in terms of implications for surface water supplies, global climate change, nutrient cycling, hydrologic modeling, and long-term research.

INTRODUCTION

Long-term paired watershed studies at four locations in the northeastern United States are analyzed in this paper: Fernow Experimental Forest in north-central West Virginia, Leading Ridge Experimental Watersheds in central Pennsylvania, Marcell Experimental Forest in north-central Minnesota, and Hubbard Brook Experimental Forest in central New Hampshire (Fig. 1). These locations span the diverse geography of the northeastern United States. At each site, one or more watersheds have been calibrated against a nearby control, then treated experimentally. Changes in water yield were then determined with results in some cases spanning up to 3 decades.

Two of the study locations, Fernow and Leading Ridge, are forested with central hardwoods. This provides an opportunity to compare streamflow responses in central hardwood forests with 2 other important forest types in the Northeast: northern hardwoods at Hubbard Brook and aspen-birch at Marcell.

Results from these studies have special significance for the northeastern United States. Forests cover >60% of the landscape, and forested watersheds serve as sources of water for more than 1,000 municipalities ranging from small, rural communities to large urban centers such as New York, Baltimore, and Boston. Although the region is well watered with 1,100 mm average annual precipitation, water shortages are not uncommon. A knowledge of how both abrupt and gradual changes in forest cover affect water yield over time periods on the order of decades is needed to manage forested watersheds for optimum water production.

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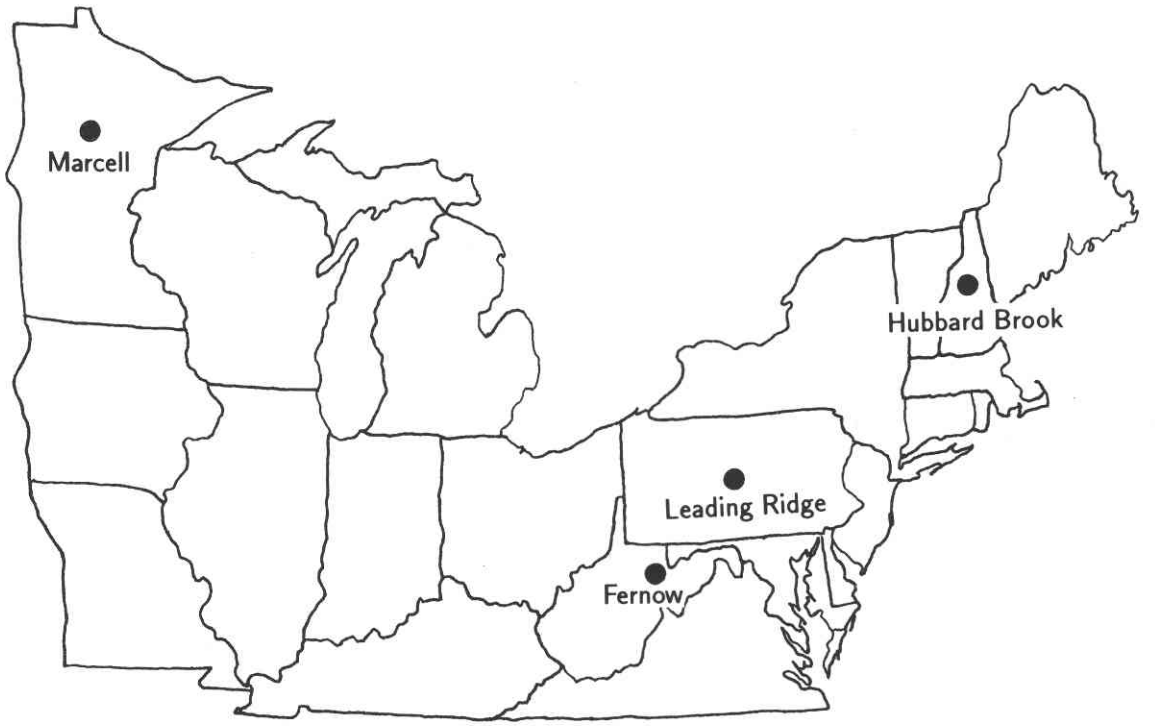

Figure 1. Northeastern United States showing study site locations.

METHODS

Results from 11 separate, treated watersheds are summarized and compared in this paper (Table 1). Record collection on many of these watersheds began in the 1950s, followed 5 to 10 years later by initial treatments of vegetation. At the time of these initial treatments, the northeastern United States was experiencing an extended period of below average precipitation and there was widespread interest in the potential for increasing water yield from forested watersheds. Also, controversies over the use of herbicides and forest clearcutting had not begun to escalate. Thus, some of the initial treatments at Fernow, Hubbard Brook, and Leading Ridge were designed to obtain benchmark information on maximum possible yield increases, and treatments included complete forest clearing and control of regrowth with herbicides. Later experiments performed in the 1970s and 1980s focused more on determining impacts of commercial harvesting operations. These treatments tended to be less drastic and did not include herbicide applications.

The longevity of the studies has in many cases allowed for determining impacts of multiple treatments on the same watershed. For example, watershed 3 on the Fernow Experimental Forest was first harvested in 1958-59 by intensive selection, a silvicultural practice that was in common use in the region and which involves cutting a relatively small fraction (in this case, 13%) of total basal area. In keeping with the silvicultural prescription, the treatment was repeated 4 years later with an additional 8% of total basal area being cut. The prescription was changed in 1968 to patch cutting, and 6% of the existing basal area was felled. Then in late 1969, to obtain information relative to a developing controversy over impacts of clearcutting (Horwitz 1974), the watershed was subjected to a commercial clearcutting during which 91% of existing basal area was cut.

Table 1. Descriptions of catchment studies in northeastern United States

Catchment	Area-ha	Mid elevation-m	Vegetation (before treatment) and soils	Mean annual precipitation-mm[1]	Mean annual streamflow-mm[1]	Description of treatment	Year(s) of treatment	Basal area cut - %[2]	Reference
Fernow Experimental Forest, West Virginia				1,460	640				
1	30	755	Central Appalachian hardwoods, Typic Dystrochrepts			Clearcut to 15cm dbh except culls.	1957-58	74	Kochenderfer and others 1990
2	15	780	same			Diameter limit cut to 43 cm. Diameter limit cut to 43 cm.	1958 1972	32 12	same
3	34	505	same			Intensive selection. Intensive selection. Patch (0.2 ha) cuttings. Clearcut.	1958-59 1963 1968 1969-70	13 8 6 91	same
6	22	805	same			Clearcut lower half. Herbicides on lower half. Clearcut upper half. Herbicides on entire catchment Plant Norway spruce. Herbicides on entire catchment	1964 1965-69 1967-68 1968-69 1973 1967-69	51 49 	Patric & Reinhart 1971, Kochenderfer and others 1990
7	24	801	same			Clearcut upper half. Herbicides on upper half. Clearcut lower half. Herbicides on entire catchment.	1963-64 1964-69 1966-67 1967-69	49 51 	same
Leading Ridge Experimental Watersheds, Pennsylvania				1,060	440				
2	43	360	Central harddwoods, Typic Dystrochrepts and Typic Hapludults			Clearcut lowest 9 ha. Clearcut mid-slope 11 ha. Herbicide lower and mid-slope areas. Clearcut 17 ha on upper slope. Herbicide all clearcut areas.	1967 1971-72 1974 1975-76 1977	24 27 40 	Lynch and others 1980
3	104	340	same			Clearcut on 45 ha.	1976-77	43	Lynch and Corbett 1990
Marcell Experimental Forest, Minnesota				760	110				
4	34	433	Aspen-birch upland, black spruce peatland, Typic Glossoboralfs, interspersed with bog wetlands			Aspen-birch upland portion (26ha) clearcut (all trees >3 m height)	1970-71	100	Verry, 1987
Hubbard Brook Experimental Forest, New Hampshire				1,340	880				
2	16	595	Northern hardwoods, Typic Haplorthods			Clearfelled. Herbicides on entire catchments.	1965-66 1967-69	100 	Hornbeck and others 1970
4	36	606	same			Progressive strip cut, 1st 3rd harvested. 2nd 3rd harvested. Final 3rd harvested.	1970 1972 1974	33 33 33	Hornbeck and others 1987
5	22	636	same			Whole-tree harvest.	1983-84	95	None available

[1]Long-term means for untreated, control catchments.

[2]Based on existing tree basal area on catchment at initiation of cutting.

Other forms of multiple treatments included a three-stage, progressive strip cutting on watershed 4 at Hubbard Brook, complete forest clearing in three stages on watershed 2 at Leading Ridge and in two stages on watersheds 6 and 7 at Fernow, and controlling vegetation with herbicides prior to or during species conversion or natural regrowth at Leading Ridge, Fernow, and Hubbard Brook (Table 1).

Impacts of treatment on water yield were determined using the paired watershed approach described by Reinhart (1967). Linear regression was used to develop a calibration relationship between annual water yield from a control watershed (independent variable, X) and a watershed to be treated (dependent variable, Y). At locations with multiple watershed experiments, the same control was used in developing all regressions. The calibrations are based on 5 or more water years of record, and although most of the calibration periods ended at least 2 decades ago, we assume the relationships still apply. Forests on all control watersheds are mature and reasonably steady-state with regard to biomass and leaf area (Bormann and Likens 1979) and annual evapotranspiration (Federer and others 1990). Thus, water yield relationships for control watersheds should be unchanging, except during a 2-year period of severe insect defoliation at Leading Ridge (Corbett and Heilman 1975).

After treatment, deviations from the calibration regressions were considered to be statistically significant and attributed to treatment if they exceeded 95% confidence intervals about the regressions. The deviations, which indicate increases and decreases in annual water yield from the treated watersheds, are presented in both graphical and tabular form in this paper. Statistical significance is not indicated when using graphs. However, on average for all sites, deviations greater than $\pm$ 30mm yr^{-1} from the calibration regressions were statistically significant.

RESULTS

The array of forest treatments across the four study locations caused a variety of responses in water yield (Fig. 2, Table 2). However, three generalizations can be used as a framework for discussing results: (1) initial increases in water yield occur promptly after forest cutting, with the magnitude being roughly proportional to percentage reduction in basal area, (2) the increases can be prolonged for an undetermined length of time by controlling natural regrowth; otherwise increased streamflow diminishes rapidly, usually within 3 to 10 years, and (3) small increases or decreases in water yield may persist for at least a decade, and probably much longer, in response to changes in species composition and climate.

Initial Increases in Water Yield

Only Hubbard Brook and Marcell Experimental Forests normally have continuous winter snowpacks. Snowmelt runoff occurred earlier at both sites, but volume of snowmelt runoff was unchanged (Hornbeck 1975, Verry and others 1983). Thus, increases in annual yield at all four study sites resulted primarily from reductions in transpiration and canopy interception. Simply stated, soils were wetter on recently treated watersheds and more water was available for streamflow. Flow-duration curves for posttreatment periods at each site show that nearly all changes in water yield result from increases at low flow levels, or as augmented baseflow or delayed flow, and that flood flows were not greatly affected (Hornbeck and others 1970, Lynch and others 1980, Patric and Reinhart 1971, Verry 1972). Further, the yield increases occurred primarily in the growing season. Complete recharge of soil moisture on both forested and treated watersheds usually occurs soon after the start of the dormant season, thus limiting further opportunities for treatment effects until the start of the next growing season.

As found in previous summaries (Bosch and Hewlett 1982, Douglass and Swank 1972), increases in yield for the first water year after treatment were roughly proportional to percentage reductions in stand basal area. Moreover, a comparison for all sites (Fig. 3) supports previous findings that reductions in basal area must approach 25% to obtain measurable responses in annual water yield (Douglass and Swank 1972). Above this threshold, there is some variability in first-year responses among watershedss with similar basal areas cut, but differences usually can be explained by factors such as configuration and timing of the cutting, and whether regrowth was controlled with herbicides.

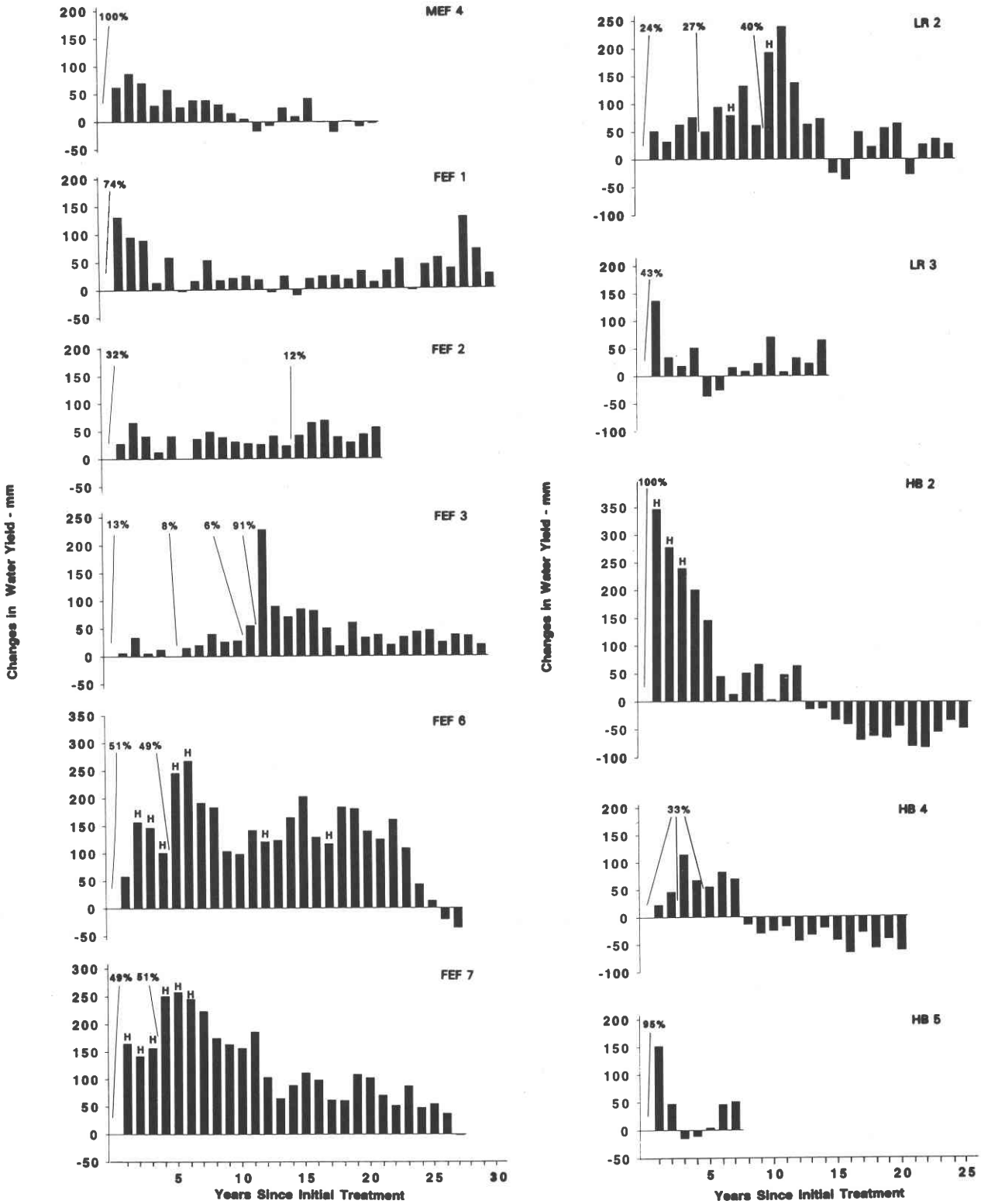

Figure 2. Changes in water yield for 11 experimental watersheds studies. MEF is Marcell Experimental Forest, FEF is Fernow Experimental Forest, LR is Leading Ridge Experimental Watersheds, and HB is Hubbard Brook Experimental Forest. Percentage values denote existing basal area cut during experimental treatments and H signifies herbicide application to cut portions of watersheds.

Table 2. Changes in annual water yield for selected watersheds

Year after initial treatment	Marcell Experimental Forest #4				Fernow Experimental Forest #7			
	Estimated streamflow if untreated[1]	Change due to treatment[2]		Precipitation	Estimated streamflow if untreated[1]	Change due to treatment[2]		Precipitation
	mm	mm	%	mm	mm	mm	%	mm
1	209	81*	39	871	715	165*	23	1,316
2	195	114*	58	817	390	142*	36	1,057
3	130	92*	70	779	669	157*	23	1,231
4	188	38*	20	862	658	251*	38	1,215
5	227	77*	34	780	637	258*	40	1,217
6	76	34*	45	458	738	246*	33	1,332
7	152	51*	34	935	675	224*	33	1,268
8	246	52*	21	882	869	175*	20	1,481
9	276	40*	15	786	1,030	164*	16	1,557
10	108	19	18	662	919	157*	17	1,553
11	198	6	3	821	944	187*	20	1,551
12	282	-22	-8	799	628	104*	17	1,243
13	100	-9	-9	748	607	65	11	1,170
14	202	33*	16	744	755	89*	12	1,328
15	286	12	4	920	988	112*	11	1,628
16	217	56*	26	796	828	99*	12	1,451
17	178	-1	0	744	755	62*	8	1,397
18	296	-25*	-8	995	987	61	6	1,587
19	281	1	0	839	801	109*	14	1,470
20	125	-12	-10	587	852	103*	12	1,529
21	89	-4	-4	711	842	71*	8	1,487
22					1,073	52	5	1,745
23					694	88*	13	1,389
24					516	48	9	1,191
25					766	55*	7	1,435
26					958	37	4	1,561
27					936	-2	0	1,583

Continued on next page

Table 2. (continued) Changes in annual water yield for selected watersheds

Year after initial treatment	Leading Ridge Experimental Watershed #2				Hubbard Brook Experimental Forest #2			
	Estimated streamflow if untreated[1]	Change due to treatment[2]		Precipitation	Estimated strreamflow if untreated[1]	Change due to treatment[2]		Precipitation
	mm	mm	%	mm	mm	mm	%	mm
1	433	51*	12	967	851	347*	41	1,279
2	302	32*	11	897	954	278*	29	1,394
3	401	63*	16	1,008	919	240*	26	1,271
4	562	76*	14	1.167	902	200*	22	1,294
5	507	50*	10	1,061	840	146*	17	1,225
6	940	94*	10	1,435	787	44	6	1,221
7	462	79*	17	1,018	1,059	12	1	1,504
8	529	132*	25	1,104	1,469	52	4	1,832
9	658	61*	9	1,210	832	67*	8	1,240
10	606	193*	32	1,214	1,305	3	0	1,659
11	682	239*	35	1,285	884	48	5	1,323
12	648	138*	21	1,220	996	64*	6	1,431
13	687	63*	9	1,255	902	-13	-1	1,287
14	260	73*	28	877	764	-13	-2	1,139
15	570	-25	-4	1,089	807	-34	-4	1,261
16	528	-37	-7	1,060	1,179	-41*	-3	1,530
17	570	49*	9	1,091	885	-70*	-8	1,328
18	559	22	4	1,121	1,099	-62*	-6	1,522
19	500	56*	11	1,106	715	-64*	-9	1,087
20	558	64*	11	1,178	958	-44*	-5	1,310
21	422	-28	-7	964	872	-80*	-9	1,210
22	410	26	6	964	790	-82*	-10	1,190
23	584	36*	6	1,163	708	-56*	-8	1,149
24	777	27	3	1,139	1,109	-34	-3	1,468
25					1,194	-48*	-4	1,590

[1]Determined form calibration regression

[2]Determined by subtracting estimated streamflow from actual streamflow

[3]Change exceeded 95% confidence interval about the calibration regression

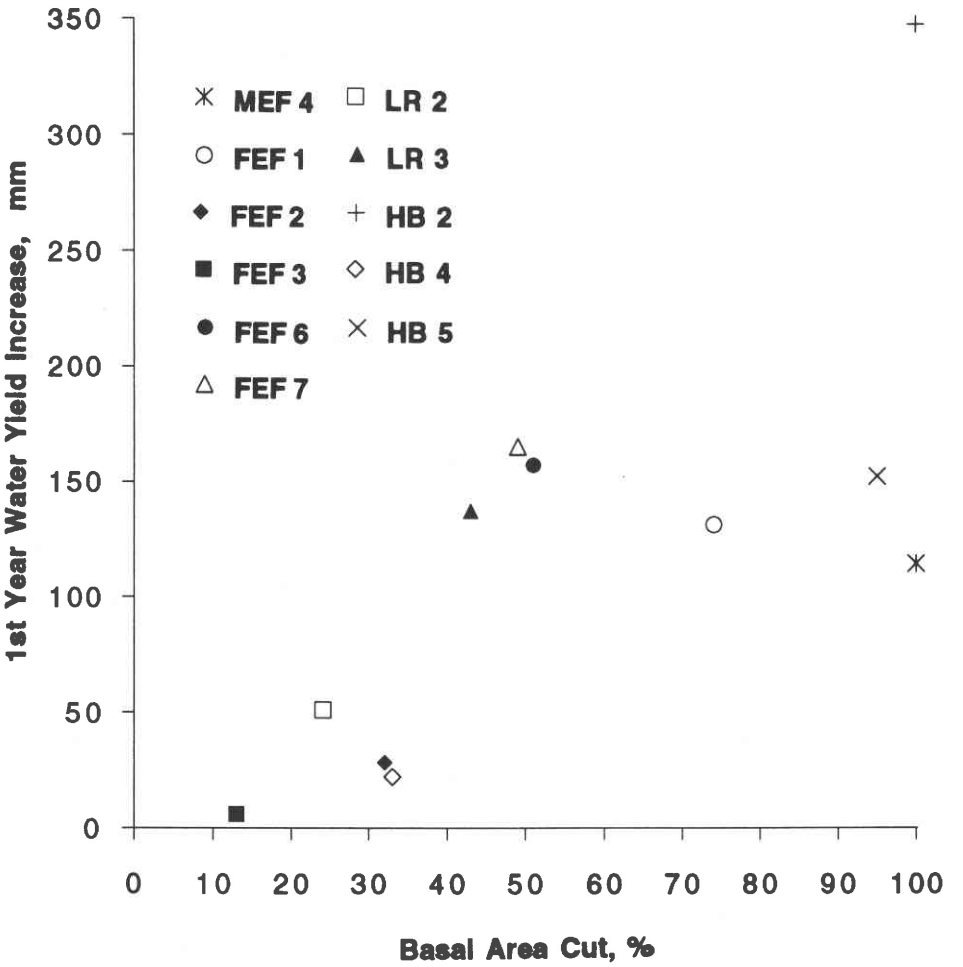

Figure 3. First-year increases in water yield in response to forest cutting.

As an example of the role of configuration, the cutting of 24% of the basal area on watershed 2 at Leading Ridge produced a nearly twofold larger increase than cutting one-third of the basal area on watershed 4 at Hubbard Brook and watershed 2 at Fernow (Fig. 3). The cutting at Leading Ridge was in a single block on the lowest portion of the watershed, the cutting at Hubbard Brook was in a series of strips spaced equidistant from bottom to top of the watershed, and the cutting on the Fernow involved harvesting individual trees scattered about the watershed. The cutting of strips and individual trees increases crown exposure and transpiration rate of residual trees, especially those bordering openings (Federer and Gee 1974). A portion of the added transpiration may be drawn from the extra water

available in the cut strips or individual tree openings, or from soil water moving downslope from cut to uncut areas. As a result, increases in streamflow were smaller than had the areas been cut like the single, low-elevation block at Leading Ridge. Larger increases for the first water years after cutting the second and third sets of strips at Hubbard Brook watershed 4 (Fig. 2) support this explanation.

The importance of timing of cutting and control of regrowth are demonstrated by comparing Hubbard Brook watersheds 2 and 5. On watershed 2, 100% of the basal area was clearfelled during the dormant season, and herbicides were applied early in the next growing season. This combination proved optimum for increasing water yield and resulted in a first-year increase of 347 mm, the maximum for all experimental treatments (Figs. 2, 3). By contrast, 95% of the basal area on watershed 5 was felled during a whole-tree harvest that spanned nearly a full year, and natural regrowth was uncontrolled. The first-year increase in water yield from watershed 5 was only 152 mm, or 44% of that from watershed 2. The difference in first-year increases from the watersheds is due largely to greater transpiration and interception by regrowth on watershed 5.

Impacts of Controlled Versus Natural Regrowth

Herbicides were used to control regrowth on watersheds 6 and 7 at Fernow, watershed 2 at Leading Ridge, and watershed 2 at Hubbard Brook (Table 1). In all cases, the effect was to prolong and substantially increase annual water yields compared to treated watersheds on which natural regrowth was uncontrolled. The use of herbicides for 3 successive growing seasons after felling all trees on watershed 2 at Hubbard Brook resulted in average annual yield increases of 288 mm for the 3-year period (Table 2). Herbicide applications to completely cleared watersheds at Fernow and Leading Ridge resulted in maximum annual increases of about 250 mm (Fig. 2, Table 2). Upon cessation of herbicide applications, sizable increases in water yield persisted for about 7 years at Fernow. However, natural regrowth quickly cut into increases in water yield at Hubbard Brook and Leading Ridge (Fig. 2).

Transpiration and interception by natural regrowth also quickly reduced increases in water yield in experiments where herbicides were not used. This was especially true for Hubbard Brook watersheds 4 and 5 where increases in water yield either disappeared or were greatly reduced within 3 or 4 years after cessation of intensive harvests (Fig. 2).

The increases appeared to decline equally rapidly on watershed 3 at Leading Ridge. Small decreases in water yield occurred by years 5 and 6 after harvest (Fig. 2). However, these decreases were anomalies resulting from a natural disturbance. During both of these years the mature forest on the control watershed was defoliated for part of the growing season by gypsy moth (*Porthetria dispar*). Such defoliations reduce transpiration and cause small increases in water yield (Corbett and Heilman 1975). Regeneration on the harvested watershed was not defoliated, resulting in greater transpiration from the harvested watershed and decreases in water yields compared to the control.

The decline with regrowth was less rapid on Fernow and Marcell watersheds (Fig. 2, Table 2). Yield increases persisted for two decades on watershed 2 at Fernow, but these were partly due to a follow-up diameter limit cut that was part of the prescribed treatment. Small increases of <50 mm yr^{-1} persisted for up to a decade or more after clearcuttings on watershed 4 at Marcell (Fig. 2). For years 12 through 21 since harvest at Marcell, the relatively small and inconsistent changes in water yield have been closely related to amount and distribution of spring and summer precipitation; increases in water yield occurred with above average precipitation and decreases occurred with drier weather conditions (Verry 1987). The same response was found for watershed 3 at Leading Ridge. The larger increases that occurred in years 10 and 14 after harvest (Fig. 2) were accompanied by growing season precipitation values that exceeded long-term means by more than 100 mm. Such findings reinforce an axiom expressed by Hewlett (1967) regarding forest cutting and increases in water yield: "It takes water to fetch water."

While above-average precipitation stimulates increases in water yield from cutting, excessive amounts can create problems with statistical procedures used in paired watershed studies. When precipitation and streamflow are well above the range normally encountered, calibration statistics must be extrapolated to accomodate resulting extremes in streamflow, providing a potential source of error in determining treatment effects (Hornbeck 1973). A case in point may be the 28th and 29th years after harvest on Fernow watershed 1. The increases in water yield were about the

same magnitude as those for the 1st and 2nd years after cutting (Fig. 2). Since the regrowing forest on watershed 1 is similar in species composition to that before harvest and there have been no recent changes in forest condition, there is no reason to expect such increases to suddenly occur late in the postcutting period. As it turns out, both the 28th and 29th years after harvest had extreme precipitation values. The calibration regressions had to be extrapolated substantially to test the resulting extremes in streamflow, and probably gave erroneous results. Obviously, the evaluation and interpretation of extreme events must be handled with caution.

Long-term Changes Related to Species Composition

Long-term changes of some consequence occurred on Fernow watersheds 6 and 7 and Hubbard Brook watersheds 2 and 4. The Fernow watersheds are recovering from clearcutting and several years of herbiciding. Watershed 6 also was planted to Norway spruce 9 years after the initiation of clearcutting, and herbicides subsequently were applied on 2 occasions to reduce competition to spruce (Table 1). Water yields had remained at elevated levels on both watersheds 6 and 7, with slightly higher values on watershed 6 (Fig. 2). In recent years, however, the spruce canopy on watershed 6 has begun to close, and water yield is showing strong indications of returning to and probably dropping below pretreatment levels (Fig. 2). This would not be unexpected for a hardwood to conifer conversion. Swank and others (1988) pointed out that transpiration and interception losses are greater from young conifers than mature hardwoods, particularly during the dormant season.

On Fernow watershed 7, a final herbicide application to kill back all regrowth took place 6 years after the initial cutting. The substantial increases in water yield (>250 mm yr^{-1}) declined over a 6-year period in relation to increasing dry-matter production of regrowth (Kochenderfer and Wendel 1983) to between 50 and 100 mm, where they persisted for 14 years (Fig. 2). Comparisons with Fernow watershed 3 suggest a possible explanation for this extended period of increases of 50 to 100 mm. After a series of selection and patch cuttings, watershed 3 was clearcut but not herbicided. Increases in water yield declined much more rapidly than on watershed 7 (Fig. 2). Species composition of regrowth is similar on both watersheds, but regrowth on watershed 3 consisted almost exclusively of sprouts. By contrast, herbicide applications on watershed 7 eliminated sprouts, and regrowth originated from seeds. By utilizing the rooting network from the previous forest, the regrowth composed of sprouts may have better access to soil moisture, and transpiration may be greater, at least during the first 15 to 20 years of stand establishment, than for regrowth originating from seeds. The final 2 years of water yield data from watershed 7 suggest a shift more in line with watershed 3, but additional data are needed to draw conclusions.

The long-term trends at Hubbard Brook are different from those at the other three study locations. Decreases in water yield were evident early in the regrowth phase on watersheds 2 and 4, and persist through the remaining 13 years of record on both watersheds (Fig. 2). The explanation may lie with a posttreatment change in species composition. Before treatment, basal area was distributed about evenly among beech, birch, and maple species. During regrowth, nearly 80% of the basal area has been in birch and pin cherry (a common pioneer species in northern hardwood forests) with the remainder divided between beech and maple. Federer (1977) showed that birch and pin cherry have significantly lower leaf resistances (3.2 s cm^{-1}) than beech and maple species (4.0 to 4.5 s cm^{-1}). Thus, transpiration may be greater from the regrowing stand dominated by birch and pin cherry than from the mature, undisturbed forest, the end result being less water available for streamflow.

DISCUSSION

Surface Water Supplies

Results from the four study sites indicate the potential to increase water yield from forested watersheds in the northeastern United States. Based on experiments at Hubbard Brook, Fernow, and Leading Ridge, in which watersheds were clearcut and then herbicided, the maximum possible increase is in the range of 250 to 300 mm yr^{-1}. However, in light of controversy over the use of herbicides, it is likely that attempts to increase water yield will be confined to cutting. Even then, the studies indicate that various sizes of clearcuts, without control of regrowth, can

provide immediate increases in annual yields ranging from about 110 mm (at Marcell) to 150 mm (at Hubbard Brook and Leading Ridge) to 250 mm (at Fernow). However, such increases diminish fairly rapidly, more so in some areas (Hubbard Brook and Leading Ridge) than others (Fernow and Marcell).

When cutting forests with an objective of increasing water yields, one must consider the possible impacts of a change in species during regrowth. The long-term results from Fernow and Hubbard Brook show that desired increases in water yield occurring immediately after cutting may be compensated in later years if hardwoods are converted to softwoods, or if there is a major shift in composition of hardwood species.

It is clear that the prolonged increases in water yield that occur after cutting in other regions of United States, such as from deeper soils of the southeast (Swank and others 1988) or from slowly regenerating forests of the west (Troendle and King 1985), cannot be expected in the Northeast. Shallow soils and rooting depths, shorter growing seasons, rapid root occupancy and leaf-area development by natural regeneration, lower evapotranspiration, and complete recharge of soil moisture during every dormant season all act to limit the magnitude and duration of increases in water yield in the Northeast.

Global Climate Change

The potential for a gradual change in species composition of forests is a major concern related to global climate change (Roberts 1989). The Hubbard Brook findings have implications regarding this concern. If one or two species were to drop out of the current hardwood forest, there could be detectable impacts on water yield. For example, the replacement of beech and maple at Hubbard Brook with birch and cherry resulted in decreases in water yield that averaged about 50 mm yr^{-1}. Although these decreases are small, they could become important if global climate change included a decrease in precipitation or conditions that favor increased evapotranspiration.

Nutrient Cycling

Forest cutting affects many processes involved in nutrient cycles and can lead to mobilization and increased leaching of nutrients (Hornbeck and others 1987). The mobilization and leaching of nutrients usually coincide with maximum increases in water yield. Thus, the larger the increases in water yield, the greater the potential to transport an additional mass of nutrients from cutover watersheds.

Forest cutting had negligible effects on nutrient leaching to streams at Marcell (Verry 1972), Fernow (Aubertin and Patric 1974), and Leading Ridge (Lynch and Corbett 1990), but caused significant increases at Hubbard Brook (Bormann and others 1968, Hornbeck and others 1987). In the 7 years after the clearfelling and herbicide experiment on watershed 2 at Hubbard Brook, increased leaching losses of nitrogen in streams represented a loss of nearly one-fourth of the total nitrogen capital of the watershed. A significant portion of this nitrogen loss was transported by the increased water yields that occurred in response to cutting and herbicide applications. By contrast, leaching losses of nitrogen after the less drastic strip cutting of watershed 4 represented <1% of total capital (Hornbeck and others 1987). Part of the explanation for this reduced loss lies with the much smaller increases in water yield, and less opportunity for nitrogen to be transported from the watersheds. Before recommendations are made to increase water yields from forests where nutrient leaching may be a problem, the potential impacts of added nutrient losses on site productivity and water quality must be considered.

Hydrologic Modeling

A primary objective of watershed studies is to provide data for developing and testing hydrologic models. The variety of responses to treatments in the northeastern United States suggests why it has been difficult to obtain good simulations of changes in water yield, especially those that are long term and more subtle.

The role of changes in species composition may have to be simulated more carefully. At present, some forest hydrology models use leaf area as the primary parameter for governing transpiration rates and water yield responses

after treatment. As leaf area increases to an established level (e.g., 4 ha ha^{-1}), water yields gradually are returned to pretreatment levels. However, as suggested by long-term results at Fernow and Hubbard Brook, the changes in water yield might be more appropriately modeled with parameters such as leaf resistance, sapwood area, or indicators of leaf and needle geometry. Such parameters might allow better simulation of the processes that eventually resulted in long-term decreases in water yield at Fernow and Hubbard Brook.

Long-Term Research.

The knowledge summarized in this paper was obtained as a result of a continuing commitment to long-term research. Counting calibration periods, the studies have spanned at least 3 decades at Marcell, Leading Ridge, and Hubbard Brook, and 4 decades at Fernow. The reward for this long-term commitment is a more complete understanding of the impacts of forests and associated treatments on the hydrologic cycle. Watershed studies have taken on an added dimension over the past 2 decades as they have been expanded into ecosystem studies (Hornbeck and Swank 1992). The merging of forest hydrology with ecosystem studies ensures that watershed studies will continue as a primary source of knowledge about the role of forests in the hydrologic cycle.

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SEASONAL ISOTOPE HYDROLOGY OF APPALACHIAN FOREST CATCHMENTS

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Abstract: Seasonal hydrologic behavior of small forested catchments in the Appalachians was studied using oxygen-18 as a tracer. Oxygen-18 in samples of precipitation and streamflow were used to determine seasonal variations of subsurface water recharge and movement within two 30-40 ha forest catchments (Watershed 3 and 4) at the Fernow Experimental Forest in northcentral West Virginia and a 1,100-ha forested catchment in northcentral Pennsylvania for the period March 1989-March 1990. Precipitation or throughfall in both regions showed expected seasonal sine-wave variations in oxygen-18 with minima occurring in winter and maxima occurring in summer. Baseflow stream samples for both small watersheds at Fernow showed similar seasonal sine-wave variations in oxygen-18 with amplitudes damped by about 90% and time lags to peak oxygen-18 of about 110 days relative to precipitation variations. Damping of amplitudes on the Fernow basins was equivalent to a mean transit time for subsurface baseflow water of about 1.5 years. Baseflow oxygen-18 on the larger Pennsylvania basin showed no discernible seasonal pattern in oxygen-18, which indicated mean transit time of subsurface water was at least 3-5 years. These results imply that both small Fernow basins exhibit relatively shallow and rapid recycling of subsurface water and that both basins would respond quickly and similarly to changes in watershed condition. On the larger Pennsylvania basin, a much larger, more slowly-cycling groundwater reservoir exists and baseflow would respond much more slowly to changes in watershed conditions.

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SPATIAL CHARACTERISTICS OF TOPOGRAPHY, ENERGY EXCHANGE, AND FOREST COVER IN A CENTRAL APPALACHIAN WATERSHED

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Abstract: Spatial variation of topography, net radiation, evapotranspiration, and forest stand in the central Appalachian watershed is described. The study area is the control watershed 4 (39°20'N,79°49"W) located in the Fernow Experimental Forest at Parsons, West Virginia. The watershed encompasses an area of 39.2 ha, it has a south-east orientation, and the average slope inclination is 14°. The forest cover is ca. 85 years old and consists of upland oak and cove hardwoods. Topographic analysis was based on data for 432 triangular segments with an average area of 360 m^2, covering the whole watershed. Partial areas for defined slope and azimuth ranges and the distribution of both parameters are illustrated. Half of the watershed area has an azimuth between 90 and 150° (east facing slopes), and 65% of the area has slope inclination ranging from 10 to 20°. Net radiation (Rn) was computed for all terrain segments. Its distribution in the watershed is illustrated, and its average yearly sum for the whole watershed was 2.2 GJ m^{-2}. Yearly sum of Rn of southwest facing slopes was 55% (lower sites) to 60% (upper sites) greater than that of east facing slopes. The average yearly precipitation (P) and evapotranspiration (Et) of the watershed are 145.5 cm and 81.7 cm, respectively. A regression formula defines yearly sum of Et as a function of P and Rn of the watershed. Using this formula, average yearly sums of Et of all terrain segments were calculated. The distribution of the yearly average sum of Et in the watershed is illustrated; yearly Et of partial areas varied from ca. 60 to 85 cm. The average air-dry above ground biomass for 112 plots was 320.3 t ha^{-1}. East facing slopes had the highest air-dry biomass (354 t ha^{-1}) and the southwest facing slopes the lowest (224 t ha^{-1}). Thirty-five species were recorded on the plots surveyed. The most frequent species were sugar maple (32.1% of the total number of trees, DBH > 5 cm), red maple (19.3%), American beech (9.7%), northern red oak (7.6%), black cherry (4.1%), sweet birch (3.6%), and chestnut oak (2.4%). Red oak had the highest total biomass (30.2% of the total for all species), followed by sugar maple (13.8%), black cherry (13.0%), red maple (9.9%), chestnut oak (6.9%), yellow-poplar (5.7%), American beech (4.7%), and white oak (4.5%).

INTRODUCTION

On a broad scale, the major plant associations of the central Appalachian region are oak forests and northern hardwoods. The relationship between forest and climate of the region is well established. Regional energy-water relations for a horizontal surface - are defined by Hare [1972 (in Miller, 1977)]. However, on smaller scales, there is a wide variety of vegetation types that, according to Rumney, (1968) is a natural consequence of variations in topography, weather patterns and underlying bedrock. There are numerous descriptions of forests in the region (e.g., Hack and Goodlett, 1960; Trimble, 1973; Burns, 1983). However, for most forest sites, quantitative information on topography related variation in meteorologic, hydrologic, and other parameters is missing. Foresters correlate forest growth in the region with such parameters as slope aspect and inclination, soil depth and fertility, precipitation, and stone content (Trimble and Weitzman, 1956; Yawney, 1964; Yawney and Trimble, 1968; Auchmoody and Smith, 1979). However, Carmean (1975) remarked that "correlations cannot be accepted as evidence of cause and effect relations" and that "features of soil, topography, and climate found to be correlated with site index are indirect indices

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of more basic growth controlling factors and conditions, such as available moisture and nutrients, and microclimatic factors that affect evapotranspiration and tree physiological processes". This calls for the understanding of ecological processes at specific forest sites and is compatible with geophysics of landscapes and with landscape ecology (Armand, 1964; Forman, 1983; Swanson et. al. 1988).

Among many processes that take place in the forest, cycles of water and energy play a key role. Hydrologic studies at the watershed level yield information on average values of the water balance components. The number of such studies is limited, and in addition the water balance components so obtained cannot be related to specific forest sites with varying topography and location in the watershed. Net radiation of the forest is regarded as the main source of energy needed for the processes of growth, etc. But, studies dealing with the distribution of net radiation in forested watersheds are infrequent (Tajchman et al. 1988; Fu et al. 1995).

For this particular study, we have selected a forested watershed (watershed 4) in the Fernow Experimental Forest at Parsons, West Virginia, which has been the subject of hydrologic and meteorologic observations for more than 40 years (Adams et al., 1993). The objectives of the study were to obtain a detailed analysis of a) topography, b) energy and water exchange, and c) forest cover.

STUDY SITE

The study area, the control watershed 4, is shown Fig. 1. The watershed area is 39.2 ha, the average watershed elevation is 804 m with the maximum of 869 m and minimum of 739 m. The watershed has a southeast orientation and ca. 65% of its area has the azimuth ranging from 90° (east facing) to 180° (south facing). Slope inclination ranges from 2 to 29° and the average value is 14°. The predominant soil is Calvin silt loam with considerable stone content. Soil depth to bedrock averages 0.8 m, ranging from 0.56 to 1.2 m. About 95% of the tree roots were found in the upper 0.90 m layer of the soil (Patric, 1973; Kochenderfer et al. 1987).

The area was heavily logged between 1905 and 1910; no surface disturbance has been permitted since that time, except for a road constructed adjacent to its upper boundary in the 1930's. The major forest types in the watershed are upland oaks and cove hardwoods. The upland oak type occurs on drier areas and consists primarily of red oak (*Quercus rubra*), chestnut oak (*Quercus primusprinus)*, and white oak (*Quercus alba*). Cove hardwood type occupies moist sites along the streams and consists mainly of sugar maple (*Acer saccharum*), and black cherry (*Prunus serotina*), frequently including yellow-poplar (*Liriodendron tulipifera*) and scattered American beech (*Fagus grandifolia*).

The growing season is May through September, and average frost-free season is 145 days (Patric, 1973). Precipitation is evenly distributed throughout the year, and the yearly average is 145.5 cm. About 44% (64.1 cm) of total precipitation occurs during the growing season. The months of June and July have the greatest average precipitation, 14.4 and 13.6 cm, respectively. September and October are the driest months, with average precipitation of 10.3 cm and 9.7 cm, respectively. The yearly average temperature in the study area is 8.8°C. July and August have the highest average monthly temperatures of 19.3 and 18.7°C, respectively. January has the lowest monthly average temperature of -3.1°C.

METHODS

Topography

For spatial analysis of the watershed, its photogrammetric map (Greenhorne and O'Mara, Inc.) in scale 1:2,500 was considered in the three dimensional coordinate system with x-axis directed toward east, y-axis directed toward north, and z-axis directed toward zenith. The watershed area was divided into 432 triangular segments with an average area of 360 m². Using the Summagraphics Microgrid II digitizer, the x and y coordinates of the corners of all triangles

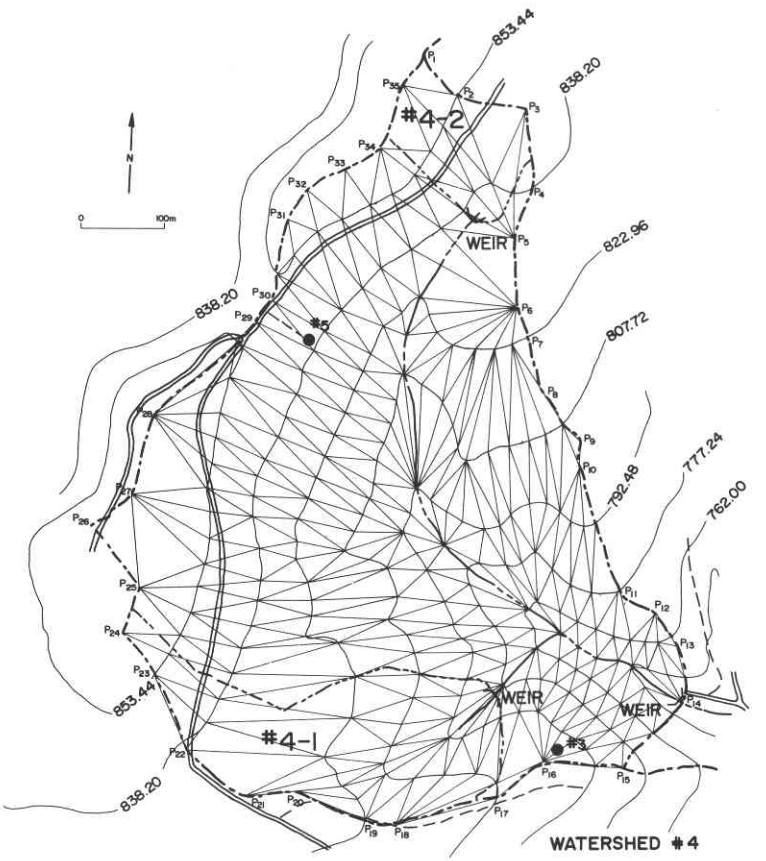

Figure 1. Topographic map and triangulated network of Watershed 4 (Fu et al., 1995).

were determined; the elevation marked on contour lines was accepted for the z-coordinate. The x,y,z coordinates of the corners of triangles were used to compute their azimuth, inclination, and area (Tajchman, 1975).

Water and Energy Balances

The following formula defines the water balance of a watershed

$$P = Et + R + \Delta W, \quad (1)$$

where P is precipitation of the watershed averaged over its area, Et is evapotranspiration of the watershed, R is runoff, and ΔW represents the change of the soil water content in the watershed during the period of observation. For periods starting and ending with the same soil water content $\Delta W = O$ and the water balance of the watershed is given by

$$P = Et + R. \quad (2)$$

The parameters P and R are known from observation and Et can be obtained from Eq. (2).

The energy balance of the watershed, after neglecting the relatively small amounts of energy used in photosynthesis and energy stored in the soil, is given by

$$Rn = L\,Et + H, \tag{3}$$

where Rn is net radiation of the watershed, L is the latent heat of evaporation of water, and H is the heat exchange between the watershed and the atmosphere. In calculating the yearly energy balance of the watershed, the latent heat of Et obtained from Eq. (2) can be applied. Calculations were carried out for 39 hydrologic years (1951-1990) which start on May 1 of a calendar year and ends on April 30 of the next calendar year.

The net radiation of the watershed is given by

$$Rn = G\,(1 - r) + Rl, \tag{4}$$

where G is global radiation, r is the reflectivity of the forest for solar radiation and Rl is the longwave radiation balance. The average r values of 0.17 and 0.20 were accepted for the growing season and for the dormant season, respectively (DeWalle and McGuire, 1973; Lee and Sypolt, 1974). The term G (1 - r) represents the amount of solar radiation absorbed by the forest. Net radiation data for the watershed are not available from direct observation. However, they can be calculated for all terrain segments and then for the whole watershed area if the following parameters are known:

a) Monthly sums of global radiation at a horizontal surface in the study area,
b) Climatological data on air temperature, humidity, and sky cover,
c) Topographic parameters of terrain segments including azimuth, inclination, and the view factor,
d) The times of sunrise and sunset of all terrain segments.

Using monthly sums of global radiation (horizontal surface) for Parsons (1965-1977) and the monthly average sky cover data for Elkins, ca. 24 km south-east of Parsons, the following regression formula was obtained

$$G = Go\,(0.82 - \alpha\,C),\ R^2 = 0.87, \tag{5}$$

where Go is the monthly sum of extraterrestrial radiation, and C is average monthly sky cover in fractions of unity. Monthly values of the coefficient α are listed in Table 1. Eq. (5) was used to calculate the missing data on global radiation at Parsons during the period 1951-1990.

Table 1. Monthly values of α (Eq. 5) at Parsons, W.Va.

Month	α	Month	α
January	0.598	July	0.514
February	0.562	August	0.516
March	0.546	September	0.534
April	0.576	October	0.613
May	0.537	November	0.639
June	0.500	December	0.622

The following regression formula describes yearly sums of Et as functions of the corresponding P and Rn.

$$Et = 0.2052\ P + 5841\ (Rn/L),\ R^2 = 0.82, \quad (6)$$

where Et and P are in cm, Rn in kJ m^{-2}, and L is the latent heat of evaporation of water. Assuming that the same relationship exists between long term average monthly sums of Et, P, and Rn, Eq. (6) was used to calculate 39 years average monthly values of Et of all terrain segments. Then, yearly sums of Et were obtained. The standard deviation of the difference between measured and calculated yearly sums of Et was 1.85 cm, and the relative error ranged from 0 to 7.8% (average = 2.8%). For more details see Fu (1992).

Above Ground Biomass and Species Composition

Data on DBH and species composition were collected during the summer of 1993 on eighty seven 804 m^2 (93 x 93 ft.) plots randomly distributed in the watershed. All trees with DBH > 5 cm were recorded. Additional data on DBH and species composition in the watershed, collected in 1990 at 25 809 m^2 (0.2 acre) plots randomly distributed in the watershed, were included in the study.

The location of the plots is seen in Fig. 2. The above ground air-dry biomass of single trees at each plot was obtained using the relationships reported by Brenneman et al. (1978).

RESULTS AND DISCUSSION

Topographic parameters were computed for all terrain segments and were used to obtain a) partial areas of the watershed with definite ranges of inclination and azimuth, and b) maps showing the distribution of inclination and azimuth in the watershed.

Figures 3 and 4 show partial areas of the watershed in percent of its total area within different intervals of inclination and azimuth. As shown in Fig. 3, slopes with the inclination ranging from 10 to 15° occupy 42% and those with inclination ranging from 25 to 30° occupy only 3% of the total watershed area. Fig. 4 shows, e.g., that, east and south-east facing slopes with the azimuth ranging from 90 to 150° occupy about 50% of the watershed area, and the north facing slopes with the azimuth ranging from 330 to 30° occupy about 7% of the total watershed area. Since site quality is related to aspect, knowing of the relative amount of area in each aspect can be useful in the preliminary evaluation of forest land in the study area.

The distribution of slope inclination and azimuth in the watershed is seen in Figs. 5 and 6, respectively. The western half of the watershed contains more or less topographically uniform areas with inclination ranging from 8 to 12° (Fig. 5), and with azimuth ranging from 100 to 140° (Fig. 6). The eastern half of the watershed is characterized by a more complex topography with inclination of partial areas ranging from 6° in the north-eastern part of the watershed to 26° in the south-eastern part of the watershed. The prevailing aspect in this part of the watershed is south and south-west.

Topography affects the radiation exchange in complex terrain. This is seen in Fig. 7 where the distribution of the average yearly sums of net radiation in the watershed is marked by isolines. In the western half of the watershed the net radiation is more or less uniformly distributed, and in the eastern half a variation is substantial. In the western half of the watershed the yearly sum of net radiation amounts to approximately 2.20 GJ m^2, and in the eastern part it varies from approximately 1.60 GJ m^{-2} to 2.20 GJ m^{-2}. The yearly average net radiation of the whole watershed is 2.20 GJ m^{-2}.

The evapotranspiration of the watershed is a component of its water and energy balances (Eqs. 1 and 3). Its average value for the period 1951-1990 was 81.7 cm, and the equivalent latent heat amounted to 2.02 GJ m^{-2}. One can determine from Eq. (3), that the yearly average sum of the sensible heat transferred from the watershed to the atmosphere amounted to 0.18 GJ m^{-2} or 8.2% of Rn.

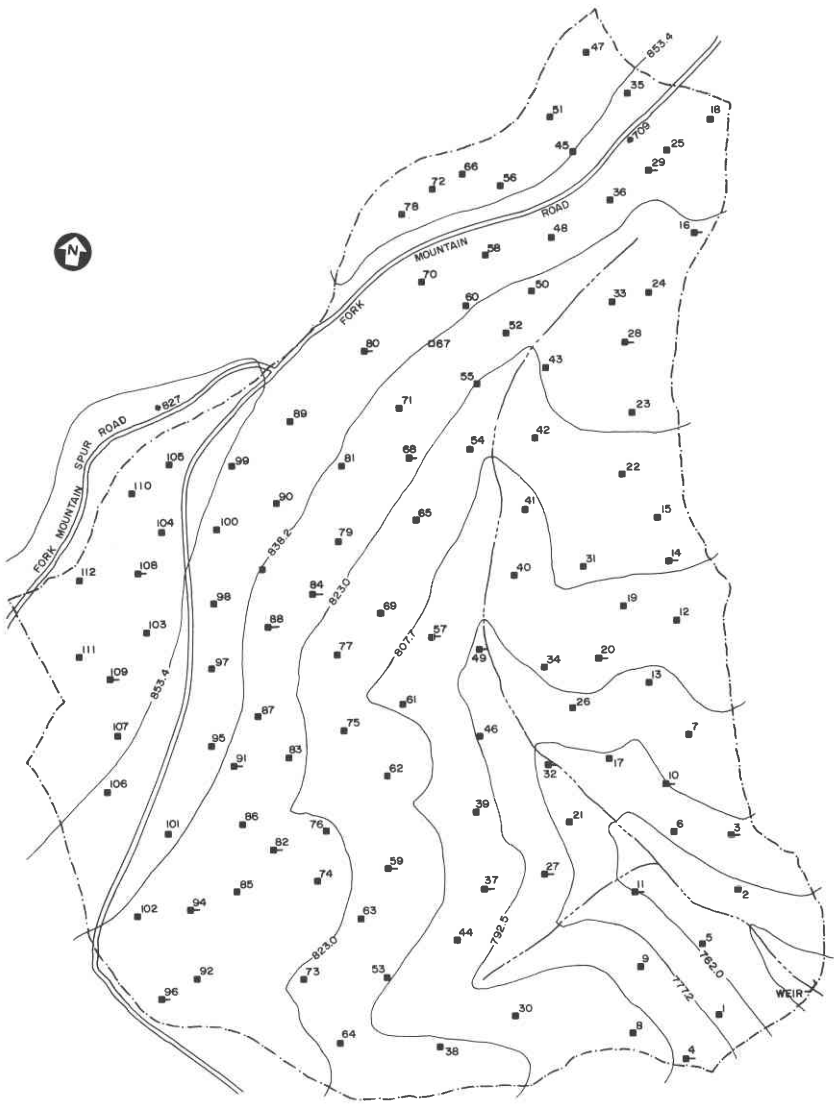

Figure 2. Location of sampling plots. Squares with a dash represent permanent plots of the U.S. Forest Service.

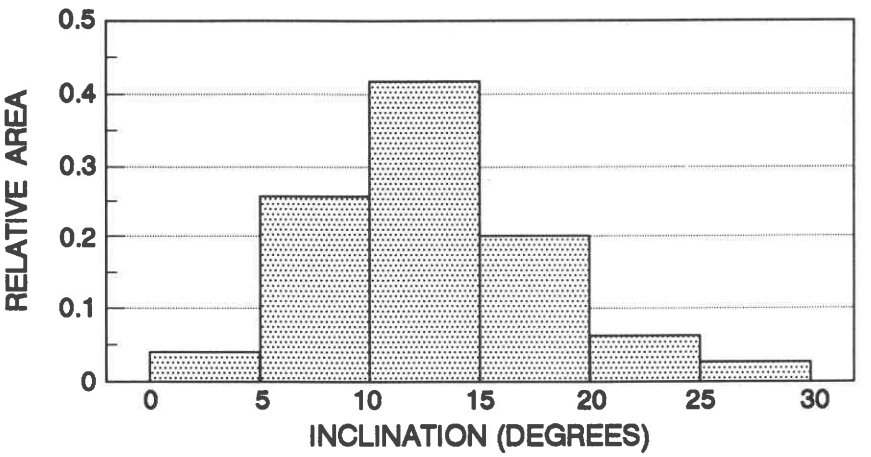

Figure 3. Ranges of slope inclination and the corresponding partial areas (%) of the watershed.

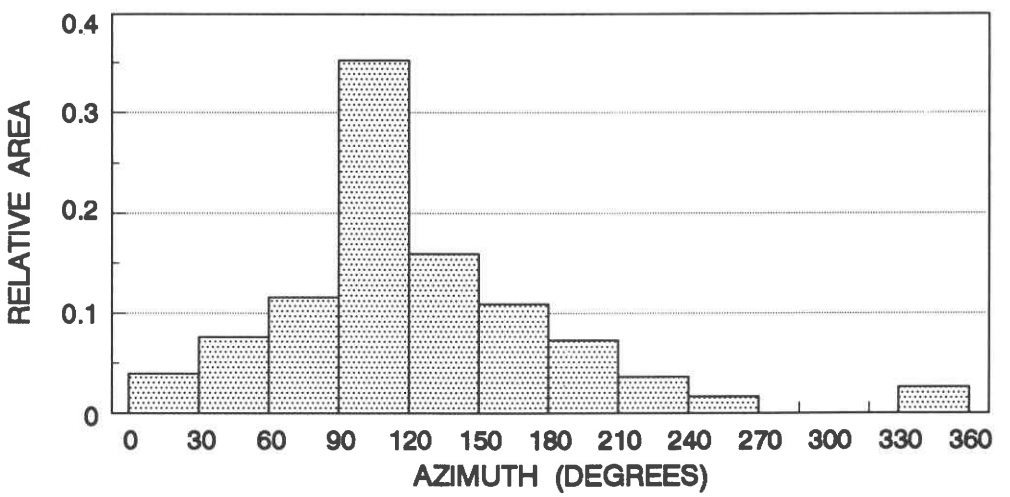

Figure 4. Ranges of slope azimuth and the corresponding partial areas (%) of the watershed.

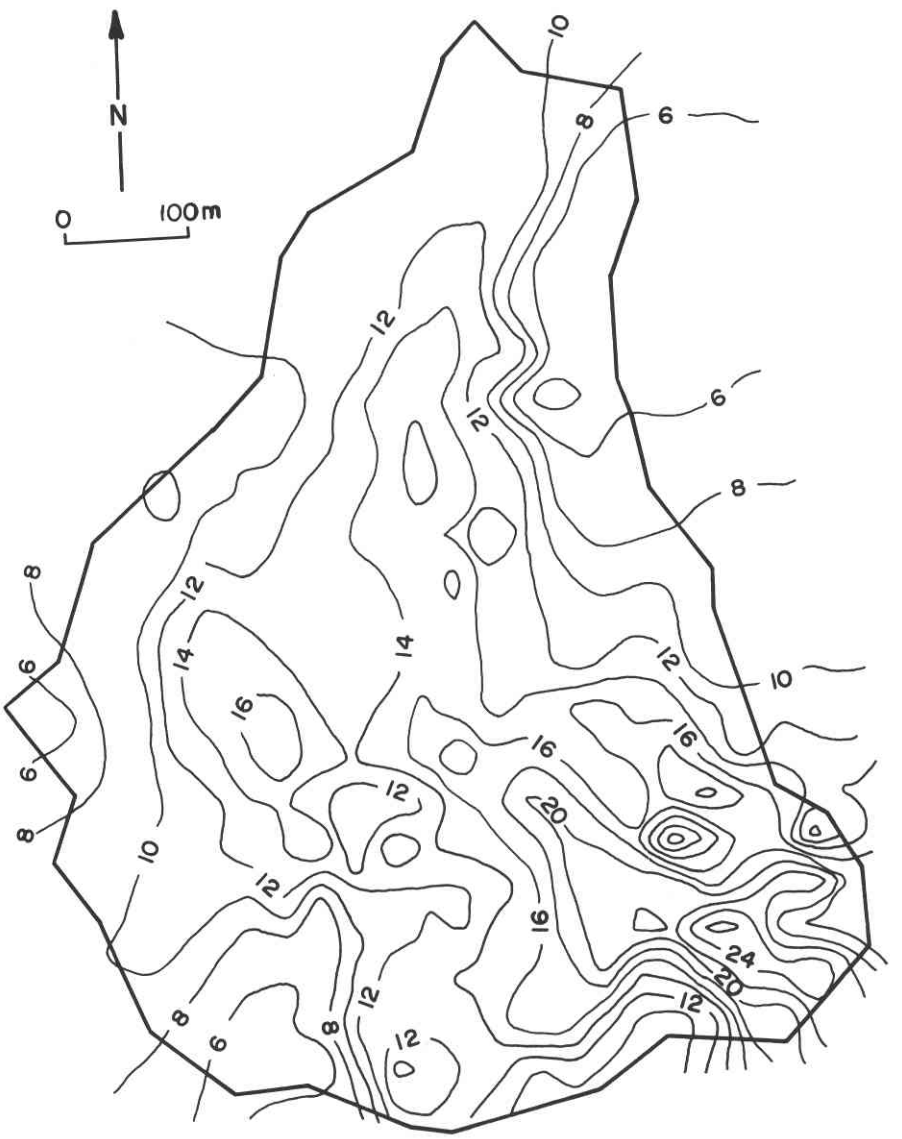

Figure 5. Distribution of slope inclination (degrees) in Watershed 4 (Fu et al., 1995).

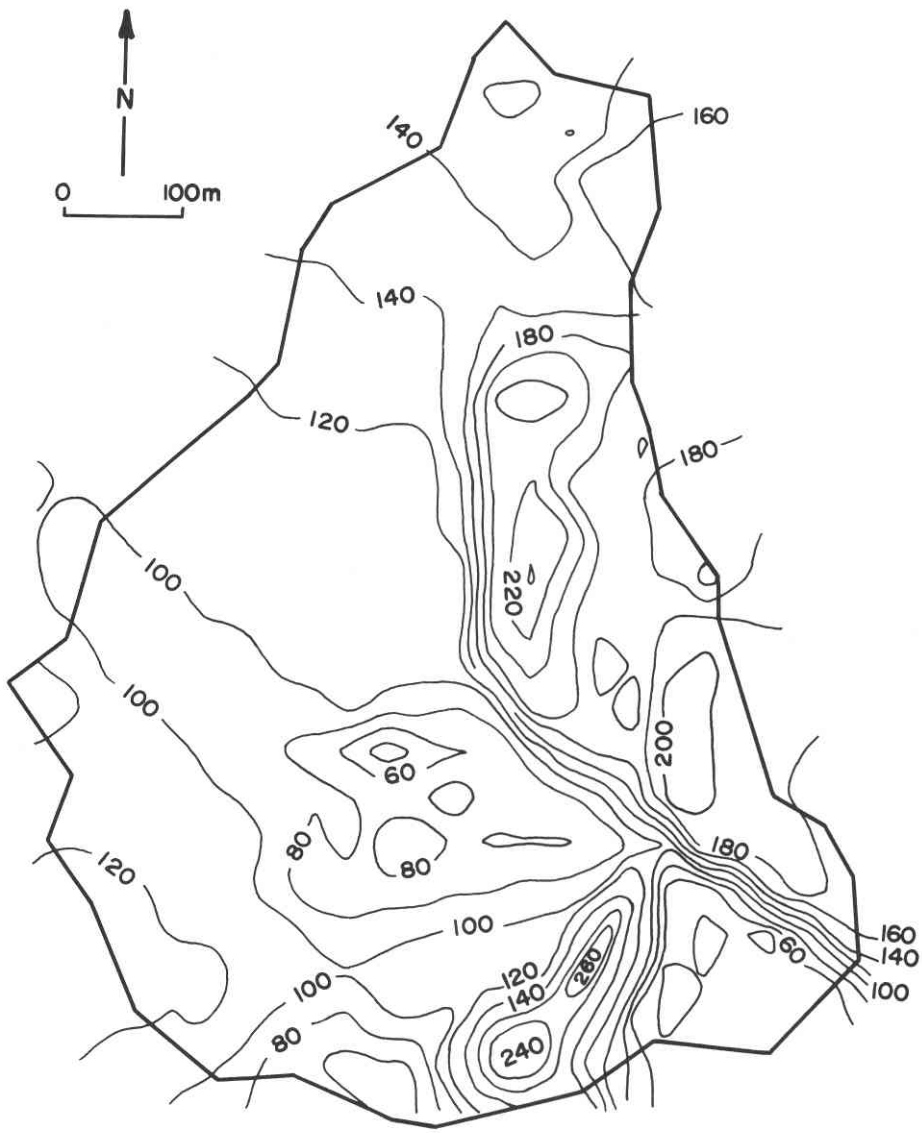

Figure 6. Distribution of slope azimuth (degrees) in Watershed 4 (Fu et al., 1995).

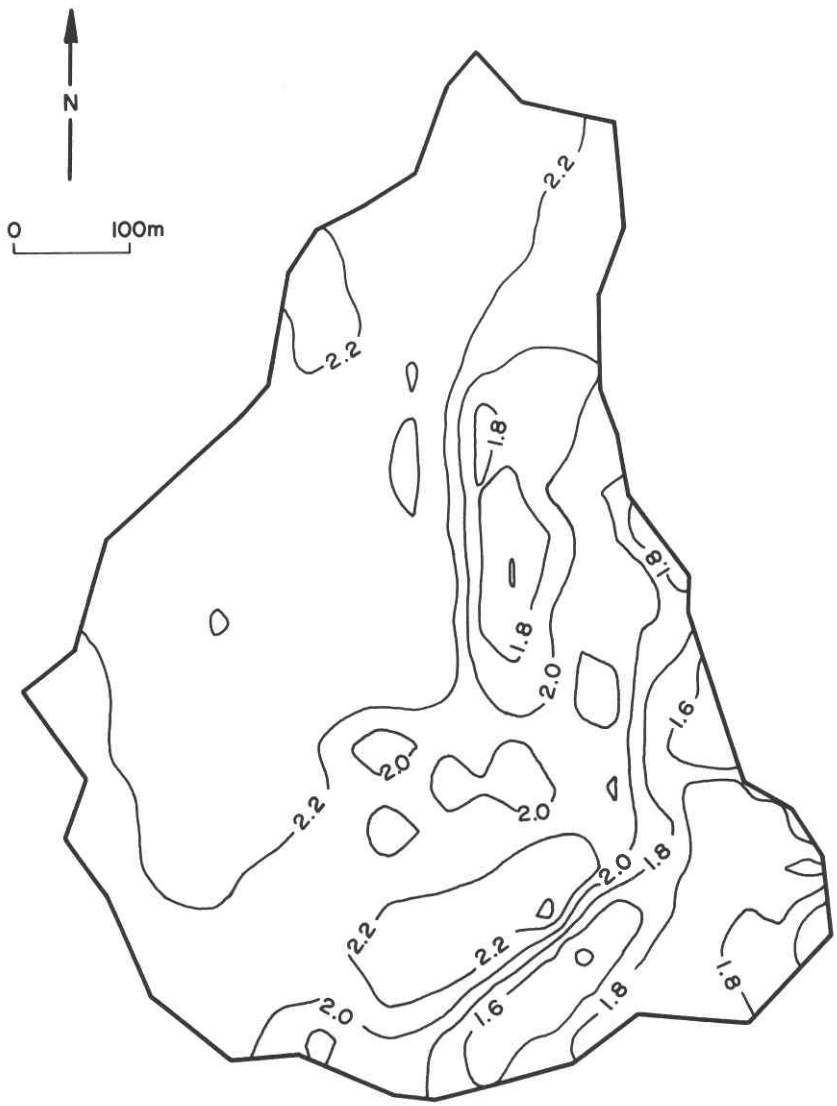

Figure 7. Distribution of the yearly sums of net radiation (G J m^{-2}) (Fu et al., 1995).

Applying Eq. (6) to all terrain segments, their Et values were obtained. A linear interpolation was applied to obtain the distribution of Et values in the watershed, represented by isolines on the map of the watershed (Fig. 8). In the western half of the watershed, Et varies from 82 to 85 cm per year and is uniformly distributed. In the eastern half of the watershed, the yearly average sum of Et varies from 60 to 82 cm, and its distribution is complex.

Figure 9 shows the location of plots where trees were sampled, and the numbers at each plot express the above ground air dry biomass in tons per hectare. A total of 35 species were found on the plots (Table 2). The average biomass for 112 plots is 320.3 t ha^{-1} . The difference between the maximum and minimum biomass of single plots is equivalent to 152% of the average value. The coefficient of variation of single plot biomass decreases when the number of plots (n) increases, and reaches a more or less steady value when $n \geq 20$.

Plot biomass values were compared to those of Rn and Et, and the possible relationship between the biomass and two other parameters was examined. In the eastern part of the watershed with a more complex topography, the coefficient of variation of the plot biomass is 29.9%; the corresponding coefficients of variation for Rn and Et are 8.9% and 6.4%, respectively. In the western part of the watershed, with more uniform topography, the coefficient of variation for plot biomass is 25.3%, and those for Rn and Et are 3.1 and 2.4%, respectively. The correlation between the above ground biomass and Rn and Et is illustrated in Table 3.

The above data (Table 3) show a poor correlation between biomass and the two other parameters for single plots. However, the correlation coefficient increases with number of plots averaged, and it exceeds 0.8 for $n \geq 15$.

The average biomass for different ranges of azimuth is given in Table 4.

For the azimuth ranges 0 - 30°, 210 - 240°, and 240 - 270° only 3, 5, and 2 plots are available, respectively. Data for these azimuth ranges should not be regarded as representative. Sites with azimuth ranging from 90 to 120° have the highest average above ground biomass. For the azimuth values decreasing or increasing from this azimuth range the average above ground biomass decreases (Table 4).

Species found on the plots surveyed are listed in Table 5 together with the number of trees and air dry biomass for each species and the corresponding percentages of the totals. According to Table V, the most prevalent species in the watershed based on number of trees are Sugar maple (32.1%), Red maple (19.3%), American beech (9.7%), Northern red oak (7.6%), Black cherry (4.1%), Sweet birch (3.6%), Chestnut oak (2.4%), Downy serviceberry (2.4%), and Yellow-poplar (1.8%). Each of the remaining species accounts for less than 1.5% of the total. The following eight species account for 88.7% of the total biomass: northern red oak (30.2%), sugar maple (13.8%), black cherry (13.0%), red maple (9.9%), chestnut oak (6.9%), yellow-poplar (5.7%), American beech (4.7%), and white oak (4.5%).

Species frequency and stand composition vary with site azimuth. Some typical examples are listed in Table 6.

CONCLUSIONS

Future improvement in ecological forest management will depend on progress in understanding processes and interactions which take place in the forest. This particular study shows results of a three-dimensional analysis of topography, radiation exchange, and evapotranspiration in a forested watershed. This is probably the first attempt to calculate the distribution of evapotranspiration in a forested watershed. However, our results on Et should be verified in independent research, e.g., on isolated plots in the watershed. The distribution of the 39-year average yearly sum of Et was obtained under the assumption that for an "average year" Eq. (6) applies to all terrain segments, and that soil conditions and forest cover are uniform. Data on spatial distribution of soil horizon thicknesses, stoniness, organic matter content, and stand density (which could be described in the x, y, z - coordinate system) would allow modifications to Eq. (6) for specific terrain segments.

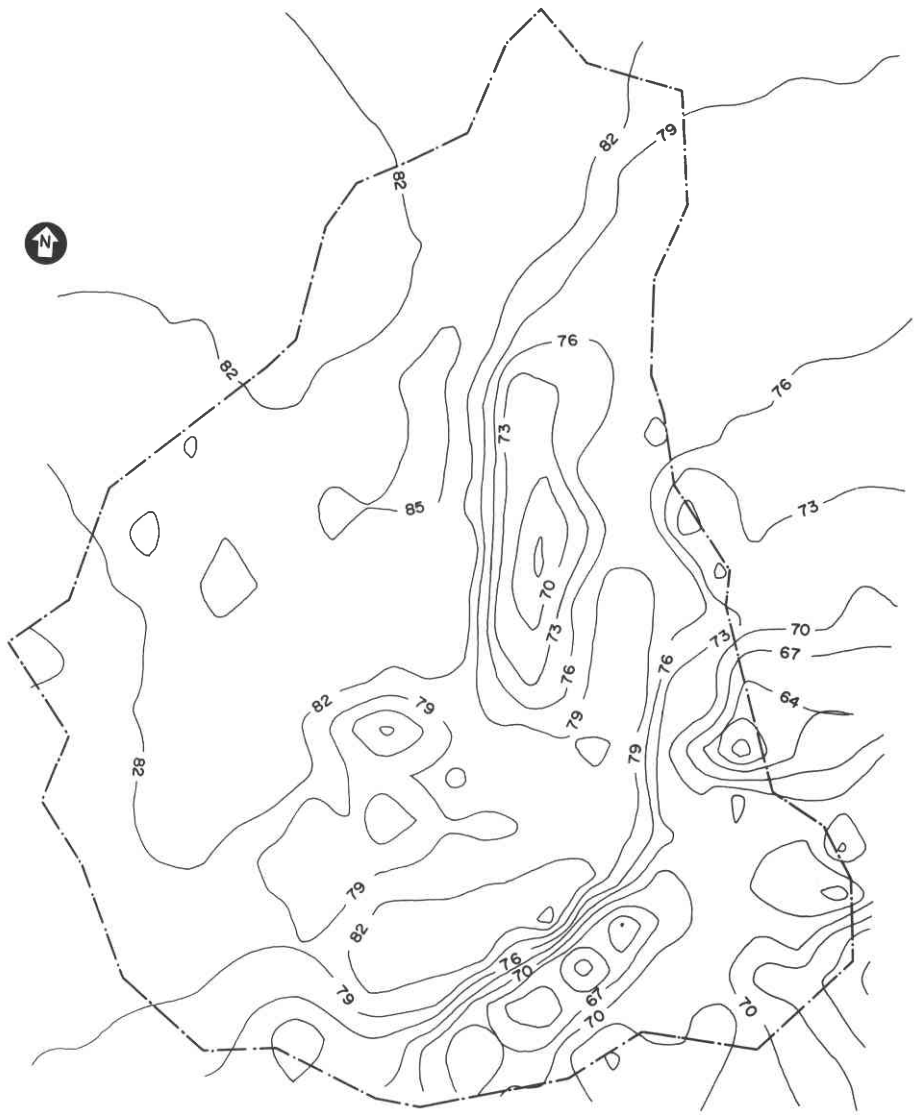

Figure 8. Distribution of the yearly sums of evapotranspiration (cm) (Fu, 1992).

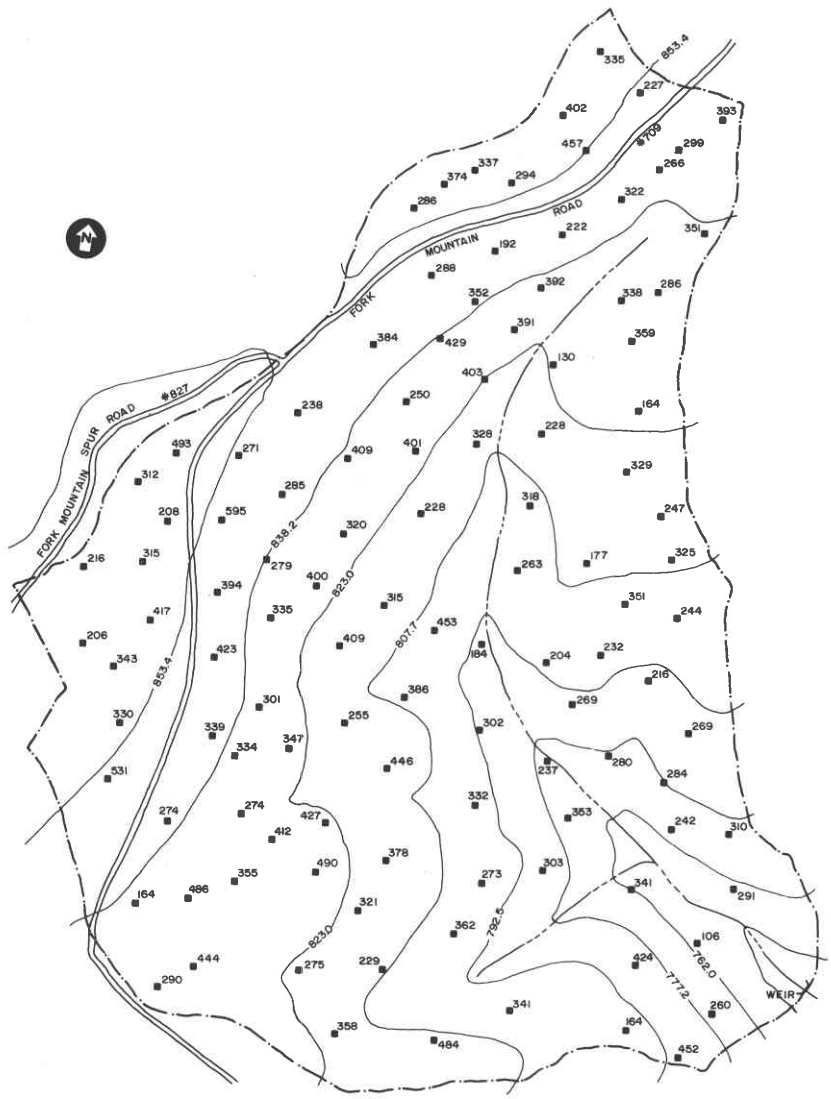

Figure 9. Above ground air-dry biomass at sampling plots.

Table 2. Species found in Watershed 4.

Common Name	Scientitic Name
Ailanthus	*Ailanthus Altissma* (Mill.) Swingle
American basswood	*Tilia Americana* L.
American beech	*Fagus grandifolia* Ehrh.
American chestnut	*Castanea dentata* (Marsh.) Borkh.
Bigtooth aspen	*Populus gradidentata* Michx.
Blackgum	*Nyssa sylvatica* Marsh.
Black locust	*Robinia Psudoacacia* L.
Cherry	
Black	*Prunus serotina* Ehrh.
Pin	*Prunus pensylvanica* L.f.
Cucumbertree	*Magnolia acuminata* L
Deciduous holly	*Ilex decidua* Walt.
Downy serviceberry	*Amelanchier arborea* (Michx. f.)Fern.
Flowering dogwood	*Cornus florida* L.
Fraser magnolia	*Magnolia fraseri* Walt.
Grapevine	*Vitis* L.
Hawthorn	*Crataegus* L.
Hickory	
Bitternut	*Carya cordiformis* (Wangenh.) K. Koch
Shagbark	*Cary ovata* (Mill.) K. Koch
Eastern hophornbeam	*Ostrya virginiana* (Mill.) K. Koch
Maple	
Mountain	*Acer spicatum* Lam.
Red	*Acer rubrum* L.
Striped	*Acer pensylvanicum* L.
Sugar	*Acer saccharum* Marsh.
Oak	
Black	*Quercus velutina* Lam.
Chestnut	*Quercus prinus* L.
Northern red	*Quercus rubra* L.
Scarlet	*Quercus coccinea* Muenchh.
White	*Quercus Alba* L.
Sassafras	*Sassafras albidum* (Nutt.) Nees.
Sourwood	*Oxydendron arboreum* (L.) DC.
Sweet birch	*Betula lenta* L.
White ash	*Fraxinus americana* L.
Witch hazel	*Hamamilis virginiana* L.
Yellow-poplar	*Liriodendron tulipifera* L.

Table 3. The correlation coefficient for above ground biomass, net radiation (Rn) and evapotranspiration (Et) for single plots and for different numbers of plots averaged.

	Biomass					
	Single plots	Number of plots averaged (n)				
		5	10	15	18	28
Rn	0.21	0.54	0.78	0.86	0.89	0.92
EIT	0.18	0.57	0.75	0.83	0.82	0.89

Table 4. Average above ground biomass in watershed 4 for azimuth ranges.

Azimuth (degrees)	Number of plots	Average biomass (t ha)
0 - 30	3	263.0
30 - 60	10	312.3
60 - 90	14	335.8
90 - 120	33	353.7
120 - 150	22	327.4
150 - 180	14	294.3
180 - 210	9	285.1
210 - 240	5	223.9
240 - 270	2	290.6

Table 6. Azimuth of maximum frequency for different species in watershed 4.

Northeast facing	East facing	South facing	Southwest facing
American beech	American basswood	Black gum	
Striped maple	Black cherry	Chestnut oak	
	Black locust	Downy serviceberry	
	Eastern hophornbean		
	Mountain maple		
	N. red oak		
	Sugar maple		Cucumber tree
	Sweet birch		Fraser magnolia
	White ash		White oak
	Witch hazel		
	Yellow-poplar		

Red maple has a relatively high frequency within the azimuth range of 80 - 200° .

Table 5. Species, number of trees with DBH > 5 cm, and the above ground air-dry biomass for 112 plots in watershed 4.

Species	Number of trees	Percent of total number of trees	Biomass t	Percent of total biomass
Bigtooth aspen	1	0.01	0.22	0.01
Bitternut hick.	6	0.09	2.44	0.08
Shagbark hick.	22	0.32	14.58	0.51
American hornbeam	5	0.07	0.05	0.00
East. hophornbeam	46	0.67	1.48	0.05
Sweet birch	246	3.59	72.56	2.50
American beech	662	9.67	135.73	4.67
American chestnut	11	0.16	0.76	0.03
White oak	70	1.02	131.96	4.54
Chestnut oak	165	2.41	200.40	6.90
North. red oak	518	7.56	876.92	30.19
Scarlet oak	10	0.15	13.18	0.45
Black oak	10	0.15	10.51	0.36
Cucumbertree	90	1.31	17.62	0.61
Fraser magolia	51	0.74	2.45	0.08
Yellow poplar	122	1.78	165.13	5.68
Sassafras	22	0.32	3.45	0.12
Downy serviceberry	164	2.39	4.22	0.14
Fire of pin cherry	2	0.03	0.58	0.02
Black cherry	283	4.13	376.77	12.97
Black locust	46	0.67	28.66	0.99
Ailanthus	1	0.01	0.03	0.00
Sugar maple	2200	32.13	399.62	13.76
Striped maple	231	3.37	1.39	0.05
Red maple	1319	19.26	287.31	9.89
Mountain maple	87	1.27	0.76	0.03
American basswood	79	1.15	18.85	0.65
Blackgum	98	1.43	33.41	1.15
Flowering dogwood	16	0.23	0.13	0.01
Sourwood	53	0.77	8.79	0.30
White ash	108	1.58	94.10	3.24
Grapevine	26	0.38	Not counted	
Hawthorn	1	0.01	0.01	0.00
Deciduous holly	8	0.12	0.01	0.00
Witch hazel	68	0.99	0.32	0.01
Total (All plots)	6848	100	2904	100

Results of this study suggest that at least 20 800 m^2 sampling plots are needed to obtain the average above ground biomass, representative for a large partial area of the watershed. The correlation between the above ground biomass, and Rn and Et is poor ($r \approx 0.2$) for single plots, increasing with the number of plots averaged, and for 18 plots it amounts to 0.89 for Rn and 0.82 for Et. Better correlations with the averaged plots area may be related to plot to plot variation in soil parameters (stoniness, thickness of A- and B-horizon) and in stand density (leaf area index).

The list of species identified in watershed 4 is similar to that for the Fernow Experimental Forest reported by Trimble (1973). Topography related variations of species frequency and above ground biomass in watershed 4 correspond more or less to those found in the Little Laurel Run catchment, West Virginia University Forest, near Coopers Rock, and elsewhere (Knight, 1980; Tajchman and Wiant, 1983; unpublished data). Because of the differences in sampling and reporting procedures and lack of information on stand development, a quantitative comparison of results from different reports would be difficult and would exceed the objectives of our study.

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DROUGHT TOLERANCE OF SUGAR MAPLE ECOTYPES

Richard J. Hauer and Jeffrey O. Dawson[1]

Abstract: Sugar maple declines periodically occur in rural and urban areas. These declines usually follow periods of below-average precipitation leading to the speculation that moisture deficiency is a primary cause of the decline. Sugar maple ecotypes with greater tolerance to drought should have greater longevity and vitality as a result of this tolerance. Sugar maple and black maple trees from the western and southern range of these species purportedly have greater drought tolerance. However, the quantification of physiological, morphological, and anatomical characteristics to explain and substantiate this purported tolerance is lacking. We present results on the response of greenhouse grown seedlings of five ecotypes from Oklahoma, Missouri, Iowa, Tennessee, and Ontario to experimental drought. Plant water relations (net photosynthesis, stomatal conductance, and water use efficiency) and tissue water relations (modulus of elasticity, osmotic potential at 0 and 100 percent turgor, and relative water content at 0 turgor) were used to test for variation in drought tolerance. Xylem anatomy, leaf structure and biomass partitioning further differentiated ecotypic variation.

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TREE REGENERATION FOLLOWING GROUP SELECTION HARVESTING IN SOUTHERN INDIANA

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Abstract: An increased interest in the use of group selection harvesting in the Central Hardwood forests has emphasized the lack of scientific information about species response under this uneven-aged management system. Tree regeneration response following group selection harvesting was studied on thirty-six group selection openings on the Naval Surface Warfare Center, Crane Division in southern Indiana. The objectives were to determine whether aspect, age, species group, size of the opening, or their interactions had any influence on the number of stems or average height of the tree regeneration. Two different aspects, three postharvest age classes, and three opening sizes were examined. Yellow-poplar and dogwood were the most abundant stems ≥ 2.5 cm dbh, and dogwood and cherry-ash-walnut were the most abundant stems ⟨ 2.5 cm dbh. Stem density (⟨ 2.5 cm dbh) was influenced by age, aspect, and the age/species group and aspect/species group interactions. The north aspect and the oldest openings contained the highest number of stems per hectare for stems ⟨ 2.5 cm dbh. Age, species group, and their interaction influenced average height for stems ≥ 2.5 cm dbh.

INTRODUCTION

Recently there has been an increased interest in the use of group selection for the regeneration of Central Hardwood forests. Since the mid 1960s clearcutting was one of the most commonly used methods to regenerate Central Hardwood forests on national forest lands. Its use was encouraged by research completed by Roach and others (1968), and by Sander and Clark (1971). Therefore, research conducted by the USDA Forest Service and other researchers has centered around the use of clearcutting with limited research effort given to uneven-aged management.

The renewed emphasis in uneven-aged management has underscored the need for information on uneven-aged silvicultural systems. There is little scientific information about Central Hardwood regeneration under uneven-aged management which addresses the effect of opening size, parent stand composition, variation within an opening, temporal change, or aspect on regeneration.

The group selection method was developed to combine the benefits of single-tree selection and even-aged management (Marquis 1978). A benefit of single-tree management is the retention of a fairly continuous canopy providing a more natural appearing forest stand. Even-aged management provides for the regeneration of shade intolerant tree species. Thus, group selection should provide for the regeneration of intolerant and intermediate trees while retaining a mostly continuous forest canopy.

The suggested size of the group opening varies from the removal of a few trees to many trees. There has been much discussion on the maximum and minimum size. Marquis (1989) suggests an opening of a tenth to a quarter acre in size with a maximum of a half acre. An opening from a half acre to one acre in size is recommended to regenerate intolerant species (Law and Lorimer 1989, Sander and Clark 1971). Openings of up to two acres have been mentioned as group selection (Mills and others 1987) but if the openings are larger than an acre Marquis (1989) and Marquis and Johnson (1989) suggest they be termed patch clearcuts.

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Light reaching the group selection openings can be managed (Fischer 1981) by controlling the shape and placement of the opening on the landscape. Thus, the regeneration of intolerant species can be encouraged by the proper shape and location of the group selection openings.

This paper will report how the quantity and height of the four most prevalent species groups; oak-hickory, yellow-poplar, dogwood, and beech-sugar maple, are influenced by the age, size, and aspect of the group selection openings.

METHODS

This research was conducted on the Naval Surface Warfare Center-Crane Division, (NSWC), located in south central Indiana. NSWC contains over 62,000 acres of which more than 48,000 are forested. Uneven-aged management has been the regeneration system used at NSWC since timber management began there in the early 1960s.

Nearly the entire NSWC is unglaciated and lies within the Crawford Upland Section of the Shawnee Hills Natural Region as described by Homoya and others (1985). It is characterized by rugged hills with sandstone cliffs and rockhouses. The forest vegetation consists of oak-hickory on the upper slopes with a mesic component in the coves and bottoms. Soils in the study area were of the Wellston association characterized as deep and moderately deep, gently sloping to very steep, well drained soils formed in loess and material weathered from sandstone, siltstone, and shale on uplands (USDA Soil Conservation Service 1988). The group selection openings ranged in elevation from 170 m to 260 m.

The overstory composition surrounding nineteen of the group selection openings was classified as oak-hickory by the NSWC stand inventories. The oak-hickory stands averaged 75% oak-hickory while no other species group contributed over 10%. Seventeen of the group selection openings were surrounded by stands of mixed hardwood. The mixed hardwood stands contained 44% oak-hickory, 21% beech-sugar maple, 11% yellow-poplar, and no other species group contributing more than 10%.

All timber sale contracts from NSWC for the period 1971 to 1985 were examined for potential group selection sample sites. The group selection harvests were then divided into three age classes depending on the time of harvest: 6-10 years from harvest (1981-1985), 11-15 years from harvest (1976-1980), and 16-20 years from harvest (1971-1975). After year 20, timber stand improvement work was begun on these stands thus influencing the composition of the groups. Stands harvested less than 6 years ago were not examined because the group openings were still changing rapidly and species dominance had generally not become well established.

Each timber sale area was located on aerial photos and examined stereoscopically to locate as many group selection openings as possible. Group openings in bottomlands or on ridge tops were not sampled. The openings were divided into groups of north and south aspect, using the aerial photos and topographic maps. The divisions, north aspect 315°-135° and south aspect 136°-314°, were based on previous work in southern Indiana by Hannah (1968) and by Van Kley (1993). Once all timber sale contracts were examined, six openings from each of the three age classes for each of the two aspects were randomly selected from the pool of openings for sampling. Thus, there were 12 openings per age class or 18 openings per aspect for a total of 36 openings.

All selected openings were examined in the field to verify: (1) that the aspect was appropriately identified, (2) that the opening was fairly distinct and discernible, (3) that the surrounding stand was mature upland central hardwoods, and (4) that the opening was not in an area of reverting field. Each group opening failing to meet conditions (2) through (4) was dropped, and a new one was randomly selected.

Woody plant species were sampled on transects extending from the center to the edge of each opening. One-meter-wide transects were used for stems less than 2.5 cm diameter at breast height (dbh, 1.37 m above ground), and 2-meter-wide transects were used for all stems greater than or equal to 2.5 cm dbh. Transects extended upslope, downslope, and in both directions across the slope, beginning 1 meter from the center of the opening and ending at the

edge of the opening as defined by the trunks of the mature trees surrounding the opening. The ending of transects at the base of the trees bordering the opening follows Runkle's (1982) method of measuring extended canopy gaps. Each transect was subdivided into 1-meter lengths for sampling woody species by location along the transect. Field data collection was completed during the summer of 1991.

Height (in 25-cm categories) and distance from the plot center (in 1-m increments) were measured by species for each stem less than 2.5 cm dbh. Dbh (to the nearest 0.1 cm), height (nearest 1 m), crown class (in relationship to the regeneration), and distance from the plot center (1 m) were recorded by species for woody stems greater than or equal to 2.5 cm dbh.

Aspect was determined in the field, and area of each opening was calculated using the length of each transect to determine the area of an ellipse. The group openings ranged in size from .07 ha to .23 ha with an average size of .12 ha. For analysis the openings were divided into three size classes; less than 0.1 ha with 12 openings, 0.1 ha to 0.14 ha with 15 openings, and greater than 0.14 ha with 9 openings. Also for analysis, the species were put into the following eight groups: (1) oak-hickory, (2) yellow-poplar, (3) dogwood, (4) beech-sugar maple, (5) shrubs and small trees, (6) cherry-ash-walnut, (7) sassafras, and (8) miscellaneous species. Only the first four species groups are discussed in this paper.

Analysis of variance was used to determine whether the variables aspect, age, size of opening, species group, or their interaction had any effect on the average number of stems per hectare or average height. When an effect was found to be significant, the Student-Newman-Keuls multiple range test was used to determine differences between the means.

RESULTS

Number Of Trees

DBH $\geq$ 2.5 cm. Species group was the only variable which influenced the number of stems per hectare $\geq$ 2.5 cm dbh. Aspect, size of the opening, and age had no influence. Yellow-poplar was the most abundant species group, followed by dogwood, beech-maple, and finally oak-hickory. This ranking was consistent for all age classes, group opening size classes, and both aspects (Figure 1a-c). While this ranking was constant, it was seldom significant. Yellow-poplar was significantly more abundant than beech-maple and oak-hickory in the 11-15 year age class. Abundance was not significantly different among species groups for the youngest and oldest openings. Beech-maple was the only group to show a significant change in number of stems between age classes. The oldest age class contained significantly more stems than the two youngest age classes.

There was no significant difference between individual species groups among opening sizes or aspects. Yellow-poplar and dogwood were significantly more abundant than beech-maple and oak-hickory in the smallest openings and yellow-poplar was significantly more abundant than beech-maple and oak-hickory in the middle size openings (Figure 1b). In the largest openings there was no significant difference between species groups.

Yellow-poplar was significantly more abundant than oak-hickory on the north aspect (Figure 1c). There were significantly more yellow-poplar and dogwood than oak-hickory, and significantly more yellow-poplar than beech-maple in openings with a southern aspect.

DBH $<$ 2.5 cm. The number of smaller stems, $<$ 2.5 cm dbh, was influenced by the age, aspect of the openings, and by the species groups. The size of the openings did not influence the number of stems. The two oldest age openings had more stems than the youngest. In the youngest openings, there were significantly more dogwood stems than the other three species groups (Figure 2). In the middle age openings, there were significantly more dogwood stems than yellow-poplar. In the oldest openings sampled, there were significantly more dogwood and beech-maple than oak-hickory and yellow-poplar.

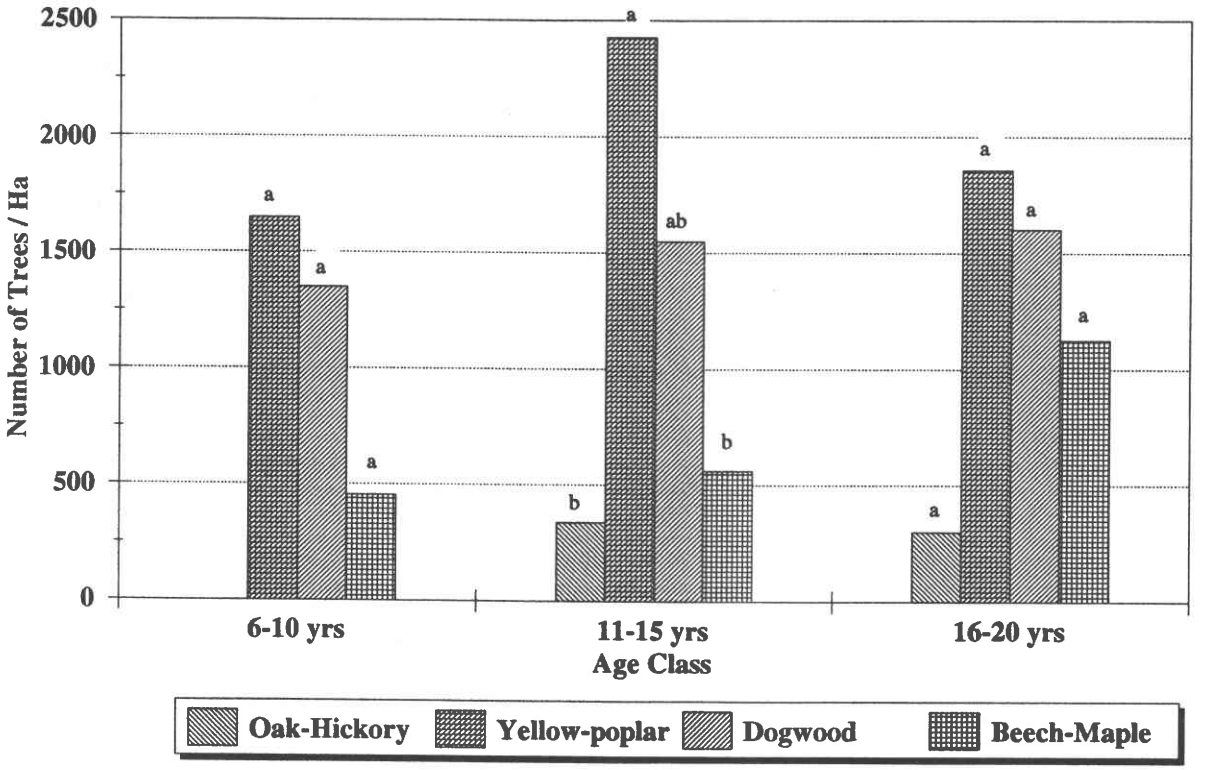

Figure 1a. Number of trees greater than or equal to 2.5 cm dbh by species group and age class. Species groups within an age class with the same letter are not significantly different at the 0.05 level.

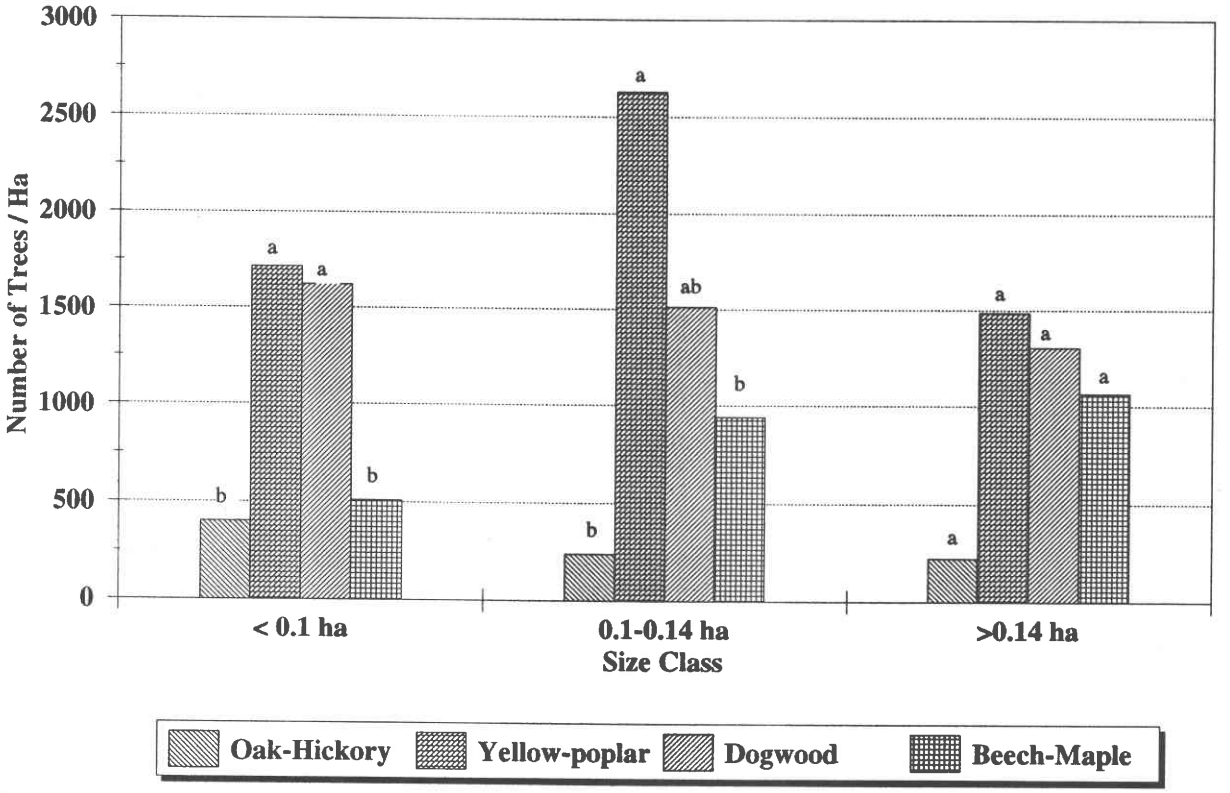

Figure 1b. Number of trees greater than or equal to 2.5 cm dbh by species group and size class. Species groups within a size class with the same letter are not significantly different at the 0.05 level.

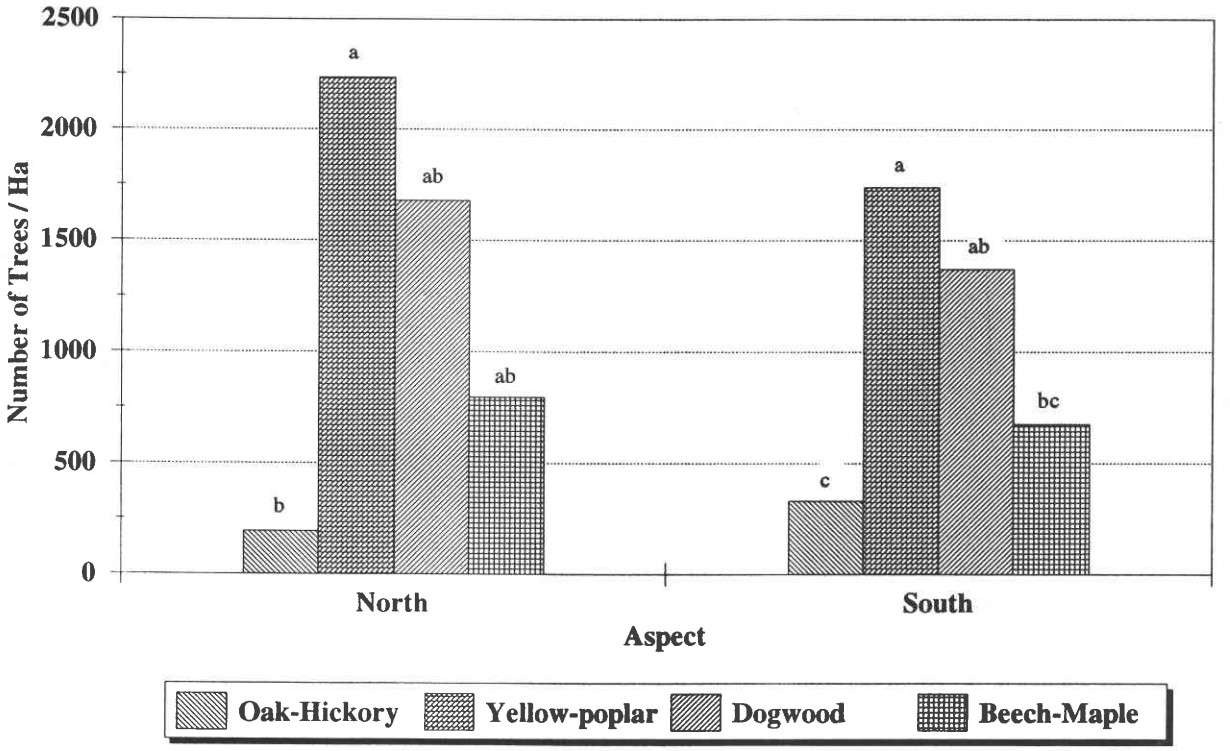

Figure 1c. Number of stems greater than or equal to 2.5 cm dbh by species group and aspect. Species groups within each aspect with the same letter are not significantly different at the 0.05 level.

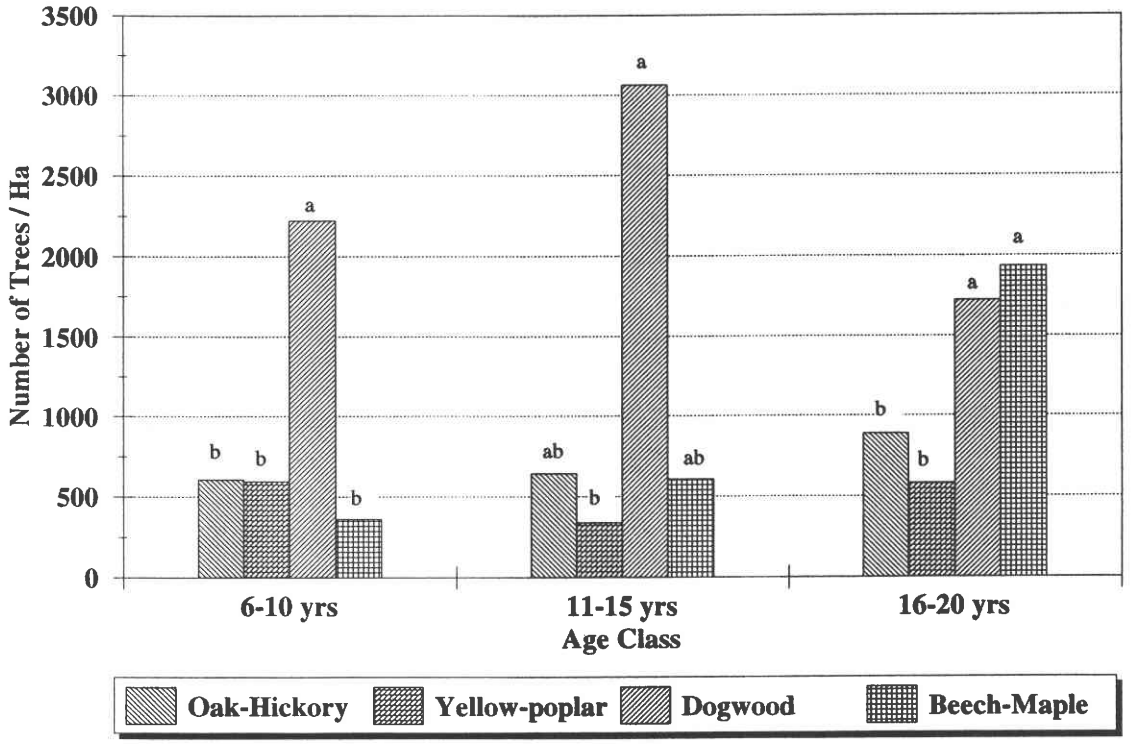

Figure 2. Number of trees less than 2.5 cm dbh by species group and age class. Species groups within an age class with the same letter are not significantly different at the 0.05 level.

Average Height

DBH ≥ 2.5 cm. Height of stems ≥ 2.5 cm dbh was significantly influenced by age of the opening and species groups, but not by size or aspect of the opening. Yellow-poplar was the only species group to show a significant increase in height between age classes. Beech-maple and dogwood showed a slight decrease in height from the middle to the oldest age class. Oak-hickory showed a nonsignificant increase in height from the youngest to the oldest openings.

In the youngest age class, beech-maple, dogwood, and yellow-poplar were significantly taller than oak-hickory; also, beech-maple was significantly taller than dogwood (Figure 3a). Beech-maple and yellow-poplar were significantly taller than the other two species groups in the middle age class. By the oldest age class, yellow-poplar was significantly taller than the other groups with beech-maple and oak-hickory significantly taller than dogwood.

Yellow-poplar and beech-maple showed a positive height increase with increasing opening size (Figure 3b). However, only the increase from the middle size openings to the largest openings was significant for yellow-poplar. Oak-hickory and dogwood showed a negative response to increased opening size. Yellow-poplar and beech-maple were significantly taller than oak-hickory and dogwood only in the largest openings.

DBH < 2.5 cm. Stems < 2.5 cm dbh decreased in height from youngest to oldest stands. This trend was significant. The other significant variable was species group. Neither aspect nor size of opening had a significant affect on the average height of the regeneration. This decrease in height was most likely due to the taller stems growing into the larger size class and to mortality.

Yellow-poplar was significantly taller than the other species groups for the two youngest age classes but by the oldest age class it was the shortest (Figure 4). Dogwood remained the only group taller than 1 meter, by the oldest age class.

DISCUSSION

The oak-hickory type is the most widespread and currently covers the greatest area in the Central Hardwood forests (Sander and Fischer 1989). The perpetuation of this type is of great concern to many people. In this study area the parent stands were dominated by the oak-hickory group. As indicated by trees ≥ 2.5 cm dbh, the future stands will not be dominated by the oak-hickory group which contained the fewest number of stems. It would appear that the future stand will be comprised of a diversity of species with yellow-poplar being the most dominant. The stems < 2.5 cm dbh confirm this trend with the oak-hickory group not among the most common.

These results differed from those reported by Runkle (1981) and Barden (1981), who reported that species replaced themselves in larger canopy gaps. However, their research was conducted in forest stands where tolerant species dominated the forest canopy and tolerant species replaced tolerant species. In stands with less tolerant oak-hickory overstories, Standiford and Fischer (1980) reported a decrease from 80% oak before harvest to 7% oak 12 to 15 years after harvest. Fischer (1987) and George and Fischer (1991) showed a similar trend for clearcuts on the Hoosier National Forest. They found that the oak-hickory group declined in numbers from the original parent stand.

One unknown in the study was the quantity and type of advanced regeneration present before harvest. Many of the group selection openings studied might have been located in areas that contained little advance regeneration, accounting for the low percentage of oak-hickory. Sander (1978) and Sander and others (1983) suggested that successful oak regeneration following harvest depends primarily on the amount and size of oak present as advance regeneration prior to harvest.

If oak advance regeneration is not present in sufficient quantities, steps must be taken to increase the quantity. Loftis (1985, 1988) recommended the use of herbicide to remove undesirable stems, allowing for the increase in size and number of desirable stems. Johnson and others (1986) suggested, in conjunction with a shelterwood silvicultural system, the planting of oak seedlings 3 years before harvest to increase the number of oaks.

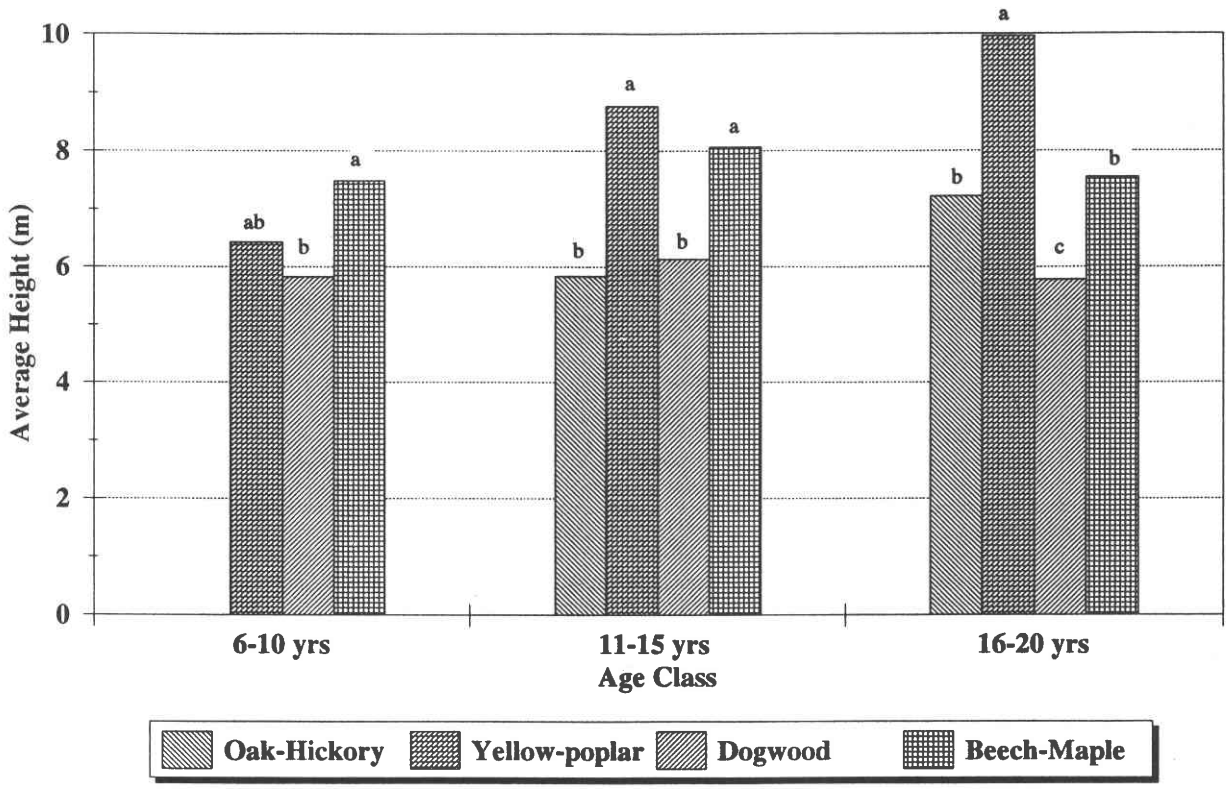

Figure 3a. Average height of trees greater than or equal to 2.5 cm dbh by species group and age class. Species groups within an age class with the same letter are not significantly different at the 0.05 level.

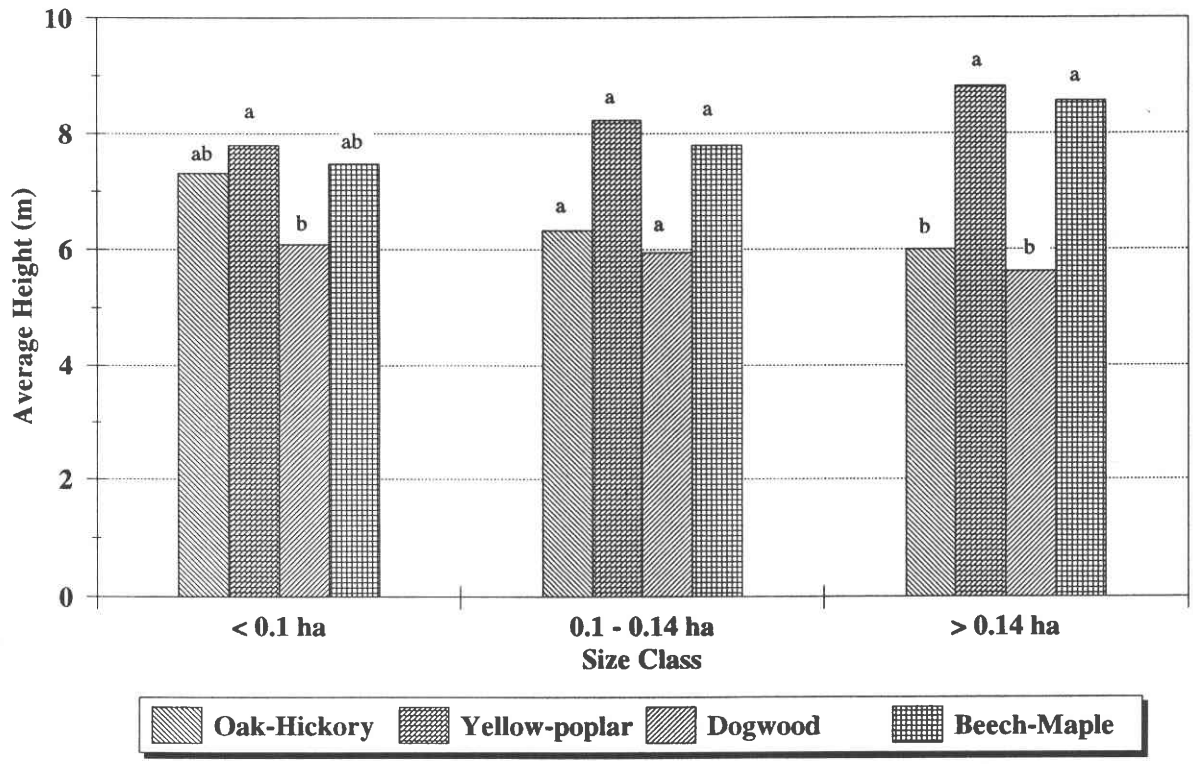

Figure 3b. Average height of trees greater than or equal to 2.5 cm dbh by species group and size class. Species groups within a size class with the same letter are not significantly different at the 0.05 level.

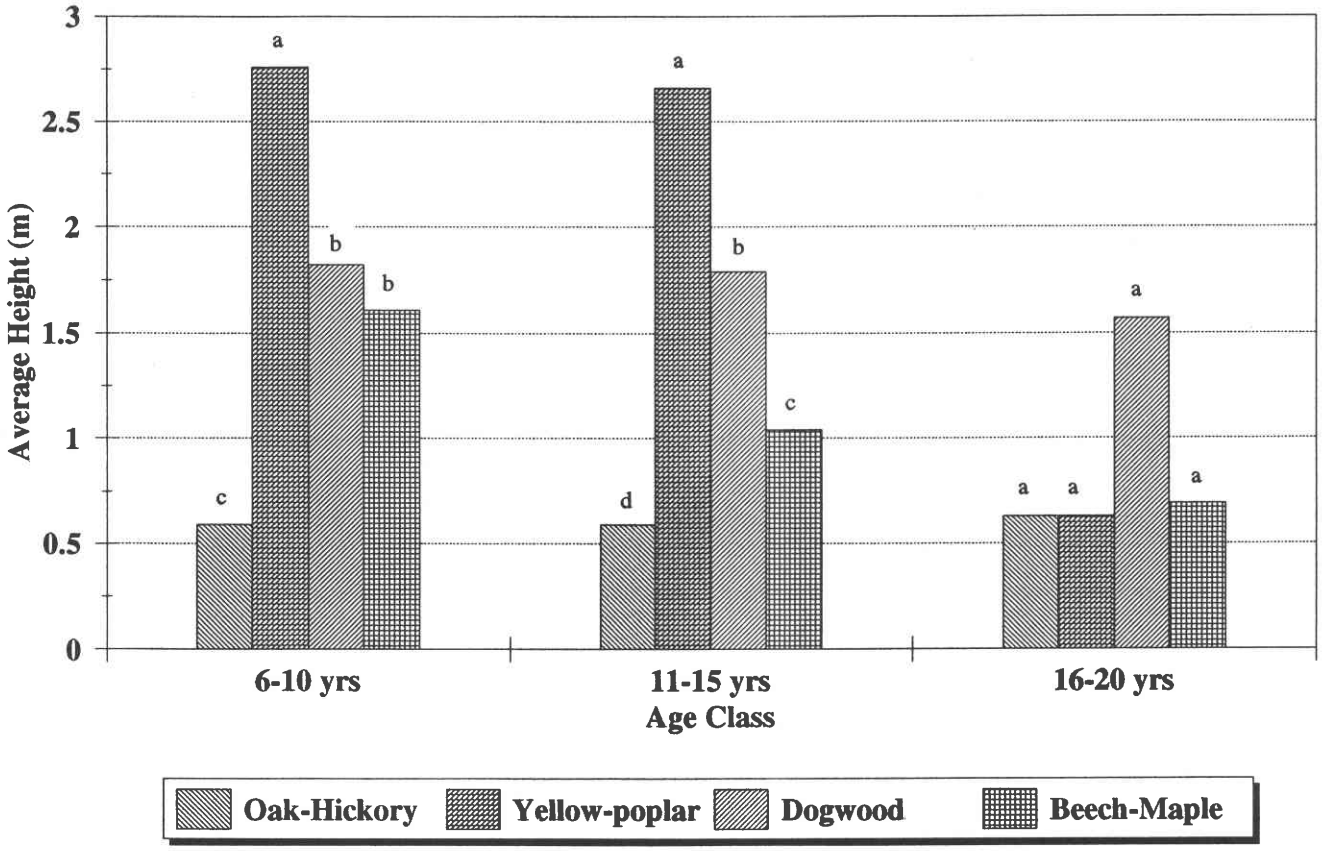

Figure 4. Average height of trees less than 2.5 cm dbh by species group and age class. Species groups within an age class with the same letter are not significantly different at the 0.05 level.

As applied, the group selection openings gave no evidence of promoting oak regeneration success. Additional investigation to monitor group selection openings with known levels of advance oak reproduction would be beneficial.

CONCLUSION

This study was an initial examination of tree regeneration following group selection harvesting in openings of three age classes. The progression of tree regeneration through time was not directly examined in this study; it was inferred from an examination of thirty-six group selection openings observed at different times since harvest. Whether or not the regeneration in the youngest group selection openings will contain the same mix of tree regeneration as the oldest group selection openings in another decade cannot be known with certainty, but it seems a likely course of development.

A better study could be conducted if it were established prior to harvesting. This would allow for the measurement of the advanced regeneration and the overstory composition. The regeneration within the openings could then be followed over time to obtain a more accurate picture of the progression of tree regeneration. Establishing the study prior to harvest would also permit the control of opening size, shape, and placement. Unfortunately this was not possible for this study.

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REGENERATION IN DEFOLIATED AND THINNED HARDWOOD STANDS OF NORTH-CENTRAL WEST VIRGINIA

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Abstract: Overstory species regeneration was examined in 1989, prior to gypsy moth defoliation and thinnings, on 16 stands in the West Virginia University Forest. Three stands were thinned and defoliated while five were thinned only and three were defoliated only. Five stands were neither thinned nor defoliated. Data were collected from these stands for three years subsequent to the disturbances. The general response to defoliation was similar in number of seedlings irrespective of thinning, i.e., there was an initial depression of regeneration, although it was more noticeable in stands that were not thinned. Total seedling production increased significantly on thinned stands two years following thinning, but differences disappeared by third year after treatment. For both defoliated and undefoliated stands, any effect of thinning on total seedling abundance was lost by 1992. Oak regeneration was greatest on control stands and stands that were thinned but not defoliated. Defoliation alone was detrimental to oak regeneration, while thinning moderated the defoliation effect. Red maple seedlings dominated numerically, but black cherry was the most abundant species in the larger size classes in all stands irrespective of treatment, and is likely to dominate the next generation of overstory trees in this area.

INTRODUCTION

A relatively thorough literature exists explaining the effects of gypsy moth defoliation on overstory species composition (Campbell and Sloan 1977, Feicht and others 1993, Herrick and Gansner 1988, Stephens and Hill 1971, Twery 1990, among others). Less is known, however, about understory vegetation and woody species regeneration responses to gypsy moth defoliation. Such knowledge is critical in predicting species replacement and determining appropriate management techniques for post defoliated stands as well as in advance of defoliation.

As silvicultural methods are employed with greater frequency in attempts to reduce susceptibility and vulnerability to gypsy moth (Gottschalk 1993), the understory responses to such treatments should be an important consideration. In many ways, defoliation resembles thinning and as such is likely to produce comparable effects, particularly in creating canopy openings, and increasing light penetration. In areas that have been thinned and defoliated, the potential for dramatic changes in species composition is great. With this study, we examine the effects of both defoliation and thinning on woody species regeneration.

Oaks (*Quercus* spp.) represent the most favored host of gypsy moth. Declining oak dominance from the potential of extreme oak mortality is exacerbated by oak regeneration problems which appear to be common throughout Eastern forests (Loftis and McGee 1993). Competition from intolerant and intermediate species contribute to regeneration problems. For example, red maple (*Acer rubrum* L.) is one of the more abundant regenerating species in many oak-dominated forests in the Midwest (Host and others 1987, Johnson 1992) and the Northeast (Lorimer 1984). In the Ridge and Valley Province in Pennsylvania, Nowacki and Abrams (1992) found that maple, cherry (*Prunus)*, and/or birch (*Betula*) are slowly replacing oak and that maple and beech (*Fagus grandifolia* Ehrh.*)* are dominating historically white oak (*Quercus alba* L.) dominated areas. Studies by Allen and Bowersox (1989), Ehrenfeld (1980), Fosbroke and others (1991), and Hix and others (1991) have examined regeneration following gypsy moth

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defoliation. Results of the previous research consistently indicate a reduced regeneration of previously dominant overstory species.

This study examines woody species regeneration, particularly focusing on oak, in mixed hardwood stands of the central Appalachians. Specifically, the study assesses the impact of silvicultural techniques used to minimize gypsy moth defoliation damage in addition to the effect of gypsy moth defoliation and mortality on woody regeneration.

METHODS

The study area is the West Virginia University Forest in north central West Virginia. The forest is located in Preston and Monongalia Counties, within the general boundaries of the mixed mesophytic forest region (*sensu* Braun 1950) in the slightly folded region of the Appalachian Plateau. The forest is generally considered a mixed hardwood forest, although is purported to have been heavily dominated by white oak at one time (Carvell and others 1978). Elevation of the study area ranges from 1800 to approximately 2300 feet, and Dekalb series represent the most abundant soil type.

Since the late 18th century the area has been heavily disturbed. Charcoal manufacture for the iron industry accounted for early use of the timber resource, followed by specialty wood products such as railroad ties and whiskey barrels. Nearly all of the Forest was eventually clearcut and much of it burned by the early 1930's (Carvell and others 1978).

For the current study, sixteen stands (eight pairs), ranging from 19.3 to 31.2 acres and accounting for a total of 408 acres were selected in 1989. A pair represents two adjacent stands of similar species composition and size structure, and reflects comparable suitability as host for the gypsy moth. One of each pair (hereafter referred to as "stands"), was thinned in Fall, 1989 or early Spring 1990. The objective of thinning was to reduce stand susceptibility or vulnerability (Gottschalk 1993) prior to potential gypsy moth defoliation. Defoliation followed in six stands in 1990 and in 1991. Each of the six stands incurred over 50% defoliation of preferred (Twery 1990) species and over 40% defoliation of all species for two years. Defoliation on the other stands, i.e. background defoliation level, was less than 15% of all species, including preferred. Using the random (defoliation) and fixed (thinning) treatments, the stands were classified as **C** (control), unthinned and undefoliated, **T** - thinned and undefoliated, **D** - defoliated but unthinned, and **DT** - defoliated and thinned. The **C** and **T** groupings are paired, and the **D** and **DT** groups are paired, e.g. **C1** and **T1** represent one pair of stands, as do **D1** and **DT1.**

Species composition in all structural layers was determined in 1989 prior to any treatment, and was remeasured in 1990, 1991, and 1992. Nomenclature follows Little (1979). Twenty 0.1 acre (0.04 ha) plots were established for overstory sampling and within each of these plots three, systematically located, 113 ft^2 (10.5 m) circular plots were sampled for woody seedling regeneration. Regeneration in the control stands (**C**), however, was not measured in 1990. Species were identified and categorized by the following size classification:

Size class 1:	all woody plants to 1 ft tall
Size class 2	1 to 5 ft tall
Size class 3	Greater than 5 ft tall, but less than 2.49 inches dbh
Overstory	Greater than 2.49 inches dbh

We used repeated measures analysis of variance to determine temporal trends in regeneration and to examine treatment effects. The data from 1989 were used as contrasts to determine significance over time. Analysis of covariance, using the pre-treatment data as covariates, was used to examine individual treatment effects.

RESULTS

Overstory Species Composition

Generally a mixed hardwood forest with a substantial component of oak, the basal area of much of the West Virginia University Forest was dominated by yellow-poplar (*Liriodendron tulipifera* L.) in 1989 (Table 1). In stands dominated by oak there were fewer overstory species overall compared to stands dominated by yellow- poplar. Although considered the dominant species in the West Virginia University Forest at one time, white oak accounted for only a small proportion of total basal area on all stands. The red maple component ranged from 4 to 17% of total stand basal area initially.

Significant decreases in overstory basal area occurred in all but the control stands (Table 1). Although stand basal areas were reduced substantially in the thinned stands, the relative contribution of the important overstory species remained somewhat constant. One of the objectives of the thinning was to reduce the overstory oak component, but this proved difficult in practice because of the actual distribution of trees. In **D** and **DT** stands, red maple proportion in the overstory increased to as much as 47% in some stands. Black cherry (*Prunus serotina*) increased substantially on one defoliated stand (**D2**).

Initial Regeneration

Table 2 includes only those species likely to attain overstory size in the 16 stands. Some non-overstory woody species were also plentiful. For example, blueberry (*Vaccinium* spp.) was the third most abundant woody species across all sixteen stands. Maple-leaf viburnum (*Viburnum acerifolium* L.), spicebush (*Lindera benzoin* (L) Blume.), and blueberry accounted for a total of 12% of all individuals sampled. By contrast, red oak (*Quercus rubra* L.), the most abundant oak species, accounted for only 4% of all individuals sampled in 1989. Oak regeneration in general was minimal and was most apparent on those stands with a strong oak component, **D1**, **D2** and **T5**. Red oak regeneration occurs most abundantly in stands with a considerable red oak overstory, but in stands with the greatest oak component in the overstory, the understory was dominated by black cherry (**D2** and **DT2**) and red maple (**D3** and **DT3**). *Vaccinium* also was a major component in the understory in those four stands. Numerically, black cherry and red maple dominated the regeneration overall. Overstory species, present but excluded from the table, due to small numbers include elm (*Ulmus rubra*), beech, black locust (*Robinia pseudoacacia*), basswood (*Tilia americana*), sugar maple (*Acer saccharum*), and American chestnut (*Castanea dentata*).

Effect of Treatments on Total Number of Seedlings

Throughout the four year period, there were slightly more seedlings on the undefoliated than on defoliated stands. Over the four years, number of seedlings on thinned, defoliated stands did not change, but thinning contributed to a significant increase in the total number of seedlings on defoliated stands in 1990 and 1991 when compared with unthinned stands. By 1992, there was no significant difference due to thinning, and seedling abundances for that year were nearly identical for all stand types. The 1992 values were similar to the 1989 values (Fig. 1). On stands that were undefoliated, thinning significantly increased seedlings in 1991, but there was no effect by 1992 (Fig. 2). Thinning did not result in an increase in total number of seedlings; rather, differences arose primarily because of a decrease in seedling numbers in unthinned stands.

Repeated measures analysis of variance indicates no general change in total number of seedlings (all species) from 1989 to 1992; however, there was a significant treatment effect in 1990 ($P<0.001$), and 1991 ($P<0.001$). The treatment effect was lost by 1992. There is a significant treatment by time interaction, suggesting that the pattern is not the same for all treatments.

Table 1. Relative overstory composition of study stands based on stems ≥2.5 inches dbh. Basal area represents the total basal area of each stand and the percent each of the major species contributes to total for 1989 (pre-treatment), and for 1992.

Stand	Basal Area (ft^2/ac) year - total	Red oak %	White oak %	Chest-nut oak %	Other oaks* %	Red maple %	Yellow-poplar %	Black cherry %
C1	1989 - 133.4	10.4	1.8	14.5	5.1	8.7	40.5	0.5
	1992 - 137.6	10.9	1.9	13.5	3.1	9.4	39.2	0.6
C2	1989 - 134.3	25.5	3.5	9.3	4.6	16.3	28.1	2.3
	1992 - 140.6	24.0	3.4	9.4	4.7	16.5	28.5	2.3
C3	1989 - 136.0	12.1	5.1	9.5	8.3	8.7	40.6	0.6
	1992 - 140.3	12.4	5.0	9.7	8.4	9.3	41.2	0.4
C4	1989 - 133.5	31.5	3.0	13.3	0.9	16.4	8.0	14.7
	1992 - 142.3	31.3	3.0	13.4	0.8	16.6	8.2	14.8
C5	1989 - 122.6	35.6	1.7	9.1	2.0	16.4	21.7	4.7
	1992 - 127.8	37.5	1.5	9.2	2.0	16.8	22.1	4.4
T1	1989 - 147.5	20.8	2.3	13.3	4.3	12.2	30.8	1.9
	1992 - 107.4	21.2	1.4	7.1	4.9	14.7	36.3	0.6
T2	1989 - 149.0	18.3	1.6	15.6	3.7	13.8	37.2	3.0
	1992 - 114.4	18.3	2.0	10.9	1.6	18.2	41.0	3.9
T3	1989 - 144.1	10.4	0	0.3	1.3	3.9	64.8	2.7
	1992 - 103.0	10.8	0	0.1	1.9	5.5	68.0	1.4
T4	1989 - 132.7	42.3	3.5	5.1	1.4	14.0	7.6	14.4
	1992 - 88.4	47.1	4.0	1.5	0.5	16.9	6.0	11.7
T5	1989 - 125.9	28.8	6.7	13.2	5.9	14.6	22.1	3.2
	1992 - 86.6	32.4	5.1	12.0	3.8	17.5	20.0	3.9
D1	1989 - 110.7	43.8	4.0	5.1	0.1	17.3	8.6	1.5
	1992 - 75.0	22.6	1.5	1.5	0	29.3	14.2	2.8
D2	1989 - 120.5	53.4	5.2	17.0	4.2	6.5	0.6	12.3
	1992 - 49.8	30.5	1.2	15.2	0.7	18.1	0	30.6
D3	1989 - 129.5	51.3	6.4	12.4	5.7	15.7	0.7	1.7
	1992 - 52.6	38.2	0.1	0.5	0	46.1	1.8	4.4
DT1	1989 - 130.8	34.2	1.9	4.0	2.0	12.5	27.0	0.9
	1992 - 84.4	11.4	0	0.3	0	22.0	44.9	1.9
DT2	1989 - 125.7	62.3	3.6	6.6	8.6	7.4	0	9.6
	1992 - 69.1	63.0	0.8	3.1	7.8	14.0	0	9.7
DT3	1989 - 127.8	40.6	11.6	7.9	6.2	16.8	0	0.4
	1992 - 38.0	18.3	0	3.6	0.6	47.4	13.4	1.9

Table 2. Overstory species regeneration prior to treatments. Values represent number of seedlings per acre of each species. All size classes are combined.

Stand	red maple	black cherry	sassa-fras	yellow poplar	red oak	black gum	chest-nut oak	white oak	sweet birch	hick-ory	sour-wood	mag-nolia	ash	black oak	scarlet oak
C1	25167	957	3000	2544	334	546	373	109	129	103	193	39	77	6	6
C2	8982	8725	1818	1298	495	777	135	77	251	19	0	45	13	0	0
C3	12734	6001	8397	1022	771	244	752	964	0	71	283	51	39	39	13
C4	3155	15330	835	874	430	6	84	6	315	13	0	0	6	0	0
C5	6624	13486	1047	3347	1092	141	180	141	77	32	0	0	308	0	0
T1	13730	1561	2339	777	655	996	668	173	180	244	148	26	6	6	0
T2	8886	8956	1144	2634	482	315	238	77	45	0	0	6	6	6	0
T3	1606	3225	5872	3540	437	6	13	13	45	64	0	6	0	39	0
T4	3553	12092	398	970	578	0	64	186	302	13	0	0	0	13	0
T5	7408	9278	1471	4446	1632	77	116	116	0	77	0	411	0	13	0
D1	13171	411	341	732	1574	167	64	90	39	13	0	0	0	6	0
D2	4106	14116	463	26	1375	6	353	58	51	0	0	0	0	32	0
D3	6772	617	141	6	1092	720	334	109	199	0	26	0	0	13	6
DT1	9638	263	1092	2872	1812	443	71	0	418	13	0	0	0	0	0
DT2	3386	11944	328	0	2050	13	135	334	32	45	0	0	0	19	0
DT3	7993	649	1439	0	572	58	32	206	334	0	0	0	0	0	13

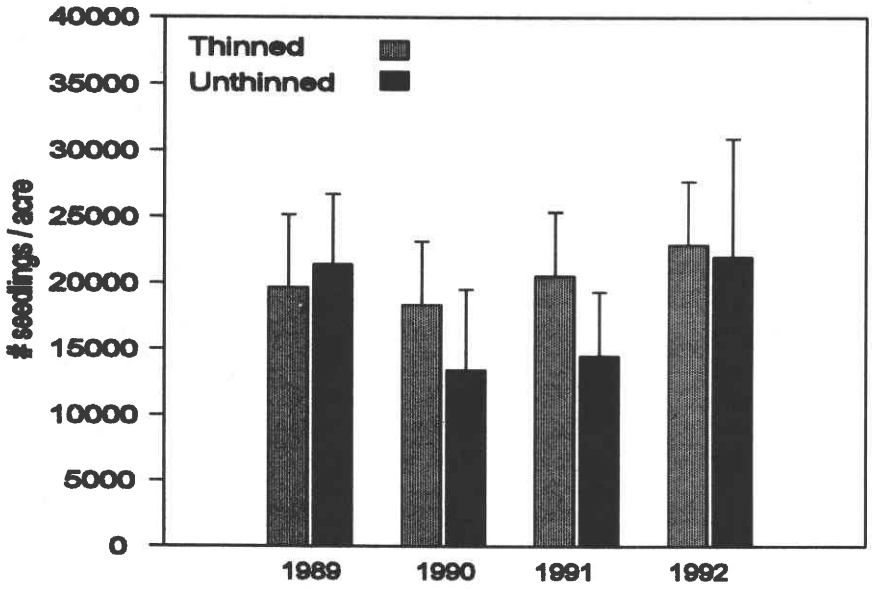

Figure 1. Average seedling abundance (all species) in defoliated stands.

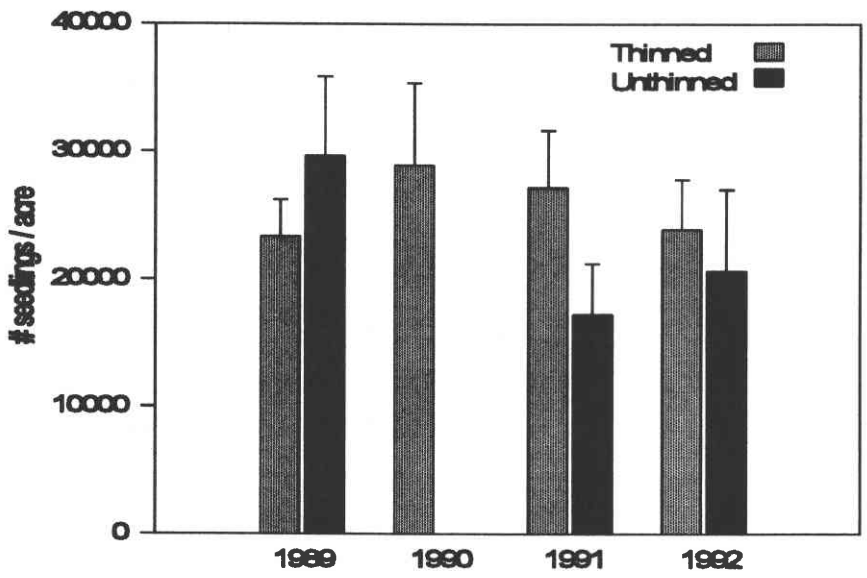

Figure 2. Average seedling abundance (all species) in undefoliated stands. Data were not collected for unthinned stands in 1990.

Oak Regeneration

The number of oak seedlings on defoliated stands (both **D** and **DT**) diminished over time (Figure 3). The decrease was greatest between 1989 and 1990 on **DT** stands; **D** stands dropped further, but after another year. Oak seedling abundance increased in both **T** and **C** stands, although, thinning increased production of oak seedling to a slightly greater extent. With time, in defoliated stands, irrespective of thinning, oak contributed less in successive years, as a consequence of both fewer oak seedlings and an increase in non-oak reproduction. This was confirmed through a significant time effect ($P<0.0001$), and a significant time by treatment effect ($P<0.001$) , but no significant treatment effect ($P=0.21$).

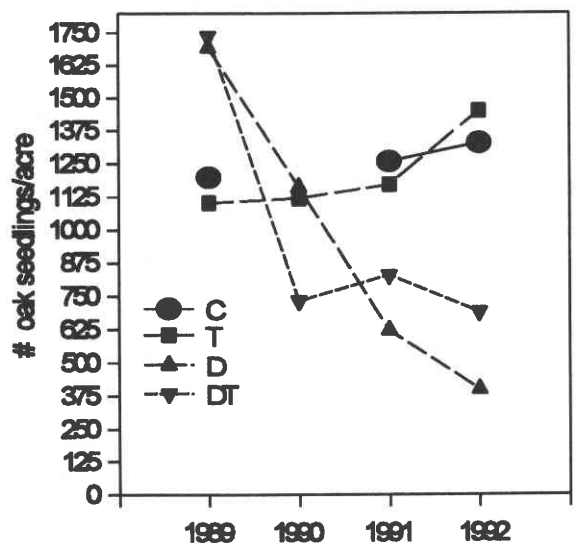

Figure 3. Oak regeneration from 1989 to 1992 for each treatment type. Regeneration was not sampled for control stands in 1990.

Treatment effects are not always consistent when individual stands are examined, and even control stands exhibit wide variation in production of oak seedlings. Figure 4 illustrates the change in oak seedlings as a percentage of original number (1989) of oak seedlings in each stand. With the exception of one stand (**T5**), thinning alone enhanced oak seedling production. Generally, thinning increased oak seedling production relative to no treatment in undefoliated stands, but one pair (**C1, T1**) did not follow the pattern of the other undefoliated stands. Defoliation, and thinning in conjunction with defoliation clearly reduced the oak component in the understory. Unthinned defoliated stands lost slightly more oak seedlings than **DT** stands in terms of absolute numbers and relative percentage of 1989 seedling numbers.

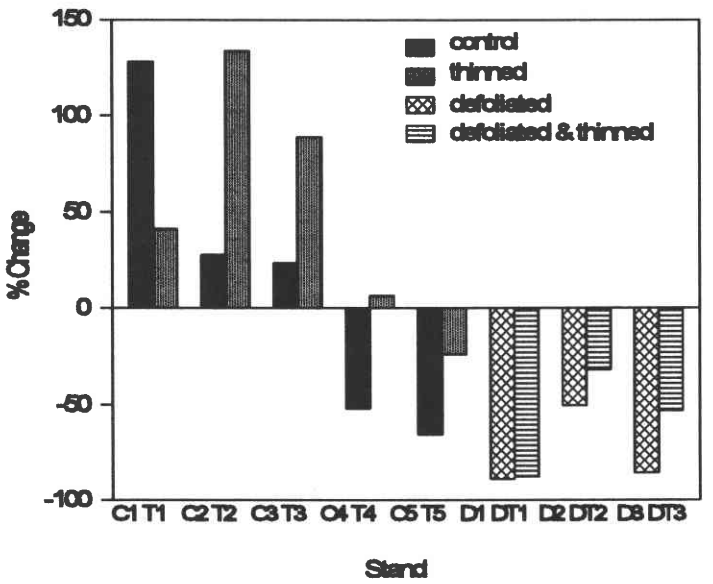

Figure 4. Percent change in oak seedlings from 1989 to 1992 for each stand. Values represent the number of oak seedlings in 1992 as a percentage of 1989 oak seedlings.

In all but one of the defoliated stands the amount that oak contributed to the regeneration decreased each year (Table 3). Pre-treatment (1989) regeneration values were not always comparable in the paired stands, and in no case was oak regeneration a very dominant group among seedlings despite the fact that some of these stands had over 70% basal area in oak originally. Oak contribution to regeneration was greatest prior to disturbance and immediately following initial disturbance (Table 3).

Regeneration Species Composition

Individual species responded differentially to the treatments (Table 4.) Over the four year period, red maple's contribution to total seedlings decreased under all treatments. With thinning alone this decrease was most dramatic. Black cherry increased only slightly in the control stands, but decreased both in raw numbers and as a percent of total seedlings in the other three treatments. Sassafras (*Sassafras albidum*), sweet birch (*Betula lenta*), and yellow-poplar increased in all treatments, but thinning, regardless of defoliation, increased sassafras and sweet birch production dramatically. Red oak increased as a proportion of total seedlings and in actual total number of seedlings in both **T** and **C** stands, while white oak decreased. In both types of defoliated stands, red and white oak diminished over the four years. Other increases in oak species were from 37 to 56 seedlings in **DT** stands for chestnut oak and from 171 to 223 in **T** stands; however, this only represents increases of 0.1%, and 0.3% of all seedlings, respectively . In **C** stands, chestnut oak increased by 1.1%.

Non-overstory woody species can account for considerable competition for regenerating species (Table 4). Of particular note is the abundance of *Vaccinium* spp. (blueberry) in defoliated stands. The presence of this species is not a function of disturbance as much as a reflection of site characteristics. Nonetheless, blueberry abundance is likely deleterious for regeneration, and it appears that thinning helps to reduce blueberry when compared to non-thinned defoliated stands.

Table 3. Oak reproduction in all stands. Each value represents the percentage of oak contribution to total regeneration. Total regeneration includes all species, irrespective of potential to exist in the overstory.

	1989	1990	1991	1992
Stand	%	%	%	%
DT - 1	10.4	1.8	1.1	0.9
D - 1	10.0	4.2	2.4	1.5
DT - 2	9.8	11.7	11.0	10.0
D - 2	6.6	5.9	5.4	3.9
DT - 3	5.4	3.6	1.2	1.5
D - 3	8.1	5.7	4.1	0.7
C-1	2.2	*	7.6	8.4
T-1	6.0	3.8	5.9	7.3
C-2	2.9	*	6.6	5.0
T-2	3.2	4.8	4.6	7.9
C-3	7.7	*	12.7	10.1
T-3	2.3	1.6	2.8	3.8
C-4	2.2	*	2.8	1.5
T-4	4.3	6.0	4.3	4.8
C-5	4.8	*	4.0	3.3
T-5	7.2	4.5	4.3	6.2

* - no data collected for control stands in 1990

Table 4. Percentage of total individuals represented by selected species for 1989 and 1992, by treatment.

	CONTROL		THINNED		DEFOLIATED		DEFOL. & THIN.	
species	1989	1992	1989	1992	1989	1992	1989	1992
red maple	38.2	27.2	30.1	17.8	37.4	35.6	35.7	28.2
black cherry	30.0	31.7	30.0	19.8	23.6	11.9	21.8	10.9
sassafras	10.2	12.1	9.6	22.2	1.5	5.4	4.8	21.7
yellow-poplar	6.1	9.0	10.6	16.4	1.2	5.8	4.9	11.7
red oak	2.1	3.2	3.2	3.9	6.3	1.5	7.5	2.7
sweet birch	0.5	4.7	0.5	6.2	0.5	2.9	1.3	8.9
white oak	0.9	0.9	0.5	0.5	0.4	0.1	0.9	0.1
scarlet oak	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0
chestnut oak	1.0	2.1	0.9	1.2	1.2	0.4	0.4	0.5
black oak	0.0	0.2	0.1	0.1	0.1	0.0	0.0	0.1
blueberry	1.2	1.3	1.9	1.3	25.2	32.8	19.3	12.8
black gum	1.2	1.1	1.2	0.6	1.4	1.7	0.9	0.7
viburnum	4.2	2.8	3.8	1.8	0.0	0.0	0.0	0.0
spicebush	0.4	0.8	3.5	4.2	0.4	0.8	0.6	0.6

Effects of Defoliation and Thinning on Size Structure

For those seedlings most likely to reach the overstory, i.e. size class three, treatments had little effect over the four year period (Table 5). Red oak recruitment into the larger size classes was only significant in the thinned stands, with a total of 273 moving into size class 2, a group that had only 33 individuals four years previous. First year seedlings of chestnut oak (*Quercus prinus*) increased in **C, T** and **DT** stands. Black and scarlet oak (*Quercus velutina* and *Q. coccinea*) also increased in 1992 in the size classes under five-feet tall in undefoliated stands. White oak showed the poorest response among the oaks.

Certain species responded in a manner characteristic of their early successional nature. Black cherry increased in most size classes everywhere but **C** stands. Sweet birch increased in size class one in all treatment types, and in size class two, everywhere but **C** stands. Yellow-poplar followed a similar pattern, but growth of this species was evident only in **T** and **DT**. Sassafras increased in size classes under five feet in all but the **C** stands also. Thinning treatments (**T** and **DT**) appear to accelerate sassafras growth dramatically, giving rise to an abundance in the 1-5 ft size class during in the four year period.

DISCUSSION

Among studies examining regeneration as affected by gypsy moth defoliation, Allen and Bowersox (1989) sampled forests in both the Allegheny Mts. and Ridge and Valley provinces. The most abundant woody regeneration in many stands in both areas was red maple, accounting for up to 90% of all commercial regeneration in the Allegheny Mountains and 49% in the Ridge and Valley provinces. Birch was a major constituent of regeneration in the Ridge and Valley. The values for red maple in studies by Allen and Bowersox (1989), and Hix and others (1991) surpassed the quantities found at the West Virginia University Forest. Stands on the West Virginia University Forest exhibit greater species richness in regeneration than the previously mentioned studies, for black cherry and yellow poplar were commonly found, in addition to sweet birch and red mapels. Yellow-poplar seedlings ranged from 1259 (**D**) to 2585 (**DT**) seedling/acre at the West Virginia University Forest, while they were not even noteworthy in the previous work. Also, black cherry seedlings surpassed all but red maple in our study and were not noted by Allen and Bowersox (1989). Hix and others (1991) found 2333 black cherry seedlings/acre, a value comparable to the number found at the West Virginia University Forest. Sassafras averaged 1776 seedlings/acre in Allen and Bowersox's study, and 1138/acre in the **D** stands and 4567 seedlings/acre in **DT** stands. Some differences among the studies may be attribuable to sampling time, because we sampled immediately after defoliation, while the other research was no fewer than five years following defoliation.

The abundance of yellow-poplar seedlings likely reflects high site productivity (Ostaff and others 1982), and yellow-poplar regeneration may aggravate problems of oak regeneration (O'Hara 1986). This species is likely to dominate certain areas if disturbance is severe (Beck and Hooper 1986). Loftis (1983) found that any canopy opening is likely to promote yellow-poplar and lead to its dominance on good sites in the southern Appalachians. In the current study, the most abundant 1-5 feet regeneration of yellow-poplar occurs on thinned stands and thinned/defoliated stands. However, our data suggest that recruitment into the larger size classes for this species (Table 5) has been minimal. This species has increased relative to total basal area, in the overstory on two of the three **DT** stands, but increased only slightly elsewhere (Table 1) during the four year period. The increase in overstory corresponds to oak mortality in the highly disturbed stands.

In this study, black cherry seem to be assuming a role analogous to yellow-poplar in the southern Appalachians. Although total numbers of black cherry seedlings decreased over the four years of the study, the decreases were apparent in the smallest size classes. The seedlings in size class 3 increased significantly in the three disturbed treatments (Table 5). Black cherry is relatively intolerant of shade and will respond favorably when the canopy has been opened (Marquis 1990). A substantial cohort of black cherry was established pre-defoliation and pre-thinning

Table 5. Total number of seedlings per acre in three size classes. Size class 1:<1 ft. tall.; Size class 2: 1 -5 ft. tall; Size class 3: >5ft. tall, but less than 2.49 in dbh.

	size	control		thinned		defoliated		defoliated & thinned	
species	class	1989	1992	1989	1992	1989	1992	1989	1992
red	1	10490	5037	6665	3736	7887	7673	6789	6066
maple	2	688	441	258	431	11	21	43	98
	3	128	123	97	73	34	28	160	81
sweet	1	92	917	37	1168	83	525	194	1592
birch	2	40	31	58	297	4	51	60	179
	3	22	21	19	19	9	9	6	4
yellow	1	1804	1846	2458	3812	253	1259	940	2553
poplar	2	9	4	9	97	0	0	11	32
	3	0	0	1	1	0	0	0	0
black	1	249	172	215	88	253	324	160	138
gum	2	54	32	37	45	13	6	0	6
	3	40	32	26	15	30	23	11	0
black	1	6595	4776	4628	2104	3704	898	3038	727
cherry	2	2155	1605	2199	2176	1249	1153	1111	1089
	3	129	153	179	455	62	357	109	506
white	1	242	182	97	82	81	14	170	18
oak	2	17	13	15	29	0	0	6	2
	3	0	0	0	0	4	4	2	0
scarlet /	1	13	31	15	34	19	2	10	20
black oak	2	0	4	0	75	0	0	0	2
	3	0	0	0	3	0	0	0	0
chestnut	1	263	405	208	228	238	66	77	88
oak	2	37	32	12	56	2	4	0	13
	3	4	3	0	1	9	6	2	2
red oak	1	604	645	718	656	1304	275	1460	466
	2	18	9	33	273	21	19	6	81
	3	1	4	4	8	13	6	2	0
sassafras	1	2846	2383	2023	4419	311	1087	898	4235
	2	167	101	217	881	2	51	49	332
	3	0	0	0	6	0	0	0	0

and the disturbance provided the opportunity for increased growth of this regeneration. Dominance in the overstory of the post-disturbance stand is a likely consequence of vegetation present at the time of disturbance. This concept of initial floristics has been used widely to characterize forest succession (Drury and Nisbet 1973, Egler 1954, Oliver and Larson 1990) as well as describe post-defoliation succession (Ehrenfeld 1980) and oak regeneration problems (Loftis 1983, Ross and others 1986).

In disturbed stands, red maple and birch are also expected to maintain importance in the overstory because of increases in the number of seedlings, and their comparable, but rather low, mortality rates as understory trees (Lorimer 1981) and ability to exploit conditions caused by defoliation (Collins 1961). Given repeated disturbance, the **T, D,** and **DT** stands are likely to be dominated by red maple and black cherry, with a component of birch.

Oak seedling abundance, both previous and subsequent to disturbance, was somewhat low relative to other studies. The range of total oak regeneration in 1989 was from 501 (Stand **T3**) to 2538 seedlings/acre (Stands **C3** and **DT2**). In 1992 the range was 193 (Stand **D1**) to 3122 (Stand **C3**) seedlings/acre. In Ridge and Valley Forests of southern Virginia, Ross and others (1986) found over 8,000 oak seedlings/acre in mixed oak types and 5059 oak seedlings/acre in mixed hardwood types. After clearcutting in those forests, however, values dropped to 1309 and 26 seedlings/acre respectively. In relatively undisturbed oak dominated ecosystems in Michigan, Johnson (1992) found a range of 5261 to 11152 oak seedlings/acre. Allen and Bowersox (1989) found 707 seedlings/acre of red oak in post defoliated stands in the Allegheny Mountains and Hix and others (1991) 1363 seedlings/acre, while we sampled 300 seedlings/acre in **D** stands and 547 in **DT** stands. Abundance of white oak seedlings was 458 and 2002 seedlings/acre, respectively for the previous studies, and we found fewer than twenty seedlings/acre in defoliated stands.

Red oak seedlings were generally taller on the thinned stands versus unthinned stands, a finding similar to Ward (1992). When compared with stands that had been defoliated only, **DT** stands had greater numbers of scarlet/black oak and chestnut oak seedlings. In **DT** stands in 1992, there were no white oak seedlings in size class 3, and only two in size class 2, whereas red oak seedlings increased from 6 to 81 in size class 2. Thinning, followed by defoliation provided suitable microclimatic conditions for growth enhancement of red oak, but not new reproduction. The increased growth phenomenon did not occur in stands that were defoliated only, but may be a function of severity of defoliation, as the extent of defoliation and mortality was not uniform throughout the stands. The decrease in total number of oak seedlings in both **D** and **DT** stands may in part be due to seedling defoliation by the gypsy moth (Hicks and others 1993) or by a decrease in acorn production due to defoliation (Gottschalk 1989).

Small size and low numbers of oak advance regeneration are primary causes of oak regeneration failure. Based on the scarcity of oak in the larger size classes (Table 5), it seems unlikely that oak will remain as large a component in the overstory as it currently is. The percentage of red oak in the overstory did not decrease in **DT2** from 1989 to 1992 (Table 1), so defoliation and thinning would appear to be less important than pre-disturbance species composition and site factors. In contrast, for all three **DT** stands, red maple increased substantially as a proportion of basal area, and is likely more sensitive to disturbance than site. In **DT1** yellow-poplar showed a dramatic increase in proportion of overstory during those four years, but this was primarily due to pre-disturbance presence of that species. Black cherry, red maple and *Vaccinium* dominate the understory of all the defoliated stands and provide severe competition for oak regeneration.

In the control stands, oak basal area remained virtually the same over the four years. Given the seedling abundance on control stands, oak regeneration might be inadequate to contribute much to the overstory. Since the goal of thinning was to reduce gypsy moth host species abundance, oak basal area decreased. Thinning also provided an opportunity for growth of oak regeneration, assuring a presence, albeit not necessarily dominance, of oak in these stands. Similarly, Carvell and Tryon (1961) found that partial overstory removal enhanced oak production, and that oak regeneration decreased with increasing basal area. Defoliation alone, and in concert with thinning reduced oak basal area in most of the thinned and/or defoliated stands in this study, but in stand **DT2**, the red oak proportion of total basal area increased slightly.

Defoliation alone adversely affected oak regeneration to a greater extent than the defoliation and thinning combination. Silvicultural practices in advance of potential overstory defoliation is warranted for perpetuating the oak resource. In these stands, however, oak regeneration is inadequate in these stands for all treatment types, in accordance by recommendations by Sander and others (1984) for the central hardwoods. Despite a sufficient total number of seedlings, there are too few in the taller size classes to assure oak in the overstory.

CONCLUSIONS

Black cherry is likely to dominate the next generation of overstory trees in the stands in this study and in comparable areas in the Allegheny Plateau. Disturbance accelerates this transition, but the response of black cherry to defoliation and defoliation plus thinning was similar. Recruitment of black cherry into the taller size classes stands that were thinned only was slightly less than stands that were thinned and defoliated and defoliated only, underscoring the importance of disturbance for black cherry success.

Despite an abundance of shade intolerant species in the understory of these stands and an increase in the number of seedlings and saplings less than a foot tall, few of these will become canopy dominants. Recruitment into the larger size classes of these species has been minimal.

Defoliation reduced the growth and recruitment of oak, but thinning moderated the defoliation effect. Thinning, without defoliation, enhanced growth of oak seedlings. Even in undisturbed stands, however, oak regeneration over 5 feet tall was negligible after four years.

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TWO-YEAR RESULTS OF HERBICIDE RELEASED, NATURALLY-REGENERATED BOTTOMLAND CHERRYBARK AND SHUMARD OAK SEEDLINGS

John F. Thompson Jr. and Larry E. Nix[1]

Abstract: After clearcuting in bottomlands, oak seedlings that naturally regenerate are often overtopped by woody pioneer species, sprouts and herbaceous material. To improve the competitive status of three- to four-year-old oak seedlings in two bottomland stands in South Carolina, several herbicides and methods of application were used to kill and/or stunt the overtopping competition. Oak seedling diameter growth two years after treatment was nearly doubled by four of the late summer herbicide oversprays and by two of the directed applications. Height growth two years after treatment was increased by 1.9 to 2.7 feet by two of the oversprays and by one of the directed soil applications. Timing the oversprays so that competing vegetation shielded the oak seedlings from herbicide damage appeared to aid in improving the oak seedling's competitive position in the rapidly developing dominant canopy.

INTRODUCTION

In the past thirty years oak (*Quercus* sp.) has been decreasing in the bottomland hardwood stands of the Southern United States (Johnson 1984). Cherrybark oak (*Quercus pagoda* Raf.) and Shumard oak (*Quercus shumardii* Buckl.) are two high quality bottomland oak species (Miller and Lamb 1985) that are not being replaced in the bottomland hardwood forest. The ecology and biology of bottomland oaks have been investigated to determine why the number of oaks are decreasing in new stands of the bottomland forests (Johnson 1979, Guldin and Parks 1989, Clatterbuck and Hodges 1988, Hodges and Janzen 1987). Because of their slow early growth, newly established oak seedlings cannot compete successfully with sprouts and fast-growing pioneer species. Oak seedlings often grow and die back for 4 to 6 years after germination before they reach a stage of rapid height growth. The slow growth of oak seedlings after germination is compounded by animal predation and overtopping by briars, sprouts, and early successional trees and brush (Gingrich 1979, Watt 1979, Hannah 1987, Nix 1989). However, if released oak seedlings can respond and achieve a dominant position in the developing canopy (Nix 1989).

Some type of release of oak seedlings after clearcutting is usually needed to have oaks in the dominant canopy in a reasonable period with quality, size, and grade (Beck 1970, Hannah 1987). Release should be accomplished 3 to 5 years after clearcutting to allow the seedling oaks to quickly catch up with the other woody species on the site (Beck 1970). Early release should allow the oaks to be a dominant component of the canopy by the time the stand is 25 years old. Early release may also shorten the normal oak rotation by 10 to 15 years for quality oak products (Clatterbuck et al. 1985, Clatterbuck and Hodges 1988). Release of crop trees with herbicides has been accomplished successfully for many years in the management of pine stands (Clason 1978, Nelson et al. 1985, Bacon and Zedaker 1987, Nelson and Cantrell 1990). Therefore, it should be possible to release oak seedlings with careful application of the same herbicide.

During the winter of 1987 and 1988, two bottomland hardwood stands with a 40-percent oak component (basal area basis) were clearcut along the Congaree River in South Carolina. These stands apparently had a good seed crop prior to harvest, as from 3- to 5-thousand oak seedlings per acre were established (Nix and Lafaye 1993). When first examined, in the winter of 1989, these oak seedlings were overtopped by a dense canopy of blackberry, herbaceous composites, and woody seedlings and sprouts. This study was implemented to evaluate several herbicides and application methods for releasing overtopped oak seedlings developing in dense young clearcuts.

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PROCEDURE

Forty-two plots approximately 18- to 22-feet wide and 25- to 130-feet long were located along two-foot wide transects cut through the vegetation of a three- and a four-year-old clearcut, approximately one-half mile apart in the first bottom of the Congaree River near Columbia, South Carolina. The transects were placed where two or three cherrybark or Shumard oak seedlings in each of three height classes were found in an overtopped condition. The plots were placed at least 15 feet apart to reduce the chance of one herbicide treatment influencing any adjoining treatment. Each plot contained 6 to 9 seedlings located along either side of the transect. Seedlings were chosen within approximately 7 to 9 feet of the transect, but at least 2 feet away such that the cut transect did not affect the seedlings' competitive status. The seedlings were selected to be approximately 6+ feet apart to accommodate the directed herbicide treatments. The length of the plot was determined by the distribution of the seedlings and the width of the plot was determined by the overspray width and the spray apparatus boom height. The beginning and end of each plot center line was marked by a white, 4-foot, plastic pipe driven into the ground.

The treatment plots were blocked for replication according to preliminary estimates of total competition (based on a height/distance index) (Daniels 1976) on the plots (high, medium or low). Three replications of the treatments were arranged with each treatment plot was randomly placed in each of the three blocks. The oak seedlings chosen for measurement were statistically similar in height on all plots. The preferred condition of the seedlings for release was overtopped, but at least 12 inches in height. Each plot contained at least two seedlings and, if possible, three seedlings in each of the three following height classes: 1-Foot Class) 1 to 1.9 feet, 2-Foot Class) 2 to 2.9 feet, 3-Foot Class) 3 feet and taller. Each seedling chosen was marked by a wire flag and metal tag bearing a coded number.

Preliminary treatments with a quick-browning herbicide in 20% increments of the recommended release rates for pines were applied after full foliage development (mid summer) to refine the rates that were used in the treatments. The methods of treatment, herbicides, and rates which were applied in late July/early August were as follows:

Control - no treatment.

Directed (Streamline) - Application of 20% Garlon 4 (a triclopyr herbicide), 10% Cide-Kick II surfactant (JLB 1984) and 70% diesel fuel applied to the lower portion of all competing stems in a 3+ foot radius around the oak seedlings (Dow 1988, McLemore and Cain 1988, Yeiser et al. 1989). This application was accomplished with a Solo piston backpack sprayer equipped with a D2 orifice disc tip for stream application.

Directed (Spot gun) - Application of Velpar L (a hexazinone herbicide) or Escort (a metsulfuron benzoate herbicide) applied in a grid pattern of 6 - 1 milliliter spots placed approximately 3.5 feet from the oak seedlings to be released. The spots of Velpar L consisted of a mixture of Velpar L and water that was 50% of the manufacturer's product strength. The application mixture of Escort consisted of 1 oz. of Escort mixed in 2 gallons of water. Due to the fine texture of the soils on the study site, the manufacturer's rate of application for use on this texture of soil indicates that there will be a 1.5- to 2-foot non-toxic zone around the treated oak seedlings (Du Pont 1987, Rachal et al. 1988, Du Pont 1988). These applications were accomplished with a standard spot gun applicator calibrated at 1 milliliter per spot application.

Broadcast (Overspray) - Simulated aerial broadcast, an overspray applied with a boom apparatus, applied at one half and full label-recommended application rate of commercial herbicides for releasing pines with 3.2 oz. per acre of TimberSurf 90 surfactant (0.25% v/v -- commonly recommended rate in release of pines) (Timberland ca.1989) at 10 gallons total mix per acre (Dow 1988) for each of the herbicides in Table 1. The simulated aerial broadcast spray was accomplished with a CO^2-pressurized backpack apparatus with a boom extended well above the vegetation (16 feet above ground) and equipped with a single horizontally aligned flood jet nozzle. The actual area treated was calculated by using the length of the plot, the boom height, and the nominal spray width within which competing vegetation would be uniformly treated (18 to 22 feet). The spray time, spray pressure, boom height, and amount of herbicide to be used were predetermined. Measurements of the spray time and pressure were recorded for each plot so the actual

rate of application could be calculated. Spray width ranged from 18 to 22 feet depending on height of competing vegetation and number of boom sections installed, but within which competition was uniformly treated.

Table 1. Simulated aerial broadcast applications (oversprays) that were used at 100% and 50% of standard pine release rates for oak release in 3- and 4-year-old clearcut bottomland hardwood stands in South Carolina.

Herbicide	Standard Pine Release Rate
Garlon 4	1.5 qt/ac
Escort	1 oz/ac
Accord	1.5 qt/ac
Velpar L	3 qt/ac
Arsenal AC/Escort	8 oz - 0.25 oz/ac[1]

[1]The rate of 0.25 oz/ac of Escort remains the same for both the 100% and the 50% rate plots.

To determine effectiveness of the release treatments, herbaceous competition, woody competition, total competition, seedling height, and GLD (ground line diameter), were measured before and after treatment. The results of the initial reduction of competition in this study were reported in an earlier paper (Thompson and Nix 1993). Seedling height (± 0.1 foot) and GLD (± 0.1 inch) were measured after two years of growth. Data were analyzed using a complete random block design and PC-SAS (1992), Anova procedure. Means were tested at the 0.01 level of probability.

RESULTS AND CONCLUSIONS

The effectiveness of any crop tree release treatment can be judged early by the subsequent tree height and diameter growth. In this study, oak seedling height and diameter growth two years after treatment are the major criteria for evaluating the effectiveness of the herbicide release treatments. Two treatments resulted in a significant increase in height growth of released oak seedlings after two full growing seasons (Table 2). Seedlings released by the low rate, broadcast application of Escort and the high rate, broadcast application of Arsenal/Escort grew 1.5 and 0.9 feet, respectively, more than the untreated seedlings (all height classes combined). Velpar L, soil-applied by spot gun, apparently increased seedling height growth (1.2 feet more then the control); however, no difference could be statistically detected because of high variance. Nine of the thirteen treatments significantly increased oak seedling ground line diameter (GLD) as compared to the untreated seedlings (Table 2).

Table 2. Two year height and diameter growth (all seedling height classes combined) of cherrybark and Shumard oak seedlings after herbicide release in three and four year-old clearcut bottomland hardwood stands on the Congaree River in South Carolina.[1]

Herbicide Treatment	Height Growth	Diameter Growth	Damaged[3]
	-- ft --	-- inches --	-- % --
BROADCAST[2]			
ACCORD HIGH	5.6	0.68**	9
ACCORD LOW	5.9	0.68**	4
ESCORT HIGH	5.5	0.78***	9
ESCORT LOW	7.0**	0.70***	0
VELPAR L HIGH	6.1	0.72***	4
VELPAR L LOW	5.9	0.51	0
GARLON 4 HIGH	4.8	0.84***	17
GARLON 4 LOW	5.4	0.73**	14
ARSENAL/ESCORT HIGH	6.4*	0.67	0
ARSENAL/ESCORT LOW	4.0	0.39	14
SPOT GUN			
ESCORT	6.2	0.56	0
VELPAR L	6.7	0.85***	8
STREAMLINE			
GARLON 4	4.8	0.99***	29
CONTROL	5.5	0.48	0

[1]Numbers with *, **, and *** adjacent, differ only from the control at the 0.10, 0.05, and 0.01 level of probability, respectively.
[2]High indicates 100% pine release rates. Low indicates 50% pine release rates.
[3]Damaged indicates the percent of the seedlings that were either killed down to the rootstock and are resprouting or have died.

The lack of significant increase in height or GLD in the 1-foot height class, except in the most drastic treatment (Tables 3 and 4), indicates that seedlings less than two feet in height at this age would not respond to the release treatments. The 2- and 3-foot height class oak seedlings, however, would respond as indicated by the number of treatments that significantly increased seedling GLD (Table 3). Nine treatments and seven treatments of the 2- and 3-foot height class seedlings, respectively, significantly increased the GLD as compared to the untreated seedlings. However, in only four treatments in the 2- and 3-foot height class were seedlings significantly taller than the untreated seedlings.

One possible reason for the lack of significant height growth response is that the released oak seedlings appear, by casual observation, to have larger and fuller crowns than the control seedlings or oak seedlings that had less reduction in competition (Thompson and Nix 1993). Another reason for less height growth by released seedlings may be the lack of close trainer seedlings/trees. Again, from casual observation the oak seedlings that were released enough to obtain some direct sunlight, but that have close (within 3 feet), taller seedlings/trees seemed to achieve more height growth than other oak seedlings.

Table 3. Two year diameter growth (according to initial height class) of cherrybark and Shumard oak seedlings after herbicide release in three and four year old clearcut bottomland hardwood stands on the Congaree River in South Carolina.[1]

Herbicide Treatment	Diameter Growth/Height Class		
	1-Foot	2-Foot	3-Foot
	--inches--		
BROADCAST[2]			
ACCORD HIGH	0.29	0.69	0.95**
ACCORD LOW	0.39	0.90**	0.78
ESCORT HIGH	0.45	0.81**	1.12**
ESCORT LOW	0.47	0.71*	0.89**
VELPAR L HIGH	0.45	0.80**	0.87*
VELPAR L LOW	0.40	0.41	0.74
GARLON 4 HIGH	0.47	0.82**	1.08**
GARLON 4 LOW	0.40	0.90***	0.80
ARSENAL/ESCORT HIGH	0.49	0.76*	0.83
ARSENAL/ESCORT LOW	0.22	0.28	0.70
SPOT GUN			
ESCORT	0.47	0.51	0.72
VELPAR L	0.38	0.92**	1.15*
STREAMLINE			
GARLON 4	0.88**	1.10***	0.97**
CONTROL	0.42	0.47	0.58

[1]Numbers with *, **, and *** adjacent, differ only from the control at the 0.10, 0.05, and 0.01 level of probability, respectively.
[2]High indicates 100% pine release rates. Low indicates 50% pine release rates.

CONCLUSIONS

It is possible to release oak seedlings in young (3 to 4 year old) clearcuts without serious losses with careful application of selected herbicides, especially if the oak seedlings are in an overtopped condition. This overtopping, which is very common in such clearcuts, seems to protect (shield) the seedlings from lethal amounts of the herbicides. Although only two treatments induced a significant increase in seedling height growth at present (2 years after treatment), nine of the thirteen treatments caused a significant increase in diameter growth; one, the streamline application caused more than double the diameter growth of the control. It may be too early to ascertain the full effect of this release on height growth as the oak seedlings appear at present to be gaining biomass from the increased sunlight, as indicated by the increased GLD. Some of these treatments appear to have potential for establishing more oaks in the main canopy at an earlier age, thereby increasing the oak component and basal area of the young stand.

These are two-year results; and because of the oak developmental pattern, the seedlings need several more growing seasons before the full treatment effects can be determined. Evaluation of the treatments in the future and an economic analysis will better assess the full practicality of such herbicide release of young overtopped oaks in recently clearcut bottomland hardwood stands.

Table 4. Two year height growth (according to height class) of cherrybark and Shumard oak after herbicide release in three and four-year old clearcut bottomland hardwood stands on the Congaree River in South Carolina.[1]

Herbicide Treatment	Height Growth/Height Class		
	1-Foot	2-Foot	3-Foot
		--feet--	
BROADCAST[2]			
ACCORD HIGH	3.6	5.5	7.1
ACCORD LOW	4.0	7.6*	6.5
ESCORT HIGH	5.4	5.9	5.3
ESCORT LOW	6.0	6.9	7.8**
VELPAR L HIGH	5.5	6.4	6.4
VELPAR L LOW	5.3	5.6	7.0
GARLON 4 HIGH	2.9	5.2	5.7
GARLON 4 LOW	3.4	6.8	5.1
ARSENAL/ESCORT HIGH	6.1	6.7	6.5
ARSENAL/ESCORT LOW	2.3	3.4	6.5
SPOT GUN			
ESCORT	5.9	6.4	6.4
VELPAR L	3.4	7.8**	8.2**
STREAMLINE			
GARLON 4	4.8	5.8	3.7
CONTROL	5.2	5.7	5.5

[1]Numbers with * and ** adjacent, differ only from the control at the 0.10 and 0.05 level of probability, respectively.
[2]High indicates 100% pine release rates. Low indicates 50% pine release rates.

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SITE PREPARATION FOR RED OAK PLANTATION ESTABLISHMENT ON OLD FIELD SITES IN SOUTHERN INDIANA

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Abstract: Little research has been conducted on the use of mechanical site preparation or fertilization in the Central Hardwood Region. Many hardwood plantings, particularly oak plantings, on old field sites in the region have resulted in high rates of mortality, stem die-back, and slow early-growth rates. The purpose of this study was to investigate the effect of mechanical site preparation techniques and fertilization on the early survival and growth of northern red oak (*Quercus rubra* L.) seedlings planted on an old field site in southern Indiana. The study was initiated in the Spring of 1993 on a severely eroded old field site, which had a history of successive plantation failures. The treatments included all possible combinations of subsoiling, tillage, and fertilization. Each treatment combination was replicated four times. Subsoiling was accomplished to a 45 - 50 cm depth. Conventional tillage consisted of a three bottom moldboard plow followed by disking to a tillage depth of approximately 25 cm. Fertilization consisted of 336 kg nitrogen/ha, 98 kg phosphorus/ha, 93 kg potassium/ha, 224 kg calcium/ha, 56 kg magnesium/ha, 112 kg sulfur/ha, and 560 kg lime/ha. Seedling survival, diameter and height growth, root growth parameters, and foliage nutrient concentrations were measured. Deer browsing was also monitored. After one growing season, survival exceeded 99% over all treatments. Fertilization produced a 47% and 60% increase in height and diameter growth, respectively, over the unfertilized treatments. Subsoiling and tillage did not appear to significantly increase tree stem growth. Fertilized seedlings had higher foliar N concentrations, dark green foliage and 2.1 growth flushes during the first growing season compared to the yellow-green foliage and the 1.2 growth flushes produced by unfertilized seedlings. Fertilized trees were favored by the deer for browse, but resprouted vigorously following browsing. Nitrogen was the main factor limiting tree growth. Fertilization enhanced height growth by inducing multiple growth flushes throughout the growing season. Because the results of this study are preliminary, treatment effects will be monitored through the establishment phase of the plantation.

INTRODUCTION

The land-use history of southern Indiana is typical of the Midwest. From the early 19th century to the present, there has been a repeated cycle of land clearing, agronomic crop production and livestock grazing, soil erosion, declines in soil fertility, abandonment, natural succession, and once again, clearing (Callahan 1973). Within the past 30 years, economics and government farm policies have generated land clearing/reforestation cycles.

In the past, reforestation of old fields occurred through natural succession, resulting in dense stands of sassafras and other low-value species. High-value timber species such as the oaks were established only after long periods of time. Government cost-share programs such as Conservation Reserve Program (CRP) and Stewardship Incentive Program (SIP) along with upward price trends in hardwood timber markets have encouraged more landowners to realize the potential value of high quality hardwood forests. More and more landowners desire to establish hardwood plantations of known stocking. There has been mixed success with hardwood plantings on old field sites in southern Indiana. Repeated stem dieback or extremely slow initial growth often results in high mortality and subsequent low stocking (McGee and Loftis 1986, Pope 1993, Russell 1971).

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Much of the research on artificial regeneration of oaks in the *Central Hardwood Region* has concentrated on tree improvement, planting stock improvement, mycorrhizal inoculation, weed control, vertebrate pest control, and the use of tree shelters (Pope, 1993). Intensive plantation establishment measures, such as mechanical site preparation and fertilization, have not been as extensively investigated in the region. Mechanical site preparation has been studied quite extensively and is widely accepted in the south particularly with the regeneration of southern pine (Berry 1979, Burger and Pritchett 1988, Lantagne and Burger 1983, Miller and Edwards 1985, Morris and Lowery 1988). Malac and Heeren (1979) discuss the use of disking and subsoiling for improving soil conditions and reducing perennial herbaceous weed competition when establishing oaks in old fields. Seifert and others (1985) studied the effects of bedding and disking on the establishment of swamp chestnut oak (*Quercus michauxii* Nutt.) on a poorly drained site in southeastern Indiana. After five growing seasons, 52% of the seedlings in the disked treatment and 45% in the disked and bedded treatments were taller than the overall mean seedling height (152 cm) compared to 16% for the control seedlings. Subsoiling fractures restrictive layers thus allowing deeper root penetration early in the tree establishment phase and providing a greater volume of soil for root exploitation (Berry 1979, Lantagne and Burger 1983). Miller (1993) outlines a "general plantation establishment prescription" for oaks including the use of mechanical site preparation.

Fertilization has been extensively researched and widely practiced in coniferous forests of the South and the Pacific Northwest (Allen 1987), but relatively few studies have examined hardwood species response to fertilization. Foster and Farmer (1970) reported significant growth responses following nitrogen fertilization of planted northern red oak in Tennessee. Auchmoody and Smith (1977) reported on the response of natural yellow-poplar and red oak stands to fertilization in West Virginia. Johnson (1980) studied the effects of fertilization with 112 kg N/ha and 244 kg P/ha on five different species of oak seedlings planted in the Missouri Ozarks. Following eight growing seasons, fertilization significantly improved scarlet oak (*Quercus coccinea* Muenchh.) survival and significantly increased the number of black oaks (*Quercus velutina* Lam.) and scarlet oaks attaining a 6, 8, and 10 feet minimum height. Northern red, white (*Quercus alba* L.) and post (*Quercus stellata* Wangenh.) oaks did not respond significantly to fertilization.

The purpose of this study was to investigate the effect of the following three intensive site preparation methods and their combinations on first growing season survival and growth of northern red oak (*Quercus rubra* L.) seedlings planted on an old field site in southern Indiana:

1. subsoiling
2. tilling (plowing & disking)
3. fertilization.

METHODS

Site Description

The study was initiated in the Spring of 1993 on a one ha old field site located in the unglaciated sandstone-shale uplands of southern Indiana, in the Crawford Upland Section of the Shawnee Hills Natural Region (Homoya and others, 1985). The site was abandoned from agricultural use approximately 40 years ago. The soils are characterized by moderately thick loess deposits over paleosols on residuum formed from acid siltstone and sandstone. Because of past management history, most of the loess cap has been eroded away. The soils are fine-silty, mixed, mesic Typic Fragiudalfs of the Zanesville series, with a fragipan at a 60 cm depth. Because of the fragipan, the site has a perched water table during the winter and spring. The site is a south facing aspect with a 6% slope. It is severely eroded, presently dominated by broomsedge, and has a history of plantation failures. The average growing season temperature is 26°C, and average annual precipitation is 92.7 cm.

Study Design and Installation

The experimental design was a completely randomized 2x2x2 factorial with a split plot (nested design), with 4 replications and a total of 32 whole plot treatment units. The area of each whole plot treatment unit was 95 m^2. The whole plot treatments consisted of factorial combinations of subsoiling, tilling, and fertilization. The following are the eight treatment combinations:

1. control
2. fertilize
3. subsoil
4. subsoil + fertilize
5. till
6. till + fertilize
7. subsoil + till
8. subsoil + till + fertilize.

Subsoiling was accomplished in late April 1993 using a single-tine subsoiler drawn by a 80-horse-power farm tractor; equipment readily available to most landowners within the region. This produced a 45-50 cm deep slit. This was not sufficient to fracture the fragipan, but did produce a zone of fracture in the dense, clayey B_t soil horizon. In mid-May, tilling was accomplished by plowing with a 3-bottom moldboard plow followed by disking to a 20-25 cm depth.

Fertilizer was applied in late April 1993, prior to plowing and disking. Fertilization consisted of a broadcast application of urea, monoammonium phosphate, potassium-magnesium sulfate, and a fast-reacting pelletized calcitic lime, resulting in the application of 336 kg N/ha, 98 kg P/ha, 93 kg K/ha, 224 kg Ca/ha, 56 kg Mg/ha, and 112 kg S/ha. These fertilizers were chosen to provide an ample supply of all plant macronutrients and to ensure that none of these elements would become limiting (Blinn and Buckner 1989).

Within each of the 32 whole plot treatment units, three different types of 1-0 northern red oak seedlings were planted. Seedling type or planting stock was the split or nested plot treatment. The following describe the three seedling types and the number planted in each split plot treatment (Table 1):

1) 15 standard state nursery seedlings; average caliper 5.7 mm; average height 44.5 cm.

2) 4 bareroot seedlings grown under enhanced nutrient and moisture regime; average caliper 6.3 mm; average height 50.1 cm.

3) 5 containerized seedlings grown under enhanced nutrient and moisture regime; average caliper 5.8 mm; average height 49.1 cm.

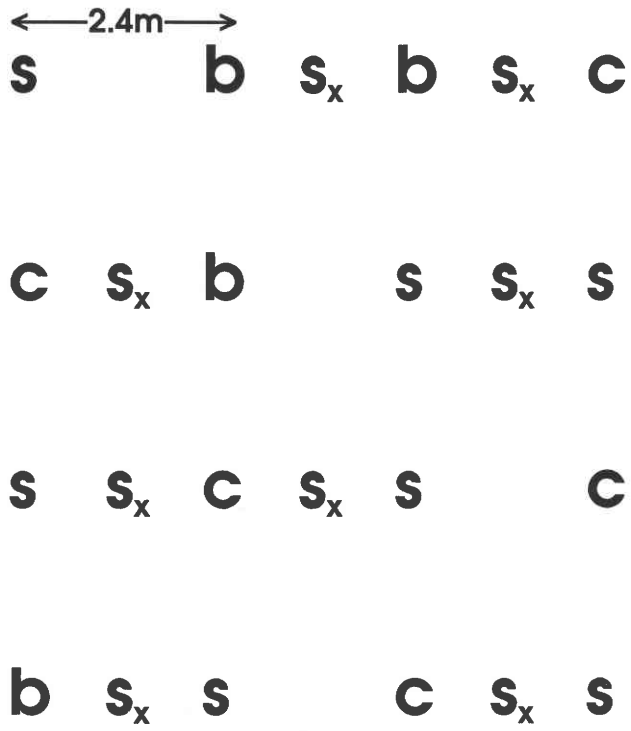
A total of 24 seedlings were planted in random fashion in each whole plot treatment unit in mid-May. All but eight of the standard state nursery seedlings were hand planted using a post hole digger on a 2.4 m x 2.4 m spacing. Trees were planted directly into subsoiled slits where that treatment occurred. All planting rows were carefully surveyed prior to treatment installation so that subsoiled slits could be readily relocated where subsoiling was followed by disking. The remaining eight standard state nursery seedlings were planted with a post hole digger on a 1.2 m x 2.4 m spacing for later excavation and root growth analyses (Figure 1).

Chemical weed control was applied uniformly over all treatments and consisted of the application of a tank mixture of simazine and atrazine, each at a rate of 2.24 kg a.i./ha at the time of planting, and glyphosate at a rate of 2.66 kg a.i./ha in July. Herbicide was applied with a backpack sprayer in 110 cm bands in the planting row. Care was taken to prevent drift onto the seedlings during the glyphosate application.

Table 1. Mean diameter growth and height growth for three different northern red oak planting stock.

Planting Stock	n	Diameter Pre-plant	Diameter Year 1	Diameter Growth	Height Pre-plant	Height Year 1	Height Growth
		(mm)			(cm)		
Std. Nurs.	480	5.71 a[1]	7.69 a	1.86 a	44.5 a	57.6 a	12.2 a
Spec. Bare.	128	6.30 b	9.20 b	2.90 b	50.1 b	71.3 b	21.2 b
Spec. Cont.	160	5.76 a	10.05 c	4.22 c	49.1 b	71.5 b	21.9 b

[1]Different letters indicate statistically significant differences at α=0.01 level of significance.



s = standard nursery stock
s_x = standard nursery stock - excavate
b = enhanced bareroot stock
c = enhanced containerized stock

Figure 1. Example of the layout of a single whole plot treatment unit.

Measurements

Caliper and height were measured on all seedlings prior to planting. The trees were measured for total height and groundline diameter following the first growing season (1993). The number of growth flushes for each seedling were

recorded as well as evidence of deer browse. On November 1, 1993, observations on tree dormancy were made. The number of trees that were not yet completely hardened off were recorded.

Following the first growing season, four of the eight standard state nursery seedlings planted for later excavation were randomly selected from each whole plot treatment unit and carefully excavated to obtain the entire root system. Lateral root development, in both a north-south and an east-west direction, and total root depth were measured in the field at the time of seedling excavation. A root volume index (RVI) was calculated by using the average between north-south and east-west lateral root extent measurements and the total root depth in the formula for the volume of a cylinder. This provided an index of the soil volume exploited by the roots.

First order laterals greater than 1 mm in caliper were counted on each of the excavated seedlings. Total length of first and second order laterals greater than 1 mm in caliper were also measured. Excavated seedlings were then clipped to sever roots from stem, oven dried at 65°C to a constant weight, and weighed to obtain stem biomass, root biomass, and total biomass. Root:shoot ratios were also calculated.

Foliar Analysis

In late July 1993, one fully expanded leaf from the top one-third of the crown was sampled from each tree in each whole plot treatment unit. All leaves sampled from a single whole plot unit were composited. Leaf samples were oven dried at 65°C and ground to pass a 1 mm sieve. Total nitrogen was determined using the micro-Kjeldahl method described by Nelson and Sommers (1973). Total P, K, Ca, and Mg were determined using wet digestion in nitric acid and hydrogen peroxide followed by analysis by inductively-coupled plasma emission spectrometry (ICP).

Data Analysis

Availability of the different seedling types used in this study precluded planting equal numbers of each type within each whole plot unit. Because of the resulting unbalanced designed, a general linear model analysis of variance (GLM-ANOVA) was used to test for overall treatment differences. In analyzing the site preparation treatment effects, the means of all three seedling types were pooled. To provide for temporal consistency in the study, the trees that were excavated in 1993/94 were excluded from the analysis of stem parameters. Root parameter data was of course only collected on the excavated trees. Fisher's least-significant difference (LSD) mean separation procedure was used to find individual treatment differences following the GLM-ANOVA. Correlations between foliage nutrient concentrations and tree growth parameters were made using simple linear regression.

RESULTS AND DISCUSSION

With the exception of mole damage to three seedlings in one of the control plots, herbicide damage to one seedling in one of the disked and fertilized plots, and damage caused by unknown agents to two other seedlings, each in separate plots, survival was over 99% across all treatments following one growing season. This may be in part attributable to the above normal rainfall that occurred throughout parts of the growing season; e.g., growing season precipitation averaged 2.4 cm/month above normal.

Planting Stock

Both the bareroot and containerized stock grown under enhanced moisture and nutrient conditions were significantly taller at the time of planting than the standard nursery stock (Table 1). After one growing season both the bareroot and containerized special planting stock were about 71 cm in height compared to 58 cm for the standard stock and they had over 75% greater height growth than the standard stock. Although the special containerized stock did not have significantly greater diameter than the standard stock at the time of planting, following one growing season it did have 46% and 127% greater diameter growth than the special bareroot stock and the standard nursery stock,

respectively. No significant interactions were detected between planting stock and site preparation treatments, therefore different seedling types did not respond in a different manner to the various treatments (Figure 2).

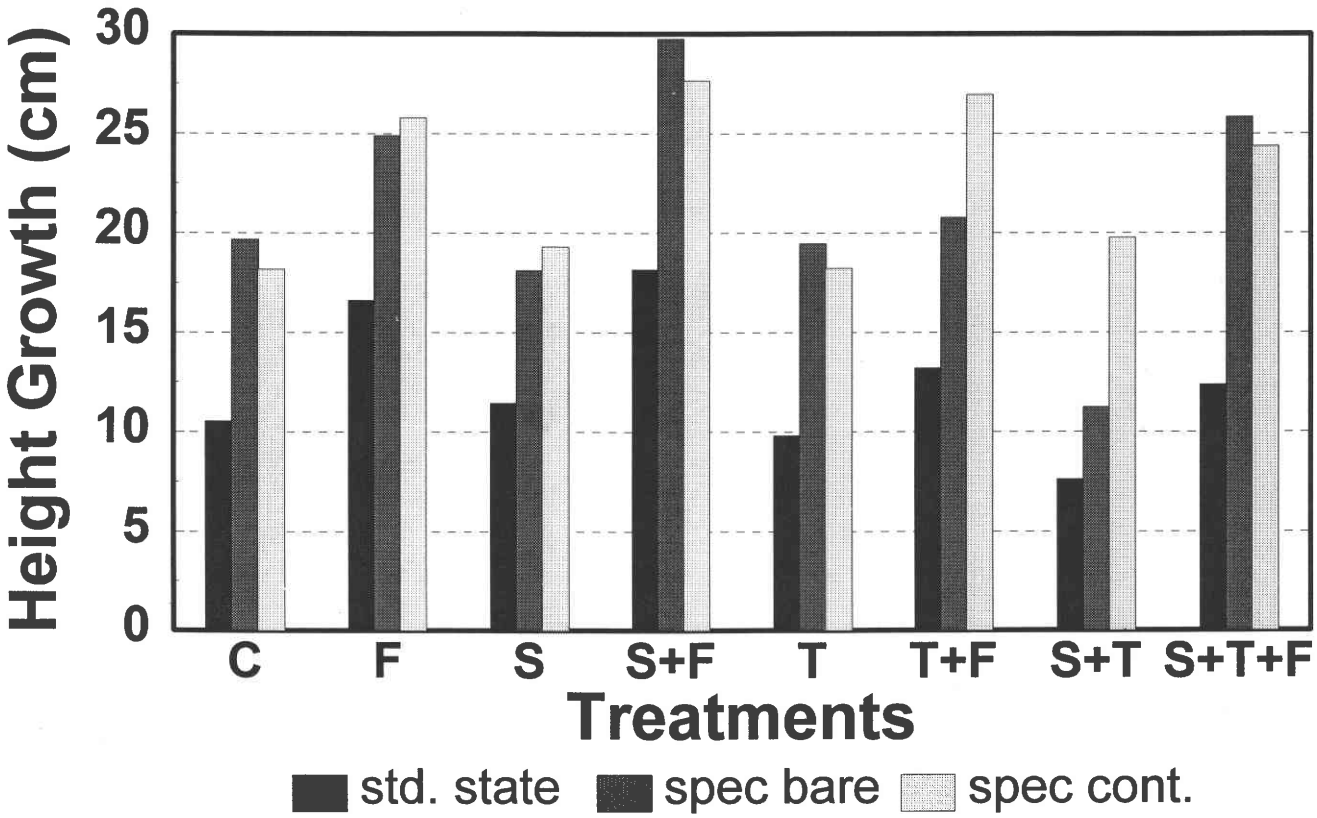


Figure 2. Year 1 height growth of northern red oak by site preparation treatment combination and seedling type.

It is generally believed that larger hardwood planting stock performs better upon outplanting than smaller planting stock (Gordon 1988, Stroempl 1985, Zaczek and others 1991). The results of this study concur with the conclusions of previous research.

Site Preparation Treatments

Root morphological characteristics. With the exception of root depth, subsoiling had little effect on root characteristics following one growing season. The roots of excavated red oak trees growing in subsoiled treatments penetrated to an average depth of 54.2 cm in the first growing season, compared to 48.3 cm for the non-subsoiled treatments (Table 2). In a year of normal or below-normal precipitation, root depth penetration during the first growing season following planting may be more critical to tree survival and growth. However, unless there is a hardpan within 45-50 cm of the soil surface, subsoiling may not produce enough additional tree growth to justify costs. On the other hand, many tree planting machines are equipped with a deep set shoe that produces a similar effect to subsoiling. Rooting depth may prove to be an important factor in subsequent years of the tree establishment phase.

Table 2. Root morphological parameters for excavated standard state nursery seedlings growing under factorial combinations of subsoiling, tilling, and fertilization.

Treatment	n	Depth[1]	Root[2] Volume Index	1st & 2nd[3] Order Lateral Length	No.[3] 1st Order Laterals	Fibrous	Biomass[4]	Root: Shoot
		(cm)	(cm^3)	(cm)		(%)	(g)	
Subsoil								
No	64	48.3 a[5]	183097	554	17.7	39	45.0	1.37
Yes	64	54.2 b	229473	582	18.7	23	46.5	1.37
Till								
No	64	49.9	162296	507	17.4	20 a	43.4	1.36
Yes	64	52.6	250273	629	19.0	42 b	48.1	1.38
Fertilize								
No	64	48.8	170525	523	17.8	47 a	42.6 a	1.38
Yes	64	53.7	242045	614	18.5	16 b	49.0 b	1.36

[1]Total depth of root penetration into the soil.
[2]Calculated from the mean of the north-south and east-west measured lateral root extent and root depth used in the formula for a cylinder. Meant to approximate the total volume of soil being exploited by the root system.
[3]Caliper greater then 1 mm.
[4]Oven-dry weight of roots
[5]Different letters indicate statistically significant differences at α=0.05 level of significance.

Tilling, as well, appeared to have little effect on root characteristics (Table 2). Although the mean rooting volume index for the tilled treatment was substantially larger (54%) than that for the non-tilled treatment, the inherently high level of variation precluded any differences from being statistically significant.

Fertilization resulted in trees with greater total root biomass (49 g) than non-fertilized trees (42.6 g) (Table 2). Fertilization did not affect the root:shoot ratios of the trees. None of the treatments had a significant effect on the number of first-order lateral roots greater than 1 mm in diameter nor on the total length of first- and second-order lateral roots greater than 1 mm in diameter.

The importance of early root growth and development in northern red oak seedlings is reported in Grime and Jeffrey (1965). Crow (1988) underscores the importance of a "large, well-developed, well-established root system" as a "key factor related to rapid seedling growth." Repeated shoot dieback is reported to be an adaptive strategy of northern red oak in building up a sufficiently large root system and increasing the root:shoot ratio. Fertilization may aid in the early development of a larger root system without the loss of stem growth due to dieback.

Stem growth. Fertilization increased height growth by 47% and resulted in a mean tree height over 5 cm greater than that for non-fertilized trees after one growing season (Table 3). Fertilization also increased diameter growth by 60%. There were no significant interactions in height growth among site preparation treatment effects (Figure 2). Neither subsoiling nor tilling increased height or diameter growth after one growing season. In fact tilling appeared to have a marginally negative, though not statistically significant effect on height growth. Both tilling and fertilization produced greater stem biomass in the excavated trees, while fertilization alone produced greater total tree biomass than non-fertilized treatments (Table 3).

Table 3. Mean diameter and height growth for nonexcavated northern red oak averaged across all seedling types and stem biomass, and total tree biomass for excavated standard state nursery northern red oak growing under factorial combinations of subsoiling, tilling, and fertilization.

Treatment	n	Diameter Pre-plant	Yr. 1	Dia. Growth	Height Pre-plant	Yr.1	Ht. Growth	Shoot[1] Biomass	Tot.[1] Biomass
		(mm)			(cm)			(g)	
Subsoil									
No	318	5.87[2]	8.76	2.80	47.6	66.5	18.3	32.5	77.5
Yes	318	5.98	9.21	3.19	48.2	67.1	18.5	34.0	80.5
Till									
No	318	6.00	8.81	2.79	47.8	68.1	20.1	31.8 a	75.2
Yes	318	5.85	9.15	3.19	48.0	65.5	16.8	34.7 b	82.8
Fertilize									
No	318	5.88	8.27 a	2.33 a	47.2	64.0 a	14.9 a	30.8 a	73.4 a
Yes	318	5.97	9.69 b	3.66 b	48.6	69.5 b	21.9 b	35.7 b	84.7 b

[1]Only includes excavated trees which were all standard nursery stock.
[2]Different letters indicate statistically significant differences at α=0.05 level of significance.

Fertilization increased the number of flushes during the growing season (Table 4). Non-fertilized trees averaged 1.2 flushes while fertilized trees averaged 2.1 flushes. Subsoiling did not influence numbers of flushes, but tilling negatively influenced the number of flushes. In fact there appeared to be an interaction between tilling and fertilization; tilling reduced the number of flushes occurring in fertilized treatments (Figure 3).

Tilling appeared to reduce the ability of fertilization to induce multiple growth flushes, in the first growing season. One explanation for this phenomenon might be the more open, harsh environment that tilling creates for the above-ground portion of the seedling. In non-tilled treatments, fertilization resulted in the growth of tall, luxuriant herbaceous vegetation between the planting rows. Excellent weed control within the planting row removed competition for moisture and nutrients but allowed between-row vegetation to moderate sunlight and wind exposure and thus moderate soil moisture and temperature and ambient air temperature immediately around the seedlings. Herbaceous growth between planting rows in the tilled treatments was low growing and relatively sparse. Crow (1988) cites a number of studies that indicate that northern red oak benefits from partial shade because of its moderating influence on temperature and evapotranspiration, while young, newly established seedlings may suffer from water stress under full sunlight. Thus, greater exposure that resulted from tilling may have reduced the trees' ability to produce multiple growth flushes, even under the favorable moisture conditions of the 1993 growing season and the abundant nutrient supplies of the fertilization treatment.

Tilling may have also resulted in increased N leaching losses. Unless surface soil compaction is a major concern, tilling may produce no early growth advantages where effective chemical weed control is used. Tilling also raises erosion and water quality concerns, particularly on the extremely erosive soils of the unglaciated region of southern Indiana. However, tilling may provide more long-term benefits that will be realized in future growing seasons.

There is a concern that excessive fertilization may delay winter dormancy in hardwoods and result in frost or winter damage. On November 1, 1993, 9% of the fertilized trees were not yet dormant, while only 1% of non-fertilized trees were not yet dormant (Table 4). This is not a perfect measure of the length of growing season created by the different

Table 4. Mean number of growth flushes, percent deer browse, and percent of trees hardened off as of Nov. 1, 1993 for nonexcavated northern red oak growing under factorial combinations of subsoiling, tilling, and fertilization.

Treatment	n	No. Flushes	Deer Browse	Percent of[1] Trees Not Hardened-off
			(%)	
Subsoil				
No	318	1.56[2]	22	3
Yes	318	1.72	18	7
Till				
No	318	1.74 a	22	6
Yes	318	1.53 b	17	5
Fertilize				
No	318	1.20 a	13 a	1 a
Yes	318	2.08 b	27 b	9 b

[1]Trees with succulent growth and incompletely developed buds on the newest flush were considered not yet hardened off.
[2]Different letters indicate statistically significant differences at α=0.05 level of significance.

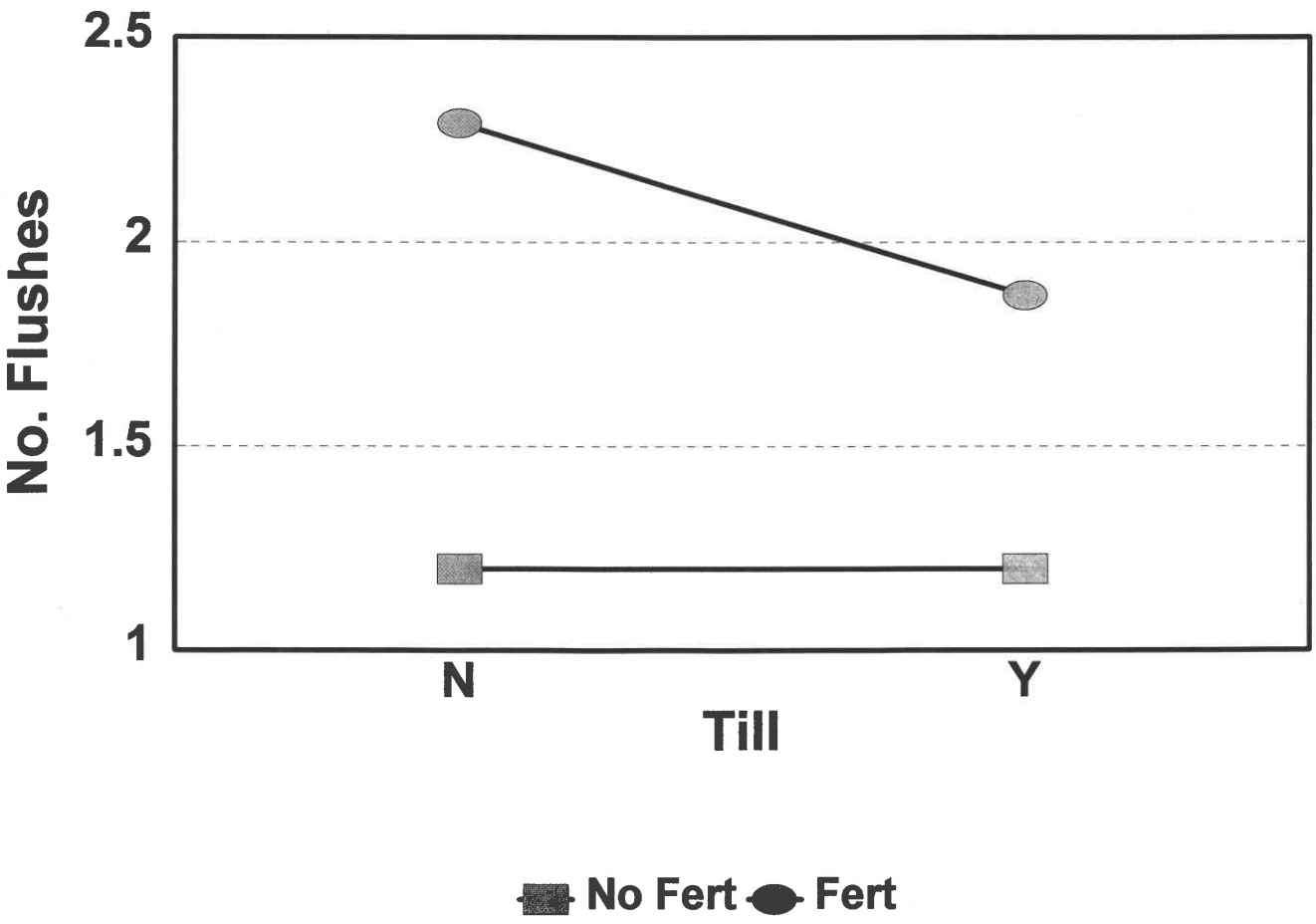


Figure 3. Interaction in the number of growth flushes between tilled treatments and fertilization treatments.

treatments, but it gives some indication of how fertilization may influence dormancy and stem hardening in anticipation of cold temperatures. The first frost at the study site occurred in mid-October while the first snow accumulation occurred in late October, 1993. Many of the fertilized trees still bore green leaves that did not show signs of impending senescence. The winter of 1993/94 was marked by below normal temperatures with periods of as long a one week where temperatures dropped to as low as -18° C. By April 1994, none of the seedlings in any of the treatments were damaged by the harsh winter conditions.

Many studies on hardwood regeneration in the *Central Hardwood Region* are incomplete without assessing the impacts of deer browsing. In fact, the deer seemed to prefer fertilized trees (Table 4). By the end of May of the first growing season, browsing had occurred on 27% of the fertilized trees while only 13% of the non-fertilized trees were browsed. In spite of the heavy browse, fertilized trees were able to respond to the browsing by reflushing and thus still outperformed the non-fertilized trees.

Nutrient relations. It appeared that in the first growing season, fertility was an overriding factor on this site. If any nutrients were possibly limiting tree growth, fertilization should not only produce an increase in growth rate but also increase concentrations of those nutrients in the foliage. Foliage nitrogen (N) concentrations (2.12 %) were significantly higher in fertilized trees than concentrations in non-fertilized trees (1.87 %) (Table 5). Phosphorus (P) and Calcium (Ca) concentrations were also significantly greater in the foliage of fertilized trees (0.11 % and 0.87 %, respectively) than non-fertilized trees (0.103 % and 0.78 %, respectively). Foliage potassium (K) and magnesium (Mg) concentrations did not respond to fertilization. Foliage color reflected these differences in nutrient concentrations. Most fertilized trees had dark green foliage while most non-fertilized trees had a greenish-yellow or even a very chlorotic appearance.

Table 5. Mean foliage nutrient concentrations for northern red oak growing under factorial combinations of subsoiling, tilling, and fertilization.

Treatment	n	N	P	K	Ca	Mg
		(%)				
Subsoil						
No	16	1.97[1]	0.107	0.587	0.867	0.296
Yes	16	2.01	0.106	0.568	0.793	0.299
Till						
No	16	1.95 a	0.107	0.588	0.861	0.306
Yes	16	2.03 b	0.106	0.568	0.798	0.289
Fertilize						
No	16	1.86 a	0.103 a	0.588	0.782 a	0.293
Yes	16	2.12 b	0.110 b	0.568	0.878 b	0.301

[1]Different letters indicate statistically significant differences at α=0.05 level of significance.

Soil fertility appeared to be the main factor limiting tree growth. Therefore, height growth should be correlated with foliage N concentration. In fact, there was a significant relationship between height growth and foliage N concentration ($p < 0.0133$), following a simple linear regression, but the two variables were not strongly correlated (R^2 =0.16)(Figure 4). Neither P nor Ca concentrations were highly correlated with height growth and it is uncertain whether any other of the applied nutrients were necessary to produce this growth response. As increased N availability

increases growth rates, demand for all other nutrients increases as well, therefore a balance of nutrients may produce a more long-term, sustained higher level of productivity.

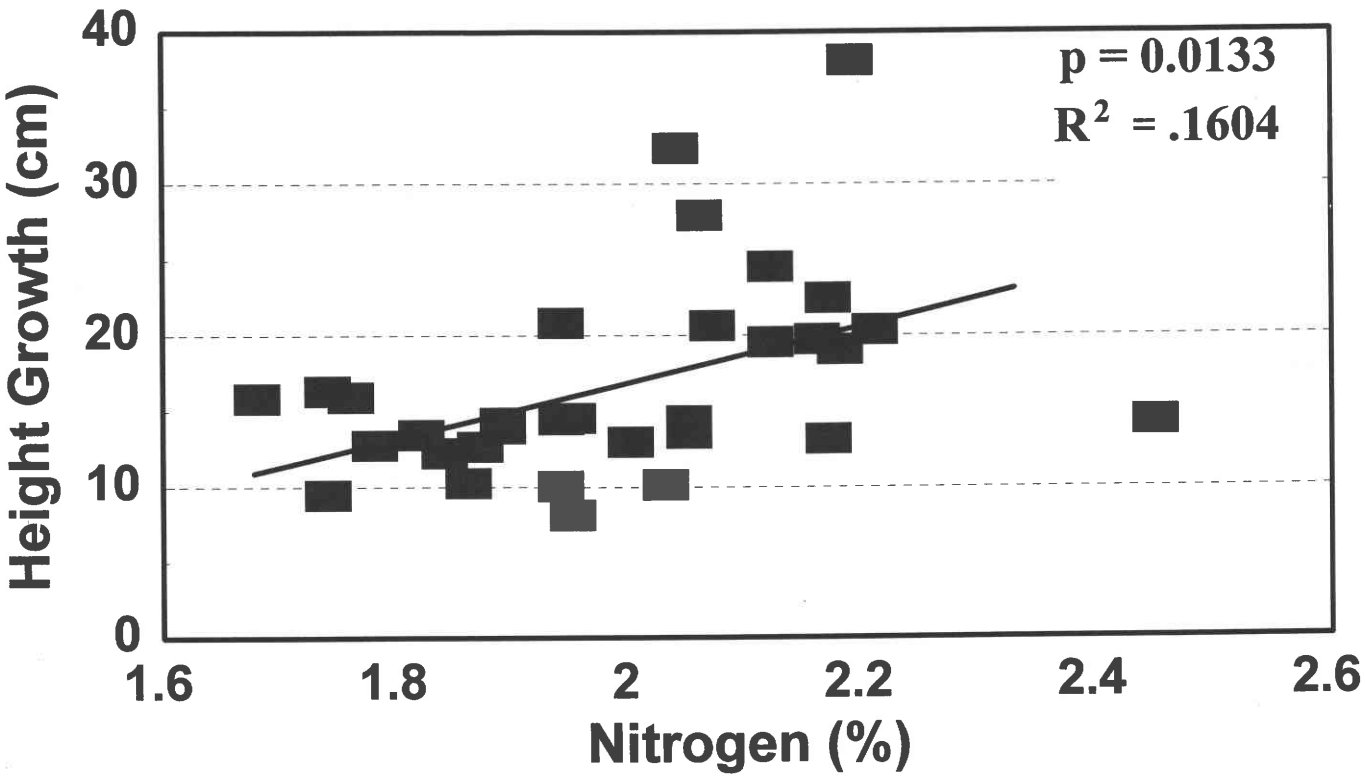


Figure 4. First-year height growth as a function of foliage N concentration for northern red oak growing under factorial combinations of subsoiling, tilling, and fertilization.

Although there was not a strong correlation between height growth and foliar N concentration, the number of flushes as a function of foliage N concentration was highly significant ($p>F=0.0001$) and strongly correlated ($R^2=0.45$)(Figure 5). Height growth was also strongly correlated ($R^2=0.56$) with the number of growth flushes (Figure 6). Therefore, the increased availability of N appeared to increase height growth indirectly by increasing the number of flushes during the growing season. Longman and Coutts (1974), Borchert (1975), and Reich et al. (1980) all reported on the occurrence of episodic shoot growth of northern red oak or other oak species; episodic shoot growth defined as periods of shoot elongation interspersed by periods of rest within the same growing season. Crow (1988) reported that under normal field conditions, oaks produce only one flush of growth and surmised that rapid growth of oak seedlings depends on multiple growth flushes within a single growing season. He also concluded that more information was needed on factors that control recurrent flushing so that conditions could be influenced or manipulated to encourage multiple growth flushes. Nutrient availability, particularly N, appears to be one of those controlling factors, and fertilization may be one way to influence site conditions to promote multiple growth flushes.

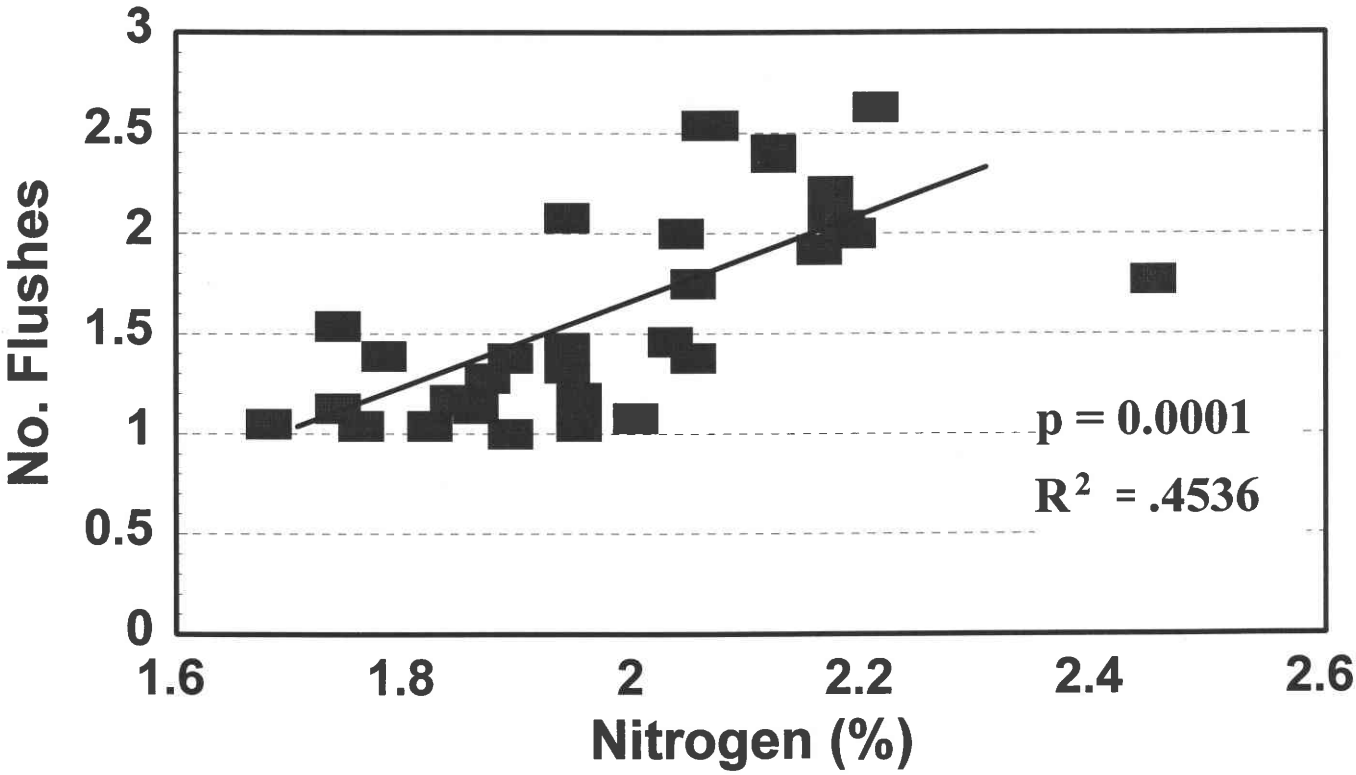


Figure 5. Number of growth flushes in the first year as a function of foliage N concentration for northern red oak growing under factorial combinations of subsoiling, tilling, and fertilization.

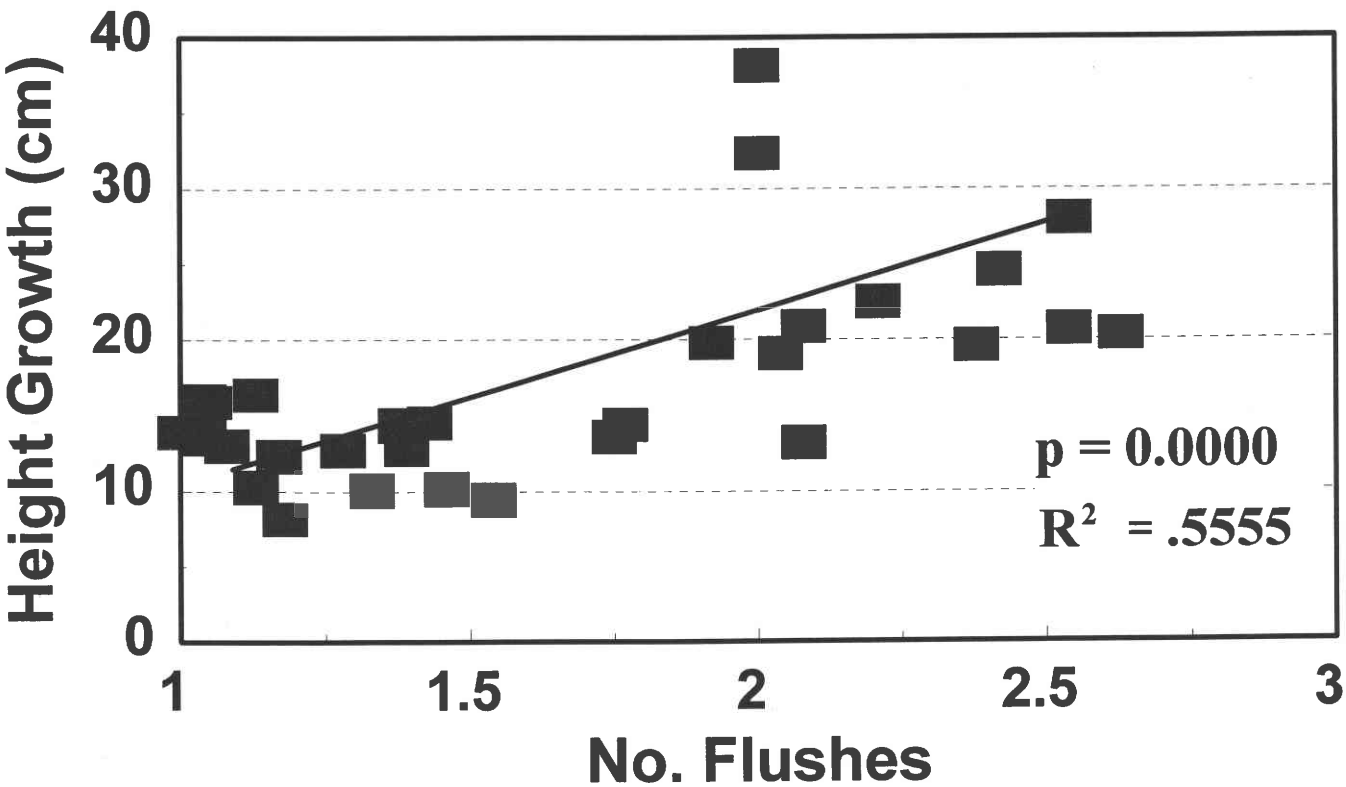


Figure 6. First-year height growth as a function of the number of growth flushes for northern red oak growing under factorial combinations of subsoiling, tilling, and fertilization.

CONCLUSIONS

The use of larger or graded planting stock significantly improves early northern red oak seedling establishment and growth. This study, along with many other studies show the importance of either growing larger stock in the nursery or culling or grading stock prior to planting.

Subsoiling improved rooting depth but no other root or stem parameters in the first growing season. Fracturing the subsoil horizon and allowing deeper root penetration at an early stage may have been more critical in a normal- or below normal- precipitation year. Tilling did not improve root growth and, with the exception of total shoot biomass, did not improve height or diameter growth. Tilling actually reduced the number of flushes in the first growing season presumably because of the more exposed condition it created for the seedlings.

Soil fertility was the most limiting factor on this site and is probably limiting on many other similar old field sites throughout the region. A high rate of N fertilization increased height growth, diameter growth, and total root biomass without altering root:shoot ratios. An increase in height growth was accomplished primarily by increasing the number of growth flushes. Fertilization of hardwood plantations in the Central Hardwood Region is not commonly practiced and rarely even considered as a viable silvicultural tool. Many hardwood plantings are made on recently row-cropped soils where tree seedlings benefit greatly from residual soil fertility. Although the results of this study are preliminary, old field sites or other low-fertility sites may be likely candidates for fertilizer applications to improve early oak establishment and growth. Treatment effects will be monitored through the establishment phase of the plantation.

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AN EIGHT-ACRE BLACK WALNUT PLANTATION:

HISTORY AND OBSERVATIONS 1982 - 1994

Charles J. Saboites[1]

Abstract: In 1982 a black walnut (*Juglans nigra*) plantation was partly established by planting 200 1-0 seedlings on the first bench adjacent to Copper Creek near its mouth draining into the Clinch River, Scott County, Virginia. In the following years, 50-500 1-0 black walnut seedlings, supplemented by transplanting germinated nuts in failed spots, were planted on the first and second benches until a total of approximately 1600 were established through 1990.

Deer, rabbits, ground hogs, mice, fusarium and herbicide drift from a 175-foot tall railroad trestle all affected the growth of the seedlings as recorded in a diary kept by the author to back up tax credits and amortization claims under the 1980 Federal Reforestation Act.

Site preparation was done by chain saw, hand tools, hack and squirt method and with the use of a propane soldering torch.

The average diameter at breast height and the average height of selected 1-0 pruned crop trees planted in 1982, 1983, 1985 and 1986 were 6.16, 4.30, 3.40 and 3.66 inches and 32.59, 23.19, 19.78, and 18.78 feet respectively in the fall of 1994.

The paper is specific as to feelings of failure, success and experiences in establishing the plantation.

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DEVELOPMENT AND QUALITY OF REPRODUCTION IN TWO-AGE CENTRAL APPALACHIAN HARDWOODS -- 10-YEAR RESULTS

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Abstract: Silvicultural practices that promote two-age stand structures have the potential to meet a wide range of forest resource goals. Such practices can overcome perceived disadvantages associated with clearcutting and still provide sustainable yields of desirable timber products and other woodland benefits. Forest managers need information on stand development following two-age harvests to develop prescriptions that meet diverse landowner objectives over long planning periods. Species composition, distribution, and stem quality of commercial hardwood reproduction were evaluated in four central Appalachian hardwood stands 10 years after a two-age regeneration harvest. Regeneration harvests cut all stems 1.0 inch d.b.h. and larger except for 12 to 15 codominant residual trees/acre comprising 17 to 25 ft^2 basal area/acre. After 10 years, codominants in the regenerated stand exhibited similar species composition, abundance, distribution, stem size, and quality as that previously reported for clearcuts in similar stands. Competition from the much taller residual trees in the older age class apparently has no effect on the 10-year development of the new age class. Forest managers can use the information provided to determine if two-age silvicultural practices applied in similar stands have the potential to meet specific stand management goals.

INTRODUCTION

Two-age management in central Appalachian hardwoods has the potential to provide a variety of aesthetic, habitat, and timber production benefits that may satisfy a wide range of land management objectives. Such practices involve leaving 15 to 20 codominant residual trees per acre, and perhaps some flowering shrubs or den trees for aesthetics and wildlife, and cutting all other stems 1.0 inch diameter breast height (d.b.h.) and larger. After cutting, two-age stands initially resemble those following a seed-tree cut. However, in two-age stands the codominant residual trees are retained for many years, often as long as a second rotation. The presence of a few codominant residual trees greatly improves aesthetics compared to that in clearcut stands (Figure 1).

Light conditions on the forest floor following a two-age cut are similar to those expected after traditional clearcutting or seed-tree practices, and result in regeneration of a variety of both shade-intolerant and shade-tolerant species (Smith and others 1989). As new reproduction becomes established and grows beneath the residuals, the vertical structure of the stand includes two distinct height strata. These strata provide a diverse habitat, particularly for migratory bird species that forage among the crowns of mature hardwoods (Nichols and Wood 1995). A brushy cover characteristic of young even-aged stands is present for many years, thus providing cover and food for a variety of wildlife species.

Logging operations in second-growth hardwoods to develop two-age stand structures are economical in most Appalachian markets (Miller and Baumgras 1994). To maintain a two-age stand structure, regeneration harvests are conducted at minimum intervals equal to one-half the length of a typical even-age rotation. In the central Appalachians, where economic rotations for sawtimber production are approximately 60 to 80 years, two-age harvests can be applied at least every 40 years. Preliminary results indicate that two-age regeneration practices can provide sustainable yields of desirable commercial timber products and other woodland benefits.

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Figure 1. A two-age central Appalachian hardwood stand 5 years after a regeneration harvest.

This report summarizes 10-year results on the quality and development of natural regeneration resulting from two-age harvests in four central Appalachian hardwood stands. Although diameter growth of the residual codominant trees is stimulated by removal of surrounding competitors, height growth of mature residual trees increases very little (Smith and Miller 1991). Similar to even-age stands, a new age class becomes established and grows rapidly in both total height and d.b.h. for many years. An earlier report described the application methods and summarized preliminary 5-year results for two stands (Smith and others 1989).

STUDY AREAS

Four study areas were established in cooperation with the Monongahela National Forest on two ranger districts and on the Fernow Experimental Forest in north-central West Virginia. Stand size ranged from 10.2 to 14.8 acres. Two stands were located on each of two northern red oak site classes, site index (SI) 70 and 80 at base age 50 years. The stands were uncut, second-growth central Appalachian hardwoods with an average age of about 75 years that became established after heavy logging in the early 1900's. Periodic fire was common throughout the study area as the stands became established, though no fires have occurred in these areas in the last 50 years. Chestnut blight also killed some large trees during the 1930's and resulted in patchy reproduction before and during World War II.

In general, soils on the study areas are medium-textured and well-drained, derived from sandstone shale with occasional limestone influence. The average soil depth in the general area exceeds 3 feet. However, the Olson Tower stand is on a Cookport silt loam soil with a fragipan at a depth of about 20 inches, resulting in shallow-rooted trees and problems with soil compaction on skid trails and log landings. Annual precipitation averages 59 inches and is well distributed throughout the year. The growing season averages 145 frost-free days.

METHODS

Regeneration harvests were applied in the four stands by retaining 12 to 15 codominant trees per acre and cutting all other stems 1.0 inch d.b.h. and larger (Table 1). In selecting the residual trees, the goal was to leave a residual basal area of approximately 20 ft^2/acre. Residual trees were selected using the following criteria:

Species -- northern red oak (*Quercus rubra* **L.**), yellow-poplar (*Liriodendron tulipifera* **L**), black cherry (*Prunus serotina* **L.**),
Crown class -- dominant or codominant,
Vigor -- no evidence of epicormic branches or other sign of decline,
Risk -- no disease, low forks, shallow roots, or other risk factors,
Quality -- current or potential high-quality butt log, and
Spacing -- residual trees well distributed throughout the stand.

Logging operations to remove merchantable sawtimber (11.0 inches d.b.h. and larger) were conducted between 1980 and 1983. Nonmerchantable stems were felled and left on the site. A truck-crane cable system was used to log the Fish Trough stand and a portion of the Shavers Fork stand. Wheeled skidders were used in the Riffle Creek and Olson Tower stands, and the remaining portion of the Shavers Fork stand.

Reproduction data were obtained before logging, and at 2, 5, and 10 years after logging from 172 permanent sample points located along systematic grids throughout the four study areas. At each point, a 1/1000-acre and 1/100-acre circular plot were used to sample small and large reproduction, respectively. Small reproduction was defined as woody stems at least 1 foot tall and less than 1.0 inch d.b.h. Large reproduction was defined as woody stems 1.0 inch d.b.h. and larger. Species, d.b.h., stem origin, quality, and crown class were recorded for each tree observed on a plot. Tree quality was based on the potential of individual trees to become sawtimber crop trees in the future. Trees with straight, clean boles, well-developed crowns, and no evidence of potential problems were classified as good. Trees with low forks, crooked or leaning stems, or evidence of low vigor were classified as poor.

RESULTS

Before harvest, small reproduction varied in terms of species composition and total number of stems per acre among the four stands (Table 2). On SI 70, shade-tolerant species such as sugar maple (*Acer saccharum* **Marsh.**), red maple (*Acer rubrum* **L.**), and American beech (*Fagus grandifolia* **Ehrh.**) accounted for 79 percent of small reproduction in

Table 1.--Summary per-acre data for four central Appalachian hardwood stands before and after a two-age regeneration harvest

	Number of trees		Basal area		Volume			Average d.b.h.	
Stand	5.0-10.9	11.0+	5.0-10.9	11.0+	5.0-10.9	11.0+	11.0+	5.0-10.9	11.0+
	---- No./acre ----		---- ft^2/acre ----		---- ft^3/acre ----		Mbf/acre[a]	----- Inches ----	
SI 70									
Riffle Creek - 14.8 acres									
Initial	120.2	68.6	41.3	87.7	741	2,202	12,932	7.9	15.3
Cut	118.9	57.4	40.6	70.9	727	1,773	10,305	7.9	15.0
Residual	1.3	11.2	0.7	16.8	14	429	2,627	9.9	16.6
Olson Tower - 12.1 acres									
Initial	100.5	79.9	30.4	102.9	519	2,593	15,721	7.4	15.4
Cut	99.2	65.9	29.7	84.9	505	2,138	12,987	7.4	15.4
Residual	1.3	14.0	0.7	18.0	14	455	2,734	9.9	15.4
SI 80									
Fish Trough - 13.1 acres									
Initial	48.9	72.2	16.5	112.7	316	3,058	20,630	7.9	16.9
Cut	48.7	58.1	16.4	88.0	314	2,379	15,697	7.9	16.7
Residual	0.2	14.1	0.1	24.7	2	679	4,933	9.6	17.9
Shavers Fork - 10.2 acres									
Initial	110.6	63.4	34.1	81.9	626	2,175	14,221	7.5	15.4
Cut	110.2	50.7	33.9	61.8	621	1,627	10,363	7.5	15.0
Residual	0.4	12.7	0.2	20.1	5	548	3,858	9.6	17.0

[a]International 1/4-inch rule.

the Riffle Creek stand, and shade-intolerant black cherry accounted for 87 percent of small reproduction in the Olson Tower stand. On SI 80, small reproduction in the Fish Trough stand was mostly sugar maple and American beech (84 percent), and mostly American beech and white ash (*Fraxinus americana* L.) (74 percent) in the Shavers Fork stand.

The distribution of small reproduction before harvest was typical of second-growth central Appalachian hardwood stands on good growing sites (Trimble 1973). In general, more than 80 percent of the survey plots had at least one seedling or sprout of a commercial species present. For the Olson Tower stand, 69 percent of the survey plots had at least one black cherry stem present before harvest. For the other stands, more than 60 percent of the survey plots had at least one sugar maple or American beech seedling present before harvest.

Table 2.--Summary of small reproduction[a] of commercial species before and 2 years after a two-age regeneration harvest

Before cut		After 2 years		Before cut		After 2 years	
Species	No./acre	Species	No./acre	Species	No./acre	Species	No./acre
			SI 70				
Riffle Creek				Olson Tower			
Beech	1,575	Red maple	1,170	Bl. cherry	4,311	Bl. cherry	9,600
S. maple	745	Beech	1,170	Beech	467	Beech	844
Red maple	340	Ch. oak	957	Basswood	67	R. maple	689
Red oak	255	S. maple	532	Others	112	S. maple	200
Bl. cherry	128	Red oak	489	--	--	Bl. birch	178
White oak[b]	128	Bl. cherry	298	--	--	Red oak	44
Ch. oak	106	Sassafras	234	--	--	Others	133
Others	105	White ash	128	--	--	--	--
--	--	Others	64	--	--	--	--
Total	3,382	Total	5,042	Total	4,957	Total	11,688
			SI 80				
Fish Trough				Shavers Fork			
S. maple	833	S. maple	2,033	Beech	1,200	Y-poplar	3,600
Beech	300	Y-poplar	1,733	White ash	520	Beech	2,300
Wh. ash	167	Beech	1,433	S.maple	380	Red oak	2,000
Others	116	Sw. birch	467	Red oak	40	White ash	1,640
--	--	Bl. cherry	433	Others	180	S.maple	700
--	--	Elm	167	--	--	Bl. cherry	220
--	--	Others	100	--	--	Others	1,800
Total	1,416	Total	6,366	Total	2,320	Total	12,260

[a]Small reproduction includes woody commercial species 1.0 feet tall to 0.9 inches d.b.h.
[b](*Quercus alba* L.).

Two years after harvest, small reproduction in the study areas averaged 9,100 commercial stems per acre composed of 60 percent seedling-origin stems (Table 2). On SI 70, red maple, American beech, black cherry, and sugar maple were the most abundant species. Yellow-poplar, American beech, and sugar maple were the most abundant species on SI 80. In the Riffle Creek and Shavers Fork stands, there were 489 (17 percent were sprouts) and 2,000 (65 percent were sprouts) northern red oak stems per acre, respectively (Table 2). Some northern red oak was present in these stands before harvest.

Five years after harvest, the canopy of the new age class developing beneath the residual overstory trees had not closed. For large reproduction 1.0 inch d.b.h. and larger, species composition and number of stems per acre varied among the four stands (Table 3). In the Olson Tower stand, reproduction was dominated by black cherry, averaging 200 stems per acre. The Riffle Creek stand also had more than 200 stems per acre, distributed among a variety of species including chestnut oak (*Quercus prinus* L.), red maple, yellow-poplar, northern red oak, and American beech. Large reproduction in stands on the more productive sites was dominated by yellow-poplar, basswood (*Tilia americana* L.), cucumbertree (*Magnolia acuminata* L.), and sugar maple. The Shavers Fork stand had more than 350 yellow-poplar stems per acre, mostly of seedling origin.

Table 3.--Summary of large reproduction[a] of commercial species 5 years and 10 years after a two-age regeneration harvest

After 5 years		After 10 years		After 5 years		After 10 years	
Species	No/acre	Species	No/acre	Species	No/acre	Species	No/acre
			SI 70				
	Riffle Creek				Olson Tower		
Ch. oak	62	Beech	179 (51)[b]	Bl. cherry	200	Bl. cherry	744 (130)
Red maple	53	Bl. birch	145 (36)	Others	18	Red maple	71 (25)
Y-poplar	43	Red maple	138 (32)	--	--	Red oak	42 (20)
Red oak	26	Red oak	128 (31)	--	--	Beech	38 (10)
Beech	15	Ch. oak	96 (31)	--	--	Others	38 (12)
Others	41	Sug. maple	89 (27)	--	--	--	--
--	--	Y-poplar	87 (40)	--	--	--	--
--	--	Bl. cherry	55 (20)	--	--	--	--
--	--	Others	145 (41)	--	--	--	--
Total	240	Total	1,062 (94)	Total	218		933 (130)
			SI 80				
	Fish Trough				Shavers Fork		
Basswood	70	Y-poplar	310 (88)	Y-poplar	352	Y-poplar	434 (72)
Sugar maple	57	Sug. maple	273 (70)	Cucumber	53	Beech	94 (19)
Y-poplar	33	Bl. birch	103 (39)	Sug. maple	16	Bl. birch	74 (17)
Other	70	Basswood	93 (74)	Red maple	16	Red maple	64 (28)
--	--	Others	221 (53)	Others	57	Sug. maple	54 (16)
--	--	--	--	--	--	Red oak	44 (16)
--	--	--	--	--	--	Others	208 (41)
Total	230	Total	1,000 (117)	Total	494	Total	972 (89)

[a] Large reproduction includes woody commercial species 1.0 inches d.b.h. and larger.
[b] Standard error of estimated mean.

Ten years after harvest, the abundance of large reproduction under the residual overstory trees was uniform among the four study areas averaging more than 990 stems per acre. Although not present in large numbers at 5 years, black birch (*Betula lenta* L.) had become a notable competitor in three stands 10 years after treatment (Table 3). For stands on SI 70, black cherry, American beech, red maple, black birch, and northern red oak were the predominant species. On SI 80, yellow-poplar, sugar maple, red maple, and black birch were predominant species. In an earlier study of regeneration resulting from the seed-tree method, stands on site indices 70 and 80 had more than 1,200 and 1,700 stems per acre, respectively, 12 years after harvest (Smith and others 1976). Data from the two-age stands reported here indicate that stand development is similar to that observed for even-age practices where overstory trees are removed during or soon after the regeneration harvest.

Codominant stems present at the time of crown closure have an important impact on species composition and quality of future crop trees developing in the new age class. In general, crown closure among the new age class was nearly complete at 10 years, though the Olson Tower stand had patchy areas where reproduction was lacking due to soil compaction. On SI 70, the Riffle Creek and Olson Tower stands averaged about 700 codominant commercial stems per acre 10 years after two-age harvests were applied (Table 4). More than 70 percent of the codominant stems were classified as good quality with the potential to become crop trees in the future (Figure 2). While the number and quality of codominant stems was similar for the two stands on SI 70, species composition was more variable in the Riffle Creek stand. On SI 80, the Fish Trough and Shavers Fork stands averaged 493 codominant stems per acre with more than 85 percent classified as good quality with the potential to become crop trees in the future. Species composition of good, codominant stems was similar in these two stands. Yellow-poplar, black birch, sugar maple, and cucumber were the predominant species 10 years after harvest.

In general, approximately one-half of the commercial reproduction in a codominant crown position after 10 years originated as seedlings among the four study areas (Table 4). Sprout-origin stems accounted for 62 percent of codominant stems in the Shavers Fork stand where yellow-poplar was most abundant. The Olson Tower stand, dominated by black cherry, had only 37 percent sprout-origin stems.

Table 4.--Summary of codominant commercial stems 10 years after a two-age harvest

Stand	Site index	Codominants	Sprout-origin codominants	Sprouts
		No. stems/acre		Percent
Riffle Creek	70	612	308	50
Olson Tower	70	735	276	37
Fish Trough	80	486	203	42
Shavers Fork	80	458	284	62

Height growth of codominant reproduction in the two-age stands was also similar to that observed in even-aged stands in the study area . Height of reproduction in the two-aged stands was measured in 1994, when new reproduction in the study stands ranged from 11 to 15 years old (Table 5). Total height of reproduction in the Olson Tower stand averaged 29 feet and was based on 11-year-old seedling-origin black cherry. Codominant reproduction in the remaining stands had reached 40 feet tall by age 15. By comparison, seedling-origin yellow-poplar and black cherry stems averaged 30 to 35 feet tall (Lamson and Smith 1989), and sprout origin yellow-poplar, black cherry, red maple, and red oak averaged 37 feet tall (Lamson 1983) in 12-year-old clearcuts on SI 70.

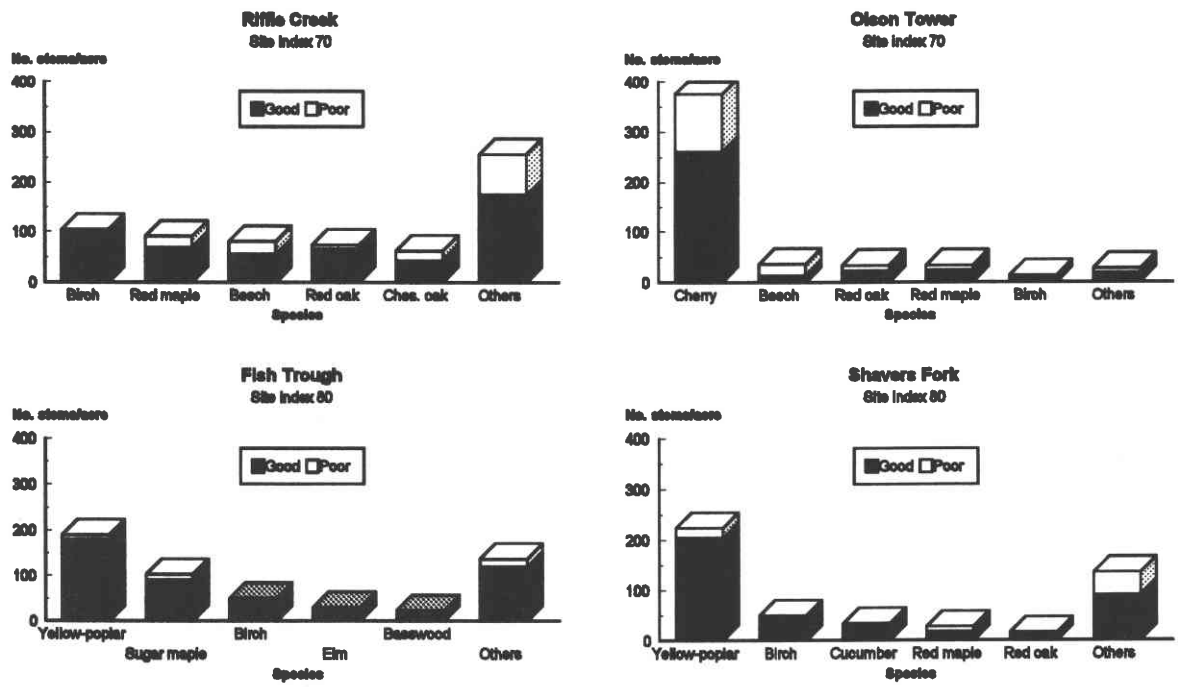

Figure 2. Distribution of codominant reproduction by stand and stem quality 10 years after a two-age harvest .

Table 5.--Average total height of codominant reproduction in two-age central Appalachian hardwood stands

Stand	Site index[a]	Age	Total height
		Years	Feet
Riffle Creek	70	15	42
Olson Tower	70	11	29
Fish Trough	80	14	42
Shavers Fork	80	13	40

[a]Northern red oak site index base age 50 years.

DISCUSSION

Similar to reproduction following clearcutting, the two-age regeneration harvest as applied in this study promoted the development of a variety of commercial hardwood species. For the new age class developing beneath the larger trees, light conditions were suitable for the development of desirable shade-intolerant and shade-tolerant species. In general, yellow-poplar, black cherry, black birch, sugar maple, red maple, and American beech were the most abundant species in a codominant crown position after 10 years. Some good quality, codominant northern red oak stems also were present, though the stands on SI 80 averaged less than 10 stems per acre.

At low residual stocking levels similar to the study stands, reproduction in the new age class can be expected to continue to develop without competition from the larger overstory trees for many years. The residual basal area in 12 to 15 overstory trees/acre ranged from 17.5 to 25.8 ft^2/acre in the four stands reported here. After 10 years, the crowns of residual overstory trees were still widely spaced with approximately 20 feet of growing space between adjacent trees. Crown expansion among the overstory residual trees was negligible after 10 years, and reproduction in the new age class should have ample growing space to allow shade-intolerant species to reach merchantable size classes in the future.

Large reproduction (1.0 inch d.b.h. and larger) also included noncommercial species such as serviceberry (*Amelanchier arborea* **(Michx. f.) Fern.**), striped maple (*Acer pensylvanicum* **L.**), pin cherry (*Prunus pensylvanica* **L. f.**), flowering dogwood (*Cornus florida* **L.**), and American hornbeam (*Carpinus caroliniana* **Walt.**) 10 years after harvest. On SI 70, noncommercial reproduction averaged 250 stems per acre with 160 stems per acre in a codominant crown position. On SI 80 stands, noncommercial reproduction averaged 340 stems per acre, with 96 stems per acre in a codominant crown position. Although noncommercial species can inhibit the growth of neighboring timber crop trees, they do provide food and cover for wildlife and some have showy flowers that enhance the aesthetics of two-age stands in the spring. However, data from this study indicate that an adequate number of timber crop trees can develop in the presence of noncommercial hardwoods.

Similar to even-age practices, light conditions following a two-age regeneration cut enhance the development of wild grapevines on good growing sites. In an earlier study of even-aged stands, grapevines were found growing in 48 to 74 percent of trees 1.0 inch d.b.h. and larger after 12 growing seasons (Smith and others 1976). Although some vines are beneficial to wildlife, stem quality and growth of commercial trees can be reduced when too many vines are present. Vines matted in trees result in crown damage by ice and snow storms in the dormant season, and slow growth because of shading during the growing season. In this study, wild grapevines (*Vitus* **spp.**) were present in all stands 10 years after harvest. In the Shavers Fork stand, 58 percent of commercial trees had at least one vine in the crown. The Riffle Creek and Fish Trough stands each had 25 percent, and the Olson Tower stand had less than 1 percent of commercial trees with vines. Grapevines were cut at ground line after 10 growing seasons to control spreading. This operation required 0.8 to 1.8 man-hours/acre to cut from 70 to 270 vines/acre, respectively.

Residual overstory trees in the four study areas are scheduled for another two-age harvest when the new age class is 80 years old. At that time, the two-age harvest method will be applied again, leaving 12 to 15 trees per acre from the new age class, and cutting all other trees 1.0 inch d.b.h. and larger. To improve the composition, spacing, and quality of the new age class, some potential crop trees in the 10-year-old age class should be released using a crown-touching technique (Lamson and Smith 1989). Such early cultural treatments will improve the value and marketability of two-age harvests in the future.

SUMMARY

The development and quality of natural hardwood reproduction following a two-age regeneration harvest can be summarized by several key observations.

- Leaving 12 to 15 residual overstory trees per acre and cutting all other trees 1.0 inch d.b.h. and larger resulted in hardwood reproduction similar to that expected after clearcutting.
- Two years after harvest, small reproduction (1.0 foot tall to 0.9 inch d.b.h.) averaged 9,100 commercial hardwood stems per acre, composed of 60 percent seedling-origin stems.
- Five years after harvest, large reproduction (1.0 inch d.b.h. and larger) averaged more than 300 commercial hardwood stems per acre, and the canopy had not closed.
- Ten years after harvest, large reproduction averaged 990 commercial hardwood stems per acre, with 574 stems per acre (58 percent) in a codominant crown position.
- Codominant trees averaged 30 to 40 feet tall after 10 growing seasons, and the canopy of the new age class was nearly closed.
- From 70 to 85 percent of codominant reproduction after 10 growing seasons was considered good quality with the potential to become timber crop trees.
- Ten years after harvest, species composition of the new age class was variable, including shade-intolerant species such as yellow-poplar, black cherry, northern red oak, chestnut oak, and black birch, and shade-tolerant species such as American beech, red maple, and sugar maple.
- Large reproduction in a codominant crown position also included serviceberry, striped maple, pin cherry, flowering dogwood, and American hornbeam.
- After 10 growing seasons, residual overstory trees left to form the older age class were still free-to-grow, with an average of 20 feet of growing space between adjacent crowns.
- Reproduction in the new age class is expected to develop for many years without serious competition from the residual overstory trees.

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SHELTERWOOD TREATMENTS FAIL TO ESTABLISH OAK REPRODUCTION ON MESIC FOREST SITES IN WEST VIRGINIA--10-YEAR RESULTS

Thomas M. Schuler and Gary W. Miller[1]

Abstract: The difficulty in regenerating oak on mesic forest sites is well known throughout the eastern and central United States, southern Ontario and Quebec, Canada. Research has shown that the establishment and development of oak seedlings prior to overstory removal, commonly referred to as advanced regeneration, is crucial for retaining oak species in the regenerated stand. The shelterwood reproduction method has been suggested as a means of developing the advance regeneration needed. In 1983, various shelterwood treatments were evaluated on the Fernow Experimental Forest in north-central West Virginia. Three overstory and two understory densities resulting in six treatment combinations were studied. Advanced red oak (*Quercus rubra*) regeneration was not abundant before treatment over most of the study area. Both natural regeneration and planted northern red oak and white ash (*Fraxinus americana*) seedlings were evaluated. Growth of planted seedlings was not significant after 5 years, though survival of red oak was improved significantly by both overstory and understory treatments. Natural regeneration of red oak was inadequate to recommend further overstory removal, and did not differ significantly by treatment combination. Overstory treatments stimulated abundant sweet birch (*Betula lenta*) regeneration, reducing the chances of establishing oak in the future. These results suggest that forest managers in the central Appalachian region may be unable to establish or develop advance regeneration of sufficient size and quantity when attempting to regenerate oaks on mesic sites with the shelterwood method as implemented here.

INTRODUCTION

The difficulty in regenerating northern red oak (*Quercus rubra*) on mesic sites is well known throughout the eastern and central United States and southern Ontario and Quebec, Canada (Smith 1993a; Wagner 1993). Early logging activities that cleared much of the old-growth deciduous forests throughout the range often were heavy but variable, and largely a function of local markets, terrain, and logging technology of the time. Yet, even with the variability of these harvests and a nearly total disregard for the regeneration potential of the stand, the oak component was largely retained in the newly established stands. More recently, retaining oak on these same sites following a regeneration cut has proven most difficult. A continuation of current management practices likely will result in a widespread reduction of oak on mesic sites throughout the region. Fewer oak trees on these sites could have far reaching economic and ecological consequences.

Research has shown that successful regeneration of oak on mesic sites, site index 70 at 50 years or greater, requires the establishment and development of oak seedlings of sufficient size and number, often referred to as advanced regeneration, prior to overstory removal (Carvell and Tryon 1961; Loftis 1990a; Merritt 1979). Variations of the shelterwood regeneration method have been suggested as a means of developing the advanced regeneration needed, and preliminary results where oak seedlings are present have been encouraging (Loftis 1990b; Schlesinger and others 1993).

Oak regeneration can be broken down into three components: establishment, development, and growth after overstory removal. Establishment refers to the germination of acorns and persistence of new seedlings. Development is the

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increase in height and basal diameter before overstory removal. Growth after overstory removal refers to the probability of advance regeneration maintaining a competitive position in the new stand.

The objective of this study was to determine the effect of several shelterwood treatments on the establishment and development of new oak seedlings before final overstory removal. Because oak seedlings were not abundant on most of the study areas, treatments were intended to "establish" new regeneration. The study design allowed us to test the hypothesis that stocking from overstory and understory components is the principal constraint to the establishment and development of oak seedlings. Manipulating overstory and understory density by the use of cutting and herbicides, enabled us to evaluate the effects of several levels of stocking on oak-seedling dynamics.

METHODS

This study was established on the Fernow Experimental Forest (39.03° N, 79.67° W), near Parsons, West Virginia, during the 1983-84 dormant season. Mean annual precipitation on the Fernow is about 58 inches and is distributed evenly throughout the year. Mean annual temperature is 48° F and the length of the frost-free season is approximately 145 days. The elevation of the study sites ranges from about 2,400 to 2,600 feet. The soils within the study sites are characterized by a Calvin channery silt loam (loamy-skeletal, mixed, mesic Typic Dystrochrepts) that developed on uplands in material weathered from sandstone and shale. The Calvin soils are well drained and strongly acid, moderately deep, and moderately permeable. The site index for the study area is about 70 for northern red oak. Overstory species in the study area included northern red oak, chestnut oak (*Q. prinus*), and white oak (*Q. alba*) in descending order of dominance as measured by basal area. Other overstory species included red maple (*Acer rubrum*), sugar maple (*A. saccharum*), sweet birch (*Betula lenta*), and yellow-poplar (*Liriodendron tulipifera*). The study area is part of the Allegheny Mountains Section of the Central Appalachian Broadleaf Forest (McNab and Avers 1994).

The study design was implemented as a 2 x 3 factorial with three overstory treatments combined with two understory treatments for a total of six distinct shelterwood/herbicide treatments with two replications on 12 plots. An additional treatment with a 45-percent residual stocking was omitted from this analysis due to blowdown and salvage logging that occurred in those areas. All data were analyzed by analysis of variance using linear model procedures incorporating the factorial design and tested for significance at the 5-percent level unless otherwise noted. The Tukey-Kramer HSD was used for pairwise multiple comparisons; it is designed to maintain overall protection regardless of the number of comparisons.

The overstory treatments consisted of reducing residual stocking to 75 and 60 percent based on the upland-oak stocking guide (Gingrich 1967) plus an uncut control group (Fig. 1). Average stocking for all plots before treatment was 115 percent. All plots were marked for cutting by designating leave trees. Two guidelines were followed in selecting leave trees: 1) favor oaks for seed source purposes, and 2) achieve uniform spacing of residuals. Species and diameter at breast height (dbh) of each leave tree were tallied and stocking was calculated in the field. Trees less than 5.0 inches in dbh were ignored during marking even though they accounted for as much as 20 percent of stocking in some plots. Thus calculations of residual stocking level included only trees 5.0 inches and larger in dbh. In all plots except for the overstory control plots (no treatment), pole-size trees were either cut and removed, cut and left in place, or treated with herbicides.

The stocking guide for upland oak was selected for control of residual stocking to evaluate its suitability as a thinning guide in the central Appalachian region. The upland oak equations were based on more xeric site oaks such as white, black (*Q. velutina*), and scarlet (*Q. coccinea*), which tend to have smaller crowns relative to northern red oak. Thus, reductions to 75- and 60-percent residual stocking may be more severe than suggested by the residual stocking percentages. Residual stand structures are shown in Figure 2 for 75- and 60-percent residual stocking levels. Of the total number of stems after treatment, 71 and 83 percent were in oak species in the 75-percent and 60-percent residual stocking plots, respectively. Oak stems in both areas averaged 36 percent before treatment.

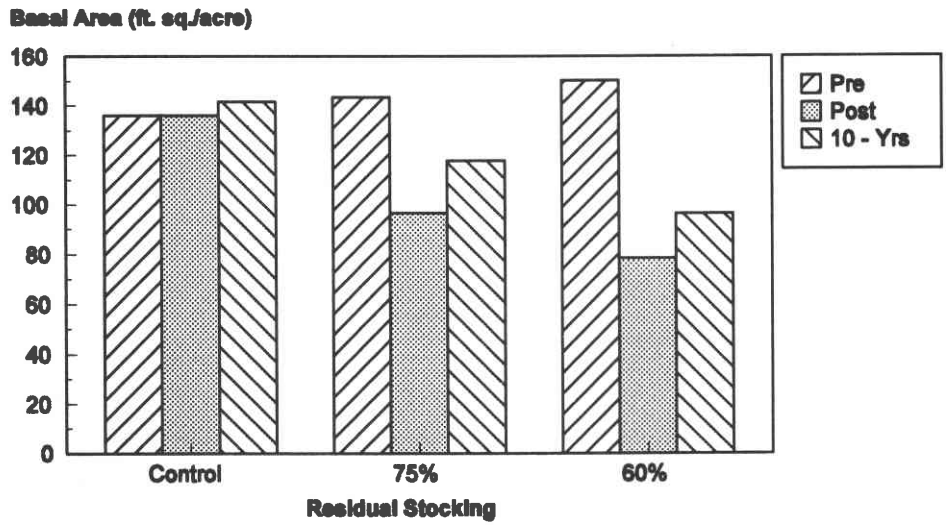

Figure 1. Average basal area per acre of trees 5.0 inches or larger in dbh before treatment (Pre), immediately after treatment (Post), and after 10 growing seasons (10-Yrs).

The understory treatments consisted of basal spraying a 2-percent solution of triclopyr (Garlon® 4) in oil on all stems less than 5.0 inches in dbh, plus an untreated control group. The herbicide treatment was designed to eliminate competition from existing shade-tolerant understory saplings, primarily striped maple and beech. This treatment was applied to two of the four replications for each overstory treatment.

Each treatment area was approximately 3 acres in size with a 0.5-acre growth plot in the center of the treatment area. Within each growth plot, all stems 5.0 inches and larger in dbh were permanently tagged (Lamson and Rosier 1984). Both small and large reproduction data were sampled. Data for small reproduction included the species, height class, and frequency of all woody vegetation observed within 20 0.001-acre sampling points distributed systematically in each plot. Sampling points were permanently marked. Height classes were defined as 0 to 6, 6 to 12, 12 to 36, 36 to 60, and more than 60 inches in total height but less than 0.99 inch in dbh. The results reported here focus on the establishment of seedlings larger than 1 foot tall due to an assumed ephemeral nature of smaller reproduction. Large reproduction was defined as woody species 1.0 to 4.9 inches in dbh observed within 10 0.01-acre sampling points and included all commercial and noncommercial woody species. The same plot centers were used to sample small and large reproduction with every other plot center used for large reproduction. Thus, data on small reproduction were obtained from 240 0.001-acre plots and large reproduction data were obtained from 120 0.01-acre plots.

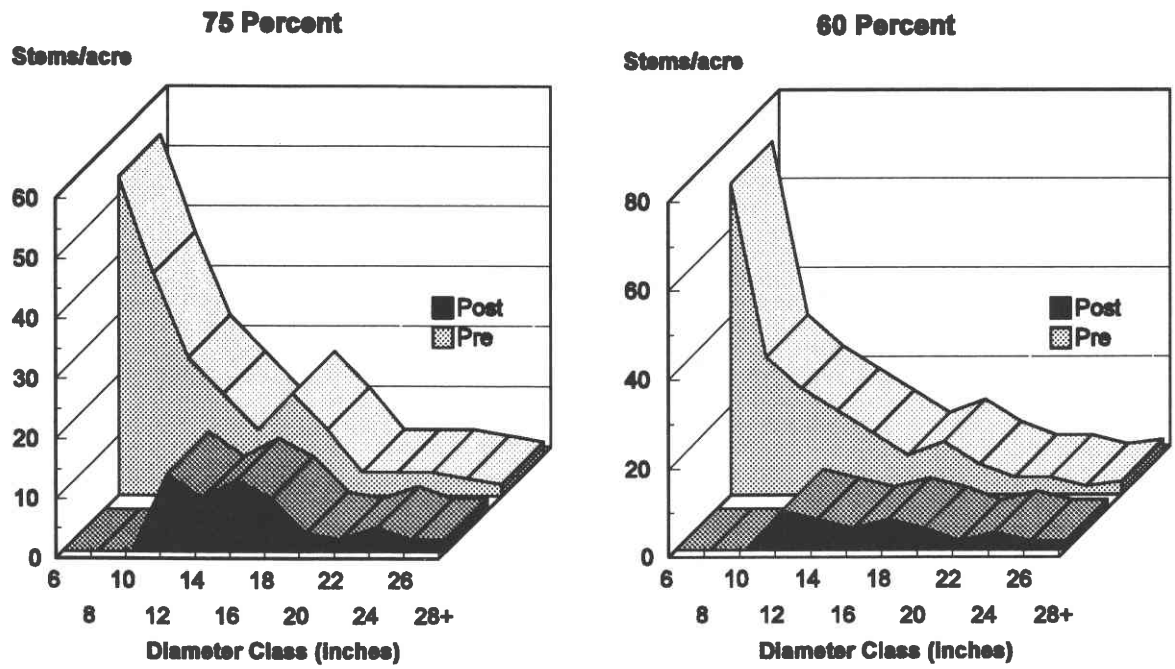

Figure 2. Characteristics of stand structure before treatment (Pre) and immediately after treatment (Post) for 75-percent and 60-percent residual stocking treatments.

All residual overstory trees were remeasured at the same interval as for the understory data. Reproduction plots and 0.5-acre growth plots were surveyed before and after logging prior to the first growing season and again 5, 7, and 10 years later. The most recent remeasurement was in the spring of 1994. During September 1991, near the completion of the eighth growing season, all of the treatment areas that received an initial understory herbicide treatment were mistblown with a 1-percent solution of glyphosate (Roundup®) herbicide. In 1991, understory vegetation ranged from 4 to 15 feet in height. Smaller red oak seedlings that germinated during the prior growing season were abundant when the understory was mistblown. The treatment was applied with backpack sprayers in such a way as to control undesirable vegetation without killing a significant portion of the red oak regeneration. Small-scale trials showed that this was achieved when glyphosate applied at a 45-degree upright angle and care was taken to avoid spraying the small desired vegetation.

In an attempt to ensure some seedling establishment, 50 red oak (2-0) and 50 white ash (*Fraxinus americana*) (1-0) seedlings were underplanted in each treatment area in the spring of 1984 following dormant-season logging. Seedlings were planted 10 feet apart so that the planting locations did not interfere with the reproduction plots. Mortality of planted seedlings was high and all seedlings were replanted before the start of the second growing season.

RESULTS

Natural Seedlings

After 10 growing seasons, the establishment of northern red oak seedlings more than 1 foot tall was low and did not differ for the overstory or understory treatment combination (Table 1). Results from a repeated measures uninvariate analysis of variance combining the results after 5, 7, and 10 growing seasons were similar and, again, demonstrated no significant differences for overstory ($P = 0.219$) or understory ($P = 0.426$) treatments. The repeated measures analysis of variance did suggest some differences in the number of large red oak seedlings within treatments over time ($P = 0.052$). Before treatment, the total number of small red oak seedlings less than 1 foot tall averaged 1,154 per acre and did not differ significantly by treatment area. Thus, some regeneration was present that had the potential to respond to the treatments even in the absence of new seedlings becoming established from acorns. After 5 growing seasons, many of these small oak seedlings were recruited into larger size classes. However, remeasurements after both 7 and 10 growing seasons showed that the total number of oak seedlings more than 1.0 foot tall declined at each measurement period. This decline coincided with increased competition from other species responding to the initial treatment (Fig. 3).

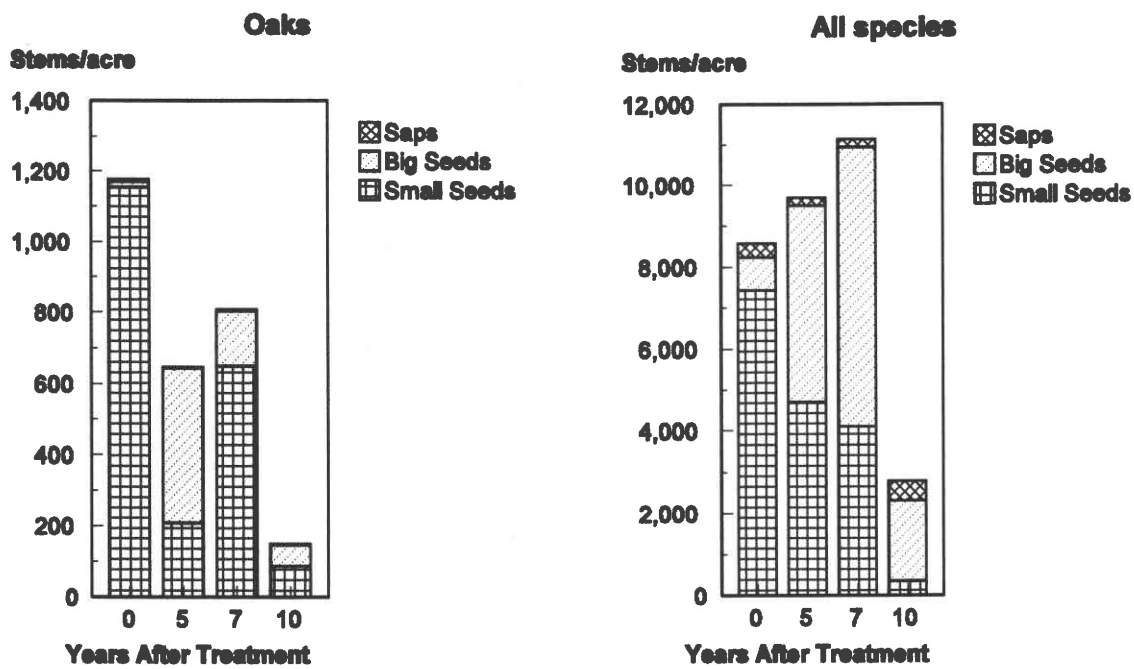

Figure 3. Characteristics of understory structure before the first growing season and 5, 7, and 10 years after treatment for oak and all species combined; "Saps" refers to trees 1.0 to 4.9 inches in dbh; "Big Seeds" includes stems taller than 1 foot but less than 1 inch in dbh; and "Small Seeds" includes stems less than 1 foot tall.

Table 1.--Statistical summary for northern red oak seedlings more than 1.0 foot tall and less than 1 inch in dbh 10 growing seasons after initial shelterwood/herbicide treatment combinations; sources of variation are denoted as OTREAT for overstory treatment and UTREAT for understory treatment

Source	Sum-of-squares	DF	Mean-Square	F-ratio	P
OTREAT	51666.667	2	25833.333	1.192	0.366
UTREAT	20833.333	1	20833.333	0.962	0.365
OTREAT*UTREAT	71666.667	2	35833.333	1.654	0.268

For species other than red oak, the abundance and species composition of small reproduction differed among treatments after 10 years. Perhaps most striking was the response of sweet birch. Analysis of variance indicated that the number of sweet birch seedlings, which are intermediate in shade tolerance (Trimble 1975), were significantly different with respect to overstory ($P = 0.045$) but not to understory treatment ($P = 0.129$). There were no significant interactions between overstory and understory treatments ($P = 0.342$). When overstory treatment alone was the grouping variable, the 60-percent stocking plots were significantly different according to the Tukey-Kramer HSD pairwise comparison (Table 2). Graphically, it appears that the threshold conditions necessary for birch regeneration were reached at 75-percent overstory stocking in combination with understory removal (Fig. 4). At higher levels of residual stocking, including the 75 percent overstory with no understory treatment, birch is a minor component of the understory.

Table 2.--Average number of seedlings per acre (1.0 foot tall to 0.99 inch in dbh) by overstory/understory treatment combination 10 years after treatment (standard error of treatment mean in parentheses)

		Species Group					
Overstory treatment (percent)	Understory treatment	Striped maple	Other tolerant	Birch	Other intermediate	Intolerant	Total
60	Yes	1,750 (200)	350 (300)	1,825 (775)a	1925 (975)	1,475 (425)a	7,325 (125)a
60	No	3,375 (25)	150 (160)	1,025 (225)a	175 (175)	75 (75)b	4,800 (200)a
75	Yes	2,450 (450)	175 (175)	1,075 (425)b	175 (175)	325 (175)b	4,200 (1400)b
75	No	2,750 (600)	100 (0)	50 (0)b	175 (175)	25 (25)b	3,100 (750)b
100	Yes	2,875 (1025)	175 (25)	100 (100)b	100 (100)	25 (25)b	3,275 (975)b
100	No	2,750 (100)	175 (75)	250 (250)b	50 (50)	0 (0)b	3,225 (125)b

Note: Values in columns followed by the same letter are not significantly different at 5-percent level using Tukey-Kramer HSD.

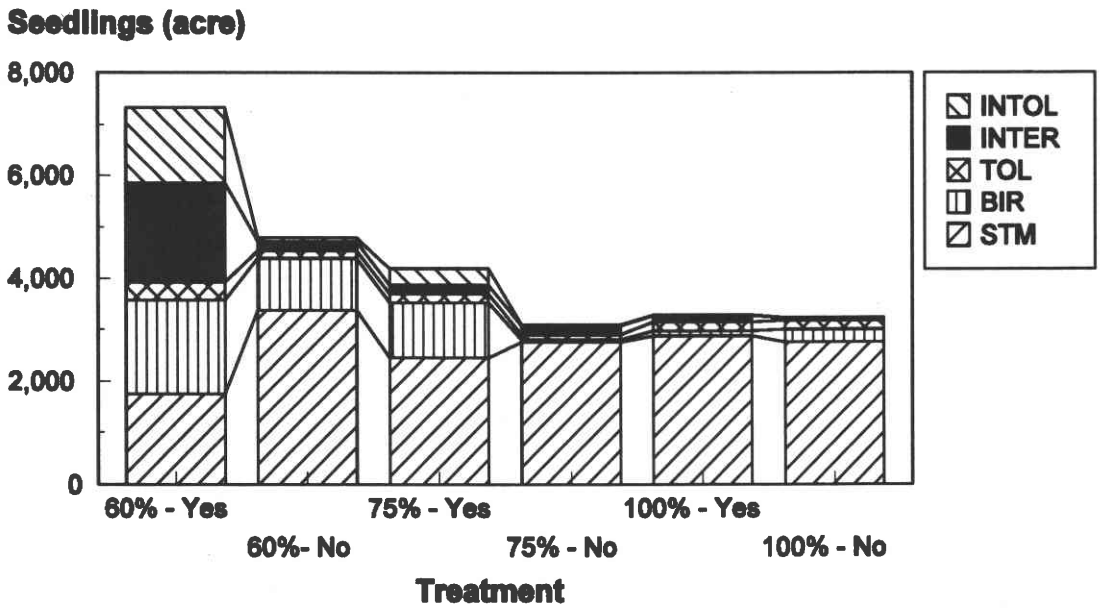

Figure 4. Number of stems per acre (1 foot tall to 0.99 inch in dbh) for each species group 10 years after treatment. "Yes" or "No" indicates understory treatment; species groups are striped maple (STM), sweet birch (BIR), shade-tolerant species excluding striped maple (TOL), intermediate shade-tolerant species excluding birch (INTER), and shade-intolerant species (INTOL).

Other intermediate shade-tolerant species demonstrated no significant differences among either overstory (P = 0.109) or understory treatments (P = 0.130). The small number of replications (n=2) and wide variability prevent a finding of significance, yet it appears that the combination of understory treatment and 60-percent residual overstory stocking promotes a variety of intermediate shade-tolerant species. This category included Fraser magnolia (*Magnolia fraseri*), cucumber (*Magnolia acuminata*), red oak, and white oak in descending order of abundance. The results also illustrate that these species are less shade tolerant than birch.

Intolerant species were significantly different among overstory (P=0.016) and understory (P=0.010) treatments. This was the only response variable that demonstrated significant differences for both types of treatments. However, only at the 60-percent residual stocking level in combination with understory removal were the intolerant species significantly more abundant. Species observed in this category included black cherry (*Prunus serotina*), yellow-poplar, sassafras (*Sassafras albidum*), and black locust (*Robinia pseudoacacia*).

Striped maple (*Acer pensylvanicum*) was isolated from other shade-tolerant species because of its relative abundance and influence on the regeneration of other species. Striped maple was abundant in all treatments, and not significantly related to overstory (P=0.880) or understory (P=0.212) treatments. The understory treatment designed to reduce competition, in part, from this noncommercial species was largely ineffective, possibly because seeds of striped maple ripen in September and October and are dispersed in October and November. The spring-applied triclopyr to existing stems did cause increased mortality of existing stems but did not negatively influence germination of striped maple

from seed. The understory treatment may even have created conditions more suitable for survival of first year seedlings of striped maple whose seeds can remain viable for up to 2 years (Wilson and others 1979). Effective control of striped maple must include both existing stems and viable seed.

Other shade-tolerant species that were largely absent from the regeneration included red maple, sugar maple, American beech (*Fagus grandifolia*), eastern hophornbeam (*Ostrya virginiana*), black gum (*Nyssa sylvatica*), and eastern hemlock (*Tsuga canadensis*). This group did not exhibit differences for the overstory (P = 0.760) or understory treatment (P = 0.485).

The total number of all woody species combined was significantly different with respect to the overstory (P = 0.021) but not understory treatment (P = 0.099). However, from the results noted, the differences in this category are due primarily to differences in the number of intolerant and intermediate shade-tolerant species, including birch.

Analyzing the response to treatments was hindered by the confounding nature of the understory treatment applied at the end of the eighth growing season. This treatment was applied only on plots with an initial understory treatment, so we excluded all of these plots and repeated the analysis for overstory treatment only. Again, neither the response among overstory treatments (P=0.116) nor within treatments (P=0.256) was significant over time in a repeated measures design for large red oak seedlings.

In summary, the response to treatments can be characterized by an initial increase in red oak seedlings more than 1 foot tall at 5 years and a subsequent decline of the same due to an inability to maintain competitive positions with other species beyond 5 years. Greater reductions in basal area evoked greater response in recruitment of smaller oak seedlings into larger size classes, though the effect was ephemeral as the other faster growing species became established and surpassed oak seedling development in total height and competitive position. Beyond 7 years, recruitment of new individual stems greater than 1 foot tall has stopped and mortality within the new cohort began. Unless the small percentage of oak in the understory can become more dominate at a later stage of development, it would seem that the efforts to establish a new cohort of oak have been unsuccessful.

Planted Seedlings

Underplanted seedlings could serve as a source of regeneration if the seedlings respond to treatments. Survival of planted red oak seedlings was significantly different for both overstory (P = 0.004) and understory (P = 0.017) treatments. Survival was inversely related to percent stocking and further enhanced by understory removal. After 4 growing seasons, survival of red oak exceeded 90 percent for both the 60- and 75-percent stocking areas where the understory was removed (Table 3). Height growth of red oak was negligible and total height declined due to browsing by deer, and neither was significantly related to overstory (P = 0.065) or understory (P = 0.063) treatment.

Survival of white ash seedlings was not significantly related to overstory (P = 0.237) or understory (P = 0.438) treatment. Seedling total height was significantly related to overstory (P = 0.005) but not understory (P = 0.111) treatment. Both the 60- and 75-percent residual stocking treatments significantly increased height growth relative to no reduction in the overstory density. The practical significance of this may be minor as height growth averaged less than 1 foot in 5 years (Table 4). Further, survival of both species declined after 7 years, perhaps due to the reemergence of the treated understory. As a consequence, planted seedlings were not protected from the understory treatment that was applied at the end of eight growing seasons so survival rates after this period no longer can be considered valid.

Table 3.--Survival and height growth of planted northern red oak seedlings (2-0) 5 years after treatment (standard error of treatment means in parentheses)

Overstory treatment	Understory treatment	Number of seedlings	Initial height	5-year height	Survival
Percent			Feet	Feet	Percent
60	Yes	90	1.40	1.59 (.01)	90 (10)a
60	No	100	1.33	1.19 (.19)	65 (1)b
75	Yes	100	1.22	1.19 (.18)	91 (1)a
75	No	100	1.50	1.33 (.00)	50 (16)b
100	Yes	100	1.35	1.24 (.03)	32 (14)b
100	No	100	1.23	0.81 (.15)	20 (4)b

Note: Values in columns followed by the same letter are not significantly different at 5-percent level using Tukey-Kramer HSD.

Table 4.--Survival and height growth of planted white ash seedlings (1-0) 5 years after treatment (standard error of treatment means in parentheses)

Overstory treatment	Understory herbicide	Number of seedlings	Initial height	5-Year height	Survival
Percent			Feet	Feet	Percent
60	Yes	110	0.67	1.39 (.17)a	94 (4)
60	No	100	0.70	1.25 (.21)a	78 (6)
75	Yes	100	0.66	1.67 (.10)a	91 (3)
75	No	100	0.73	1.25 (.15)a	83 (1)
100	Yes	100	0.66	0.77 (.12)b	64 (26)
100	No	99	0.67	0.70 (.05)b	55 (29)

Note: Values in columns followed by the same letter are not significantly different at 5-percent level using Tukey-Kramer HSD.

DISCUSSION

The results of this study indicate that residual stocking levels alone may not affect the establishment of oak seedlings on mesic sites in the central Appalachian region. In fact, attempts to establish oak through partial overstory reductions could create conditions that favor the establishment of other species such as sweet birch that can regenerate well in partially shaded understories. Also, in this study, a range of conditions was created that favored the establishment of species from shade tolerant to shade intolerant. That red oak is classified as intermediate in shade tolerance demonstrates that while stocking and its influence on light reaching the forest floor may be a constraint to the establishment of red oak seedlings in some situations, there are other constraints that are even more limiting.

For the southern Appalachians, Loftis (1990b) cautioned that the goal of basal area reductions to regenerate oak on mesic sites should be to develop existing seedlings only. Our results clearly illustrate the basis for this warning. The greater the reduction in basal area, the greater the response by non-oak species to use that growing space. Others have suggested that competing non-oak species must be controlled with fire or herbicides to provide an opportunity for oak species to develop (Schlesinger and others 1993). However, understory treatments in this study were insufficient to retard the development of competing species.

Disturbance patterns in the presettlement forest that created conditions suitable for oak stands on mesic site may have differed greatly from existing patterns. For example, fires of varying intensity may have been much more common. Postfire mortality was highly correlated with tree size, fire severity, and species in Virginia 2 years after wildfire (Regelbrugge and Smith 1994). In the Virginia study, the relative basal area of chestnut oak increased following severe fires, indicating characteristics of species-specific fire resistance. In the Fernow study, birch quickly became established in the understory following reductions in basal area , apparently preventing further recruitment by other species. Yet birch is one of a group of species that are easily damaged by ground fires. Because of its thin bark, even light scorching at the base of a birch tree will lower its resistance to attacks by insects and diseases. And birch is not a prolific sprouter (Lamson 1990). These characteristics are consistent with theories that suggest that periodic ground fires created conditions that favored the development of oak in the understory while discouraging the development of birch and similar species (Crow 1988; Rouse 1986; Reich and others 1990).

Undisturbed stands were once thought to provide optimal conditions for the establishment of oak seedlings (Korstian 1927). In the southern Appalachians, establishment of new seedlings is seldom a management problem. Inventories usually encounter 1,000 or more small red oak seedlings per acre in undisturbed, mature stands, and abundant acorn crops, which usually occur about twice per decade, result in many more oak seedlings becoming established (Loftis 1988). Many of the mature oak stands of today appear undisturbed in the sense that they are fully stocked and no cutting has occurred for several decades. Fire has been controlled and records indicate an absence of fire in many of these stands for 50 to 60 years. Yet, deer population densities have risen to record levels and the impact of deer browsing can be profound (Marquis 1981, Michael 1988).

Results from underplanting seedlings after 5-years suggest little potential for this technique; most discouraging was the lack of height growth. Survival of natural red oak seedlings after overstory removal is a function of the abundance and the size of the advanced regeneration (Loftis 1990a; Sander and others 1976). Since densities of planted seedlings as high as several thousand per acre would be impractical, success with underplanting seem to hinge on the potential size of the individuals before overstory removal. Survival of underplanted red oak seedlings was significantly related to overstory treatment, but height growth was not. In the Ozarks, Johnson and others (1986) found that the survival and growth of underplanted red oak are correlated with initial size of the seedling. Our results may reflect inadequate size characteristics for underplanting red oak in the central Appalachian region.

In the buffer strips of the 60- and 75-percent stocking plots, 20 natural red oak seedlings were sheltered with 5-foot-tall tan Tubex® tree shelters in April 1990. Preliminary results showed that the use of these shelters in shelterwood environments is not encouraging as a method for stimulating the height growth of seedlings. Mortality rates were high and height growth negligible for both areas. Tree shelters reduce solar radiation within the shelter and may not

be useful in shelterwood situations. Planting with tree shelters may have more potential when combined with other forms of management that expose the sheltered seedling to full sunlight (Smith 1993b).

The results presented here illustrate the difficulty that forest managers face when attempting to regenerate oak on mesic sites. Managers need to be aware that oak regeneration on mesic sites in the central Appalachians is not achieved through manipulation of stocking levels alone. A better understanding of the role of fire and other forms of disturbances that resulted in long-term dominance by oak seral communities is needed (Abrams 1992; Abrams and Nowacki 1992) before management guidelines can be recommended for natural oak establishment and development prior to overstory removal. Underplanting, as applied in this study, seems to have little potential when combined with reductions in stocking level. Growth of planted seedlings was not sufficient to improve survival probabilities given the limited number of seedlings that can be planted. Until reliable silvicultural guidelines can be developed for the regeneration of oak on mesic sites, prudence should be used when harvesting these stands. If desired ecological conditions include the presence of oak on these productive sites following harvesting activities, partial retention of the preharvest oak component may be the only reliable method to achieve this objective. An oak presence provides a continuous source of hard mast for wildlife, contributes to species diversity, and provides a suitable seed source for future regeneration. If an adequate oak component develops in the regenerated stand, the retained oak trees could be harvested as in the final cut of a shelterwood regeneration, incorporated into two-age management scenarios, or deferred for an additional rotation. If an oak component is not regenerated, future natural regeneration of oak still is possible and the benefits of oak on mesic sites can be retained in part from existing residual trees.

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INTENSITY OF PRECOMMERCIAL CROP-TREE RELEASE INCREASES DIAMETER AND CROWN GROWTH IN UPLAND HARDWOODS

Jeffrey S. Ward[1]

Abstract: In 1988 seven study areas were established in Connecticut to examine the effects of precommercial crop tree release on bole and crown growth. Each study area has 27 8x8 m plots centered on a northern red oak, black oak, or scarlet oak identified as a potential crop-tree (PCT). The 27 plots at each study area were divided into 3 treatments: no cutting, removal of all stems with crowns within 1 m of the crown of a PCT, and removal of all stems with crowns within 1 m of the crowns of a PCT and 2 other potential crop-trees. After cutting, 7191 stems remained. A consequence of crop-tree release is the partial release of trees surrounding crop-trees. The percentage of release of these surrounding trees was assessed in 25% increments. Diameter growth increased with amount of release and crown class. Relative to unreleased trees, 4-yr diameter growth of northern red oak increased by 86%, black/scarlet oak by 65%, red maple by 56%, and black birch by 52%. Release slowed height growth of dominant and codominant oaks for only the first 2 yr. In sapling stands with few oaks in upper canopy positions, precommercial release could be used to augment oak density. Survival and diameter growth of oaks in the intermediate and suppressed crown classes increased with release intensity. Release also increased height growth of northern red oak in the suppressed crown classes.

INTRODUCTION

Approximately 59% of Connecticut's land area is classified as commercial forest and more than half of the forest is oak-hickory (Dickson and McAfee 1988). Many of these forest stands were established around 1900 and are becoming economically mature. Sawtimber stand area has increased from 17% of commercial forest area in 1952 to 35% in 1972. There has been a concurrent 81% increase in the volume of sawtimber harvested (Dickson and Bowers 1976).

One consequence of the increased harvesting is an increase in the number and area of sapling stands. Unfortunately, many sapling stands are dominated by less valuable species such as red maple and black birch. The shift towards more mesophytic species following final harvest of mature oak forests has been noted throughout the central hardwood region (Lorimer 1989). The change in species composition will affect not only the quality and makeup of forest products available to future generations, but will also affect the quality and variety of wildlife habitats (Scanlon 1992).

Precommercial crop-tree release provides an opportunity to simultaneously manipulate stand composition and concentrate growth on individual stems of high value species. Growth concentrated on high value species by crop-tree release increases the value of the mature stand. Increasing the growth of individual stems shortens the rotation by reducing the time needed to grow stems to a desired size. However, crop-tree release is risky in that the management effort is concentrated on relatively few stems (Trimble 1974) and degradation of bole quality is possible (Holsoe 1947, Lamson and Smith 1978, Sonderman 1985, 1986).

Precommercial thinning generally increases individual stem diameters relative to stems in unthinned stands (Downs 1946, Holsoe 1947, Allen and Marquis 1970, Della-Bianca 1975). Although release usually increases diameter growth of northern red oak (Trimble 1974, Lamson and Smith 1978, Lamson 1983, Pham 1985, Sonderman 1985), some

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studies have reported no diameter growth gain (Trimble 1974, Smith 1977, Lamson 1988). Precommercial release has also increased diameter growth of scarlet oak (Mitchell and others 1985), red maple (Marquis 1969, Trimble 1974, Smith 1977, Lamson 1983, Pham 1985, Lamson 1988) and black birch (Smith and Lamson 1983).

The effect of release on height growth is less clearly understood. Downs (1946) and Sonderman (1986) reported increased total height growth was associated with precommercial thinning. Other studies found height growth was unaffected (Hilt and Dale 1982), or reduced (Holsoe 1947, Allen and Marquis 1970) by thinning. Release generally has been reported to have little effect on height growth of oaks (Trimble 1974, Smith 1977, Sonderman 1985, Mitchell and others 1985), red maple (Trimble 1974, Smith 1977, Lamson 1983, Sonderman 1985), and black birch (Smith and Lamson 1983).

Oak may have a 2-stage height growth response to release. Lamson (1983) reported 5-yr height growth of northern red oak was reduced by release, but that heights of released and unreleased trees were similar after 10 yr (Lamson 1988). Similarly, Trimble (1974) found the 5-yr height growth of released and unreleased northern red oak were similar because released trees grew slower for 2-yr and then faster the next 3-yr.

The objective of this study was to determine how intensity of crown release affected survival, diameter and height growth of hardwood saplings. These results will provide information for forest managers considering precommercial crop-tree release to alter stand composition and concentrate growth on selected trees on the eastern boundary of the central hardwood forest.

METHODS

Study Design

Seven study areas (stand ages 7-22 yr) were established in 1988 in western and central Connecticut (Table 1). Canopy closure was complete on all plots at the beginning of the study. Each study area has 27 8x8 meter plots (1/63 ac). Each plot was centered on a northern red, black, or scarlet oak identified as a potential crop-tree (PCT). Within each plot, location of each stem (> 2 cm dbh) was mapped and stem diameter (at 1.4 m aboveground), crown class (Smith 1962), and top height of live crown were recorded. Diameter height was permanently marked with paint. Diameter measurements were to nearest millimeter. Height measurements to the nearest decimeter were made with 10.5 and 13.5 m telescoping poles. A total of 9429 stems were mapped and measured. Diameter and crown class have been

Table 1. Description of study areas used in precommercial thinning study in Connecticut and median initial size of dominant and codominant northern red/black/scarlet oaks.

Study area	Stand age	Site quality	Number of stems on plots		Median initial size	
			Before cutting	After cutting	Dbh (cm)	Height (m)
Tunxis	7	High	2023	1660	4.4	6.3
Hunter's Mountain	11	Medium	1318	972	4.8	6.9
Overlook	12	Low	1414	1120	4.1	5.9
Blueberry	12	Low	1196	920	3.9	5.4
Woodchopper	15	Medium	1185	856	6.0	7.5
Mott Hill	19	High	1128	795	6.8	8.6
Rockytop	22	Low	1165	868	6.7	7.1

measured yearly on all live trees during the dormant season. Because of time constraints, heights were not remeasured on all plots the same year. Heights on the Tunxis plot remeasured in 1989 and 1991. Heights on all other plots were remeasured in 1990 and 1992.

At each study area, the 27 plots were divided among 3 treatments: no cutting within the plot (control); removal of all stems with crowns within 1 m of the crown of a PCT; and removal of all stems with crowns within 1 m of the crown of three PCTs. All cutting was done after the 1988 growing season. After cutting, 7191 stems remained. It became apparent after cutting that many trees surrounding PCTs on the thinned plots had been partially released (Fig. 1). This provided an opportunity to examine growth response to partial release. Therefore, the percent release of trees surrounding PCTs was assessed in 25% increments. This rating system is similar to the free-to-grow rating system detailed in Lamson and others (1990). Trees on control plots were assigned a release factor of 0%.

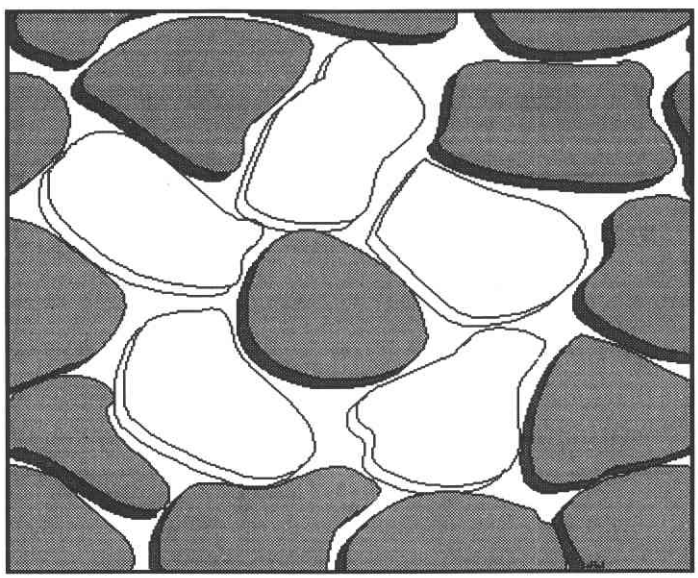

Figure 1. Hypothetical crown map of upper canopy trees with released crop-tree (dark shaded in center), cut trees (unshaded), and surrounding trees.

Data Analysis

Four species groups were selected for analysis: northern red oak (*Quercus rubra*), black/scarlet oak (*Q. velutina, Q. coccinea*), red maple (*Acer rubrum*), and black birch (*Betula lenta*). Number of trees in each combination of crown class and species group are shown in Table 2. Tukey HSD test (SYSTAT 1992) was used to test differences in diameter and height growth among crown classes, release categories, and release categories by crown class. Least square means are presented in tables of combined crown classes and release categories to adjust for larger sample sizes in lower crown classes and lesser release categories. Chi-square statistics were used to determine whether mortality/survival was independent of crown class and release factor. Differences were considered significant at $P \leq 0.05$.

Table 2. Number of saplings in Connecticut by crown class, percent release, and species.

Initial Crown class	Percent release	Northern red oak	Scarlet\ Black oak	Red maple	Black birch
Dominant	0-100%	127	129	185	144
Codominant	0-100%	200	203	516	156
Intermediate	0-100%	238	242	704	164
Suppressed	0-100%	188	150	659	131
Combined classes	0- 24%	440	432	1328	398
Combined classes	25- 49%	128	138	406	109
Combined classes	50- 74%	63	69	232	48
Combined classes	75-100%	122	85	98	40
Dominant	0- 24%	54	69	111	90
Dominant	25- 49%	10	16	24	19
Dominant	50- 74%	9	13	29	19
Dominant	75-100%	54	31	21	16
Codominant	0- 24%	103	115	318	101
Codominant	25- 49%	28	28	93	25
Codominant	50- 74%	21	27	62	11
Codominant	75-100%	48	33	43	19
Intermediate	0- 24%	149	161	462	118
Intermediate	25- 49%	53	55	149	32
Intermediate	50- 74%	19	13	72	12
Intermediate	75-100%	17	13	21	2
Suppressed	0- 24%	134	87	437	89
Suppressed	25- 49%	37	39	140	33
Suppressed	50- 74%	14	16	69	6
Suppressed	75-100%	3	8	13	3

RESULTS

Diameter growth

Median diameter of dominant and codominant northern red, black, and scarlet oaks at the beginning of the study ranged from 3.9 cm (1.5 in) on the Blueberry plot to 6.8 cm (2.7 in) on the Mott Hill plot (Table 1). Diameter growth was not independent of either the degree of release or crown class (Table 3). Releasing trees more than 75% (complete release) nearly doubled diameter growth compared with unreleased trees. Diameter growth of dominant trees was more than 5 times greater than for suppressed trees. All crown classes responded to release. Within each crown class, 4-yr diameter growth of completely released trees was approximately 1 cm (0.4 in) more than for unreleased trees.

Table 3. 4-yr diameter growth (cm) of saplings in Connecticut by crown class, percent release, and species.

Initial Crown class	Percent release	Northern red oak		Scarlet\ Black oak		Red maple		Black birch	
Dominant	0-100%	3.08	a[a]	3.38	a	1.81	a	2.88	a
Codominant	0-100%	2.16	b	2.14	b	1.34	b	1.88	b
Intermediate	0-100%	1.33	c	1.25	c	.74	c	1.13	c
Suppressed	0-100%	.83	d	.63	d	.41	d	.65	c
Combined classes	0- 24%	1.37	a	1.42	a	.81	a	1.27	a
Combined classes	25- 49%	1.67	b	1.58	a	.96	b	1.61	b
Combined classes	50- 74%	1.80	b	2.05	b	1.25	c	1.74	b
Combined classes	75-100%	2.56	c	2.34	b	1.26	c	1.92	b
Dominant	0- 24%	2.50	a	2.75	a	1.59	a	2.45	a
Dominant	25- 49%	3.12	ab	3.33	ab	1.57	a	2.96	a
Dominant	50- 74%	-[b]		3.75	b	2.14	b	3.11	a
Dominant	75-100%	3.79	b	3.69	b	1.94	ab	3.00	a
Codominant	0- 24%	1.77	a	1.64	a	.96	a	1.63	a
Codominant	25- 49%	1.84	a	1.82	ab	1.26	b	1.81	ab
Codominant	50- 74%	2.12	a	2.22	bc	1.62	c	1.77	ab
Codominant	75-100%	2.90	b	2.87	c	1.50	bc	2.29	b
Intermediate	0- 24%	.85	a	.92	a	.49	a	.71	a
Intermediate	25- 49%	1.16	b	.71	a	.64	b	1.08	b
Intermediate	50- 74%	1.25	b	1.32	ab	.88	b	1.19	b
Intermediate	75-100%	2.07	c	2.06	b	.94	b	-	
Suppressed	0- 24%	.36	a	.38	a	.19	a	.27	a
Suppressed	25- 49%	.56	b	.48	a	.39	b	.61	b
Suppressed	50- 74%	.91	c	.92	b	.38	b	-	
Suppressed	75-100%	-		-		.68	b	-	

[a] Values within a column followed by the same letter were not significantly different at $P \leq 0.05$. Tested using Tukey HSD test.
[b] Less than 10 trees.

Diameter growth of all species groups was increased by release. Release had a greater effect on the diameter growth of oaks than on red maple and black birch (Table 3). Relative to unreleased trees, 4-yr diameter growth of completely released northern red oaks was increased 1.2 cm, black/scarlet oak by 0.9 cm, black birch by 0.7 cm, and red maple by 0.5 cm. Diameter growth within a crown class for each species group also increased with thinning intensity (Table 3).

Diameter growth of released dominant and codominant trees has been greater than unreleased trees through the first 4 years (Fig. 2). Decreased growth during the 1990 and 1991 growing seasons was probably related to extended mid-summer droughts. Annual diameter growth of northern red oak, black oak, and scarlet were similar and greater than for black birch and red maple.

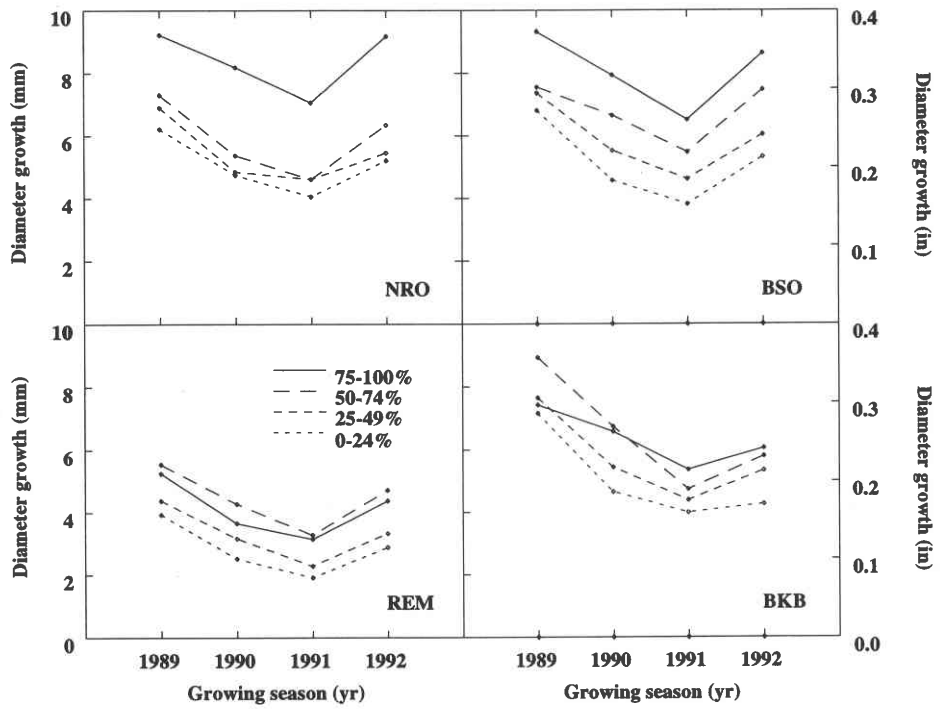

Figure 2. Mean annual diameter growth (mm) of upper canopy trees (dominant and codominant) by year, percent release (0-24%, 25-49%, 50-74%, 75-100%), and species group (NRO-northern red oak, BSO-black/scarlet oak, REM-red maple, BKB-black birch).

Height growth

Median initial heights ranged from 5.4 m (18 ft) on the Blueberry plot to 8.6 m (28 ft) on the Mott Hill plot (Table 1). Median height on most plots was less than the 7.6 m (25 ft) recommended by (Perkey and Wilkins 1994). Height growth was independent of the degree of release, but not crown class (Table 4). Four year height growth of dominant trees, 1.8 m (5.9 ft), was more than 3 times that of suppressed trees, 0.5 m (1.6 ft). There was an interaction between the degree of release and crown class for height growth. Release slowed height growth of taller trees and increased height growth of smaller trees. Height growth of unreleased dominant and codominant trees was greater than for completely released trees. In contrast, height growth of unreleased suppressed trees was significantly less than for trees released 50-75% when, because of small sample size, completely released trees were removed from the analysis.

Release resulted in significant height growth depression codominant black/scarlet oak and red maple (Table 4). Height growth depression of dominant northern red oak by release was not significant ($P \leq 0.058$). Although height growth of suppressed northern red oak and red maple saplings which were completely released was greater than for unreleased trees, the increase was only significant for northern red oak.

Height growth depression on codominant and dominant oaks by release was short-term (Figure 3). ANOVA of height growth between 1988-1990 revealed significant differences among release intensities ($F = 5.56$, d.f. = 3, $P \leq 0.001$). Mean 2-yr height growths between 1988-1992 ranged from 1.0 m for unreleased trees to 0.8 m for completely released trees. There was no significant difference of height growth between 1990-1992 among release intensities ($F = 0.30$, d.f. = 3, $P \leq 0.829$).

Table 4. 4-yr height growth (m) of saplings in Connecticut by crown class, percent release, and species.

Initial Crown class	Percent release	Northern red oak		Scarlet\ Black oak		Red maple		Black birch	
Dominant	0-100%	2.00	a[a]	2.30	a	1.41	a	1.80	a
Codominant	0-100%	1.80	a	1.85	b	1.31	a	1.74	a
Intermediate	0-100%	1.31	b	1.38	c	.88	b	1.44	a
Suppressed	0-100%	.71	c	.52	d	.50	c	.64	b
Combined classes	0- 24%	1.38	a	1.54	a	1.12	a	1.28	a
Combined classes	25- 49%	1.44	a	1.44	a	1.05	a	1.35	ab
Combined classes	50- 74%	1.49	a	1.60	a	.99	a	1.70	b
Combined classes	75-100%	1.51	a	1.47	a	.95	a	1.29	ab
Dominant	0- 24%	2.01	a	2.30	a	1.60	a	1.78	a
Dominant	25- 49%	2.26	a	2.38	a	1.51	a	1.98	a
Dominant	50- 74%	-[b]		2.31	a	1.31	a	2.01	a
Dominant	75-100%	1.70	a	2.21	a	1.22	a	1.42	a
Codominant	0- 24%	1.97	a	2.03	a	1.45	a	1.83	a
Codominant	25- 49%	1.72	a	1.81	ab	1.39	ab	1.62	a
Codominant	50- 74%	1.71	a	1.92	ab	1.30	ab	1.89	a
Codominant	75-100%	1.79	a	1.66	b	1.11	c	1.62	a
Intermediate	0- 24%	1.23	a	1.36	a	1.02	a	1.07	a
Intermediate	25- 49%	1.38	a	1.23	a	.84	a	1.43	a
Intermediate	50- 74%	1.22	a	1.60	a	.84	a	1.65	a
Intermediate	75-100%	1.40	a	1.31	a	.83	a	-	
Suppressed	0- 24%	.29	a	.49	a	.41	a	.44	a
Suppressed	25- 49%	.41	ab	.32	a	.46	a	.38	a
Suppressed	50- 74%	1.00	b	.59	a	.50	a	-	
Suppressed	75-100%	-		-		.63	a	-	

[a] Values within a column followed by the same letter were not significantly different at $P \leq 0.05$. Tested using Tukey HSD test.
[b] Less than 10 trees.

Mortality

Release reduced 4-yr mortality rates for smaller, but not larger trees (Table 5). Mortality rates were significantly different among release levels for suppressed ($\chi^2 = 47.60$, d.f. = 3, $P \leq 0.001$) and intermediate trees ($\chi^2 = 8.15$, d.f. = 3, $P \leq 0.043$). Mortality rates for unreleased suppressed and intermediate trees was more than 3 times higher than for completely released trees. Releasing suppressed trees decreased mortality for all species groups (Fig. 4). Mortality rates of codominant and dominant trees were independent of release intensity (Table 5).

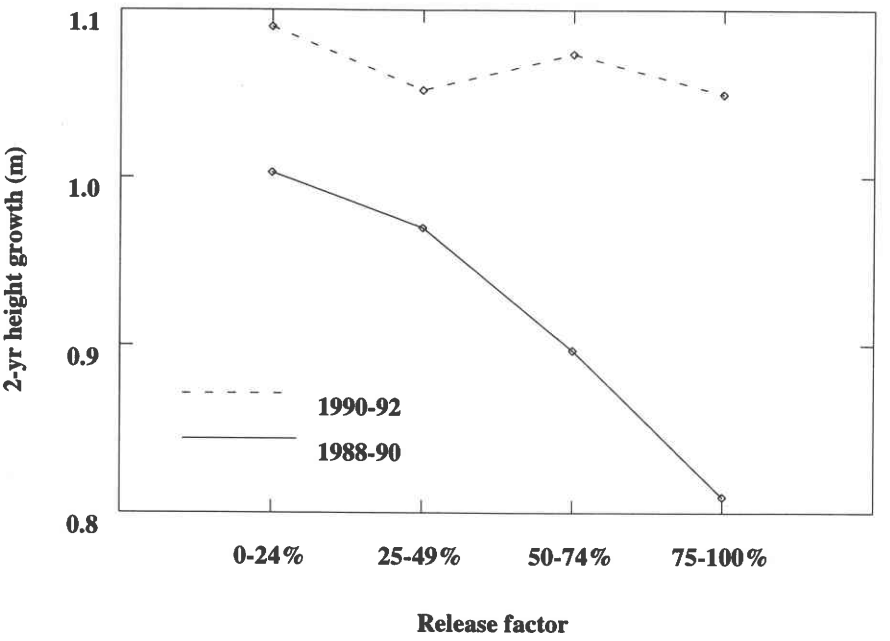

Figure 3. 2-yr height growth (m) of upper canopy oaks between 1988-1990 and 1990-1992 by percent release.

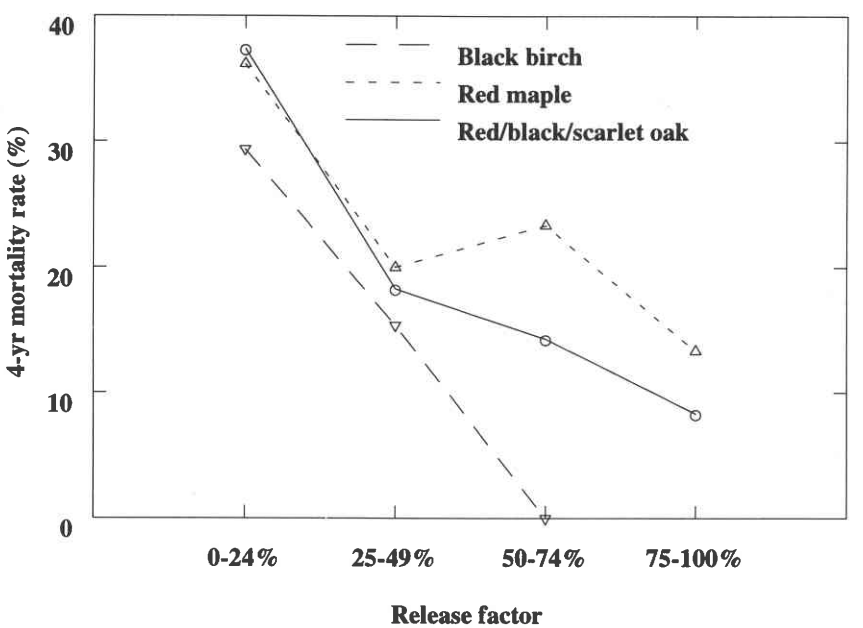

Figure 4. 4-yr mortality rates of suppressed trees by percent release and species group.

Table 5. 4-yr mortality rates (%) of saplings in Connecticut by crown class and release intensity.

	Crown class in 1988			
Percent release	Suppressed	Intermediate	Codominant	Dominant
0- 24%	35.8	6.2	1.9	1.8
25- 49%	18.9	2.4	1.7	1.4
50- 74%	20.0	4.9	0.8	0.0
75- 100%	10.0	1.9	2.1	2.4

DISCUSSION

Precommercial thinning increased diameter growth of both dominant and codominant trees. Both the diameter growth and the increase in growth with degree of release are similar to results in West Virginia (Lamson and others 1990). Not surprisingly, diameter growth response to complete release was greater for codominant trees, 80% increase, than for dominant trees, 47% increase. The diameter growth response of oaks was better than for red maple and black birch (Table 3). Released trees have continued to grow faster than unreleased trees for 4-yr after thinning. An earlier study suggests diameters of released trees should continue to grow faster than unreleased trees for 6 yr (Lamson 1988).

Unfortunately, release also decreased height growth of dominant and codominant oaks (Fig. 3). However, the depression of oak height growth was short-term and height growth 2-4 yr after release was independent of release. Trimble (1974) reported height growth of northern red oak was slower for 2 yr after release and then faster for next 3 yr. Because diameter growth of released upper canopy oak saplings continues to be greater than for unreleased trees, the short-term decreased height growth appears to be a transient and inconsequential phenomenon.

Precommercial crop-tree release dramatically and significantly increased diameter growth of saplings in intermediate and suppressed crown classes (Table 3) without negatively impacting height growth (Table 4). Releasing suppressed oaks was especially effective. Relative to unreleased trees, diameter growth of completely released intermediate trees increased by 145% and suppressed trees by 226% (Table 3). This study found that young suppressed oaks can response positively to release, unlike older suppressed oaks (Sander 1977). Reducing stocking to 10% doubled diameter growth of suppressed and intermediate oaks and yellow-poplar (Allen and Marquis 1970). Diameter growth of intermediate northern red oaks was increased by a cutting all trees within 5-ft of each crop-tree (Trimble 1974). However, partial release often does not increase diameter growth of intermediate oak saplings (Smith 1977, Lamson and Smith 1978). While this study did not find that release increased height of intermediate northern red oak as reported by Lamson and Smith (1978), release did not decrease height growth of intermediate oaks. Height growth of suppressed northern red oak was increased by release (Table 4).

There is little doubt that an immediate benefit of precommercial thinning is reduced mortality rates in the suppressed and intermediate crown classes (Fig. 3). Reduced mortality following precommercial thinning has been noted for northern red oak (Trimble 1974, Della-Bianca 1983, Pham 1985), scarlet oak (Mitchell and others 1985), and red maple (Trimble 1974, Pham 1985).

These results indicate that not only can precommercial release be used to increase growth of larger trees (Trimble 1974, Smith 1977, Lamson 1983,1988), but also intermediate and suppressed trees (Allen and Marquis 1970). The ability of suppressed oak saplings to respond to release provides forest managers with a technique to increase oak densities in sapling stands with few oak in upper canopy positions. It is likely that suppressed and intermediate trees will need to be released at least once more to assure their continued presence in the upper canopy. While precommercial crop-tree release is difficult to justify on an economic basis (Dwyer and others 1993), the practice may

be profitable for valuable species on high quality sites (Miller 1986). Increasing oak density also increases wildlife and esthetic values.

ACKNOWLEDGMENTS

I would like to thank the Connecticut Department of Environmental Protection for their help in selecting and protecting study areas; and William Cullina, William Holbrook, Mike Short, Tony LaRussa, David Weinstein, and Darryl Nicholson for assisting in plot establishment, maintenance, and data collection. This research was partly funded by McIntire-Stennis Project No. CONH-537.

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INDIVIDUAL TREE- VERSUS STAND-LEVEL APPROACHES TO THINNING: IS IT A CHOICE OF ONE OR THE OTHER, OR A COMBINATION OF BOTH?

Christopher A. Nowak[1]

Abstract: Thinning guidelines have existed for most eastern hardwood forests for 20 to 30 years. While these guidelines are presented in varying degrees of detail, they generally all contain recommendations on levels of residual stand density and stand structure, along with information on crop tree requirements. Recent attempts have been made to simplify thinning guidelines by using an individual tree approach, referred to as "crop tree management". This is in contrast to the "area-wide approach", which purportedly focuses on the stand-level alone. In this paper, I will show how the two approaches can be used together to improve thinning practice. In such a combined approach, the main focus of thinning remains on the crop tree, and the long standing tools of stand density and stand structure are used to direct stand development along desired paths.

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FOREST HEALTH IN WEST VIRGINIA: PAST, PRESENT AND FUTURE

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Abstract: This report chronicles the status of forest health in West Virginia as of 1993. Primary data sources are the Forest Inventory and Analysis reports, West Virginia state forest pest reports, National Oceanographic and Atmospheric Administration weather records and numerous other publications. We attempted to describe primary stressing agents affecting the forest, including drought, defoliation, fire and overstocking and to relate the occurrence of stress to the performance of the forest (growth, mortality, etc.). Furthermore, we divided the state into zones representing the major cover types (mesophytic hardwoods, Appalachian oaks, northern hardwoods and spruce). Indexes to the health status of these types, such as growth and mortality, were examined, and comparisons were made between high and low stress zones, using the expected rates from published reports.

Forests in West Virginia are generally healthy, although certain areas have been subjected to fairly heavy stress loads over the past 10 years. Forest fires have severely impacted the southern West Virginia counties of Mingo, Logan and Boone. Drought has been severe in several areas of the state, including part of north central West Virginia and the Ridge and Valley Province of the Eastern Panhandle, extending into Pendleton County. Counties experiencing high stress loads showed growth rates about one third lower than areas experiencing lower stress. Appalachian oak forests seem most vulnerable to future stresses, since oaks are highly susceptible to defoliators and occur in drought-prone areas. oaks are also particularly vulnerable to decay, and numerous studies report that there is a lack of adequate oak regeneration in our present-day stands.

The area in spruce forests in West Virginia has decreased steadily over the past few hundred years, and the current rate of mortality in mature spruce exceeds that of other forest types in the state. Whether this is due to the relict nature of spruce, to natural processes, such as cohort senescence, or to anthropogenic influences, such as air pollution or global warming, cannot be determined. The encouraging news is that, in many areas, natural spruce regeneration is apparently healthy and thriving.

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SPATIAL TRENDS IN RELATIVE STOCKING POINT TO POTENTIAL PROBLEMS IN FOREST HEALTH

David A. Gansner, Susan L. King, Stanford L. Arner, Richard H. Widmann and David A. Drake[1]

Abstract: The term "forest health" means many things to many people and we do not know how to measure it. Baseline standards for conducting a physical examination of a stand of trees do not exist. One factor that can be considered when making judgments about the health of a particular forest tree species is change in the relative stocking of that species, that is the extent to which the species is gaining or losing ground in its ecosystem. The forest survey unit at the Northeastern Forest Experiment Station is using remeasured forest inventory plot data to estimate current average annual change in the relative stocking of common forest tree species in the Northeast. Spatial shifts in the relative stocking of individual species are being mapped. The procedure can be readily extended to other species in other regions. Information on shifts in relative stocking can provide a symptomatic guide to recognizing problems of forest health, and it gives us a better understanding of the complex workings of a dynamic ecosystem.

INTRODUCTION

The health of our Nation's forests has become a major issue. Unfortunately, the term "forest health" is vague. It means many things to many people and we do not know how to measure it. Baseline standards for conducting a physical examination of a stand of trees do not exist. The Northeastern Forest Experiment Station's Forest Health Monitoring Program and other organizations are working on how to assess the dimensions of forest ecosystem health and how to analyze and report health trends.

Until clearly defined baseline standards are developed, one factor that can be considered when making judgments about the health of a particular forest tree species is change in the relative stocking of that species, that is, the extent to which the species is gaining or losing ground in its ecosystem. This we can measure with the help of forest inventory plot records.

STOCKING AS A MEASURE OF SITE OCCUPANCY

Stocking is another term that means different things to different people. In this application, it is a measure of the extent to which trees utilize a plot of forest land. Stocking is expressed as a function of the number, size, and basal area of trees. Formulae for calculating stocking levels have been developed for a number of individual species. For example, the equation we use for sugar maple is : $s = .00694(DBH)^{1.86}$, which was obtained by translating the tree-area ratio developed by Stout and Nyland (1986) to a power function of diameter. Solving this equation for one tree on an acre of forest land tells us how much that tree contributes to stocking on the acre. When a stocking formula is not available for a particular species, the formula for a species with similar characteristics of growth and competitiveness is used.

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ESTIMATING CHANGE IN RELATIVE STOCKING

Stocking equations can be applied to remeasured forest inventory plot data to estimate the average annual change in relative stocking (RS) for any tree species on any acre. The following example demonstrates the process.

	Stocking (%)	
	1978	1989
Sugar maple	16	18
All species	64	90

The relative stocking of sugar maple on the acre for 1978 was 16/64 = 25%. The relative stocking of sugar maple for 1989 was 18/90 = 20%. The average annual change in relative stocking for sugar maple on the acre is:

[RS(1989) - RS(1978)]/Years between inventories
or
(20-25)/11 = - 0.45% per year

Note, that even though the absolute stocking of sugar maple increased on this acre, the change in its relative stocking was negative. In other words, in a relative sense, sugar maple was losing ground to other species between inventories.

The USDA Forest Service updates timber-resource information statewide approximately every 10 years. For land that remains in forest, remeasured plot records provide a history of change in the inventory of all live trees 5 inches and larger in d.b.h. By design, each inventory plot represents a proportional share of the forest area in a state; so, appropriate weights can be assigned to plot data to derive statewide and regional averages of current annual change in relative stocking for individual species. We applied this procedure to Kentucky, Maryland, Ohio, Pennsylvania, and West Virginia where a network of approximately 6,000 permanent inventory plots were remeasured during the most recent inventories in each state (Alerich 1990, Frieswyk and DiGiovanni 1988, Griffith et al. 1993, Alerich 1993, and DiGiovani 1990).

OVERALL STOCKING IS UP

At the time of the most recent forest inventories, the stocking of all live trees 5.0 inches d.b.h. and larger per acre of forest land averaged about 59 percent in the five-state area (Table 1). At the time of previous inventories, stocking averaged only 51 percent. Obviously, growth on original trees plus ingrowth of new trees into the 5.0-inch size class more than offset losses to cutting and mortality between inventories. Stocking has increased an average of at least one percent per year in each of the five states.

Table 1. Change in stocking between inventories, by state.

State	Average Stocking		Change(%)
	Previous	Most Recent	
	----------Percent----------		
Kentucky	44.4	52.2	7.8 (18)
Maryland	54.2	59.5	5.3 (10)
Ohio	43.0	52.3	9.3 (21)
Pennsylvania	57.3	63.4	6.1 (11)
West Virginia	54.8	65.3	10.5 (19)
All States	51.3	59.3	8.0 (15)

All but a few forest inventory subunits of the five-state area recorded healthy increases in average stocking (Figure 1). One of the exceptions was south-central Pennsylvania where gypsy moth defoliation, drought, cutting, bark beetles, root rot, deer browsing, and other agents took a heavy toll on the oak resource during the 1980's. Growth of residual oaks, maple, black gum, yellow-poplar, cherry, and other species counter balanced the oak loss. But, gains in average stocking for all species combined remained minimal. On the lower Eastern Shore of Maryland, declines in the relative stocking of oak were offset by increases in loblolly pine, red maple, black gum, beech, and other hardwoods. Thus, average stocking there remained about the same.

MAPLES AND YELLOW-POPLAR GAIN WHILE BLACK LOCUST AND OAKS LOSE GROUND

Red maple is a pioneer species--a shade tolerant, prolific seeder and sprouter that can occupy a wide variety of forest sites. So, it is no wonder that red maple has recorded significant gains in relative stocking throughout the five-state area (Figures 2-7). Overall, the annual gain in relative stocking of red maple averaged 0.31 percent between inventories.

There has been much recent concern over the demise of sugar maple. Not too long ago (12/7/86) an article in the New York Times suggested that sugar maple was becoming extinct because of damage from acid rain. Such news prompted establishment of The North American Maple Project and the installation of plots to measure annual trends in the condition of this valuable hardwood species. It is too soon to draw conclusions from that study. But, our analysis indicates that, on the whole, sugar maple is doing quite well. In terms of gaining ground, it ranks right behind red maple. Substantial increases in the relative stocking of sugar maple were recorded in all five states.

Another prevalent species that made significant regional gains in relative stocking was yellow-poplar. White pine also recorded significant increases in states where it is common.

At the opposite end of the scale were black locust and the oaks (particularly chestnut, black, white, and scarlet oak). Black locust lost ground at a rate of 0.30 percent per year across the region. And why not? Forest land in this area has undergone significant increases in average tree size and density in recent years. Black locust is sensitive to competition and intolerant of shade. Under stress, it falls easy prey to locust borers, leaf miners, heart rot, and other insects and diseases.

Much of the oak decrease can be associated with gypsy moth. But oaks are also losing ground in areas where the pest is not yet a problem. Sharp declines in the relative stocking of hard pines are noteworthy. Virginia pine in Maryland, Ohio, and West Virginia; pitch pine in West Virginia and Kentucky; and shortleaf pine in Kentucky are all losing ground at significant rates (Huntley 1990).

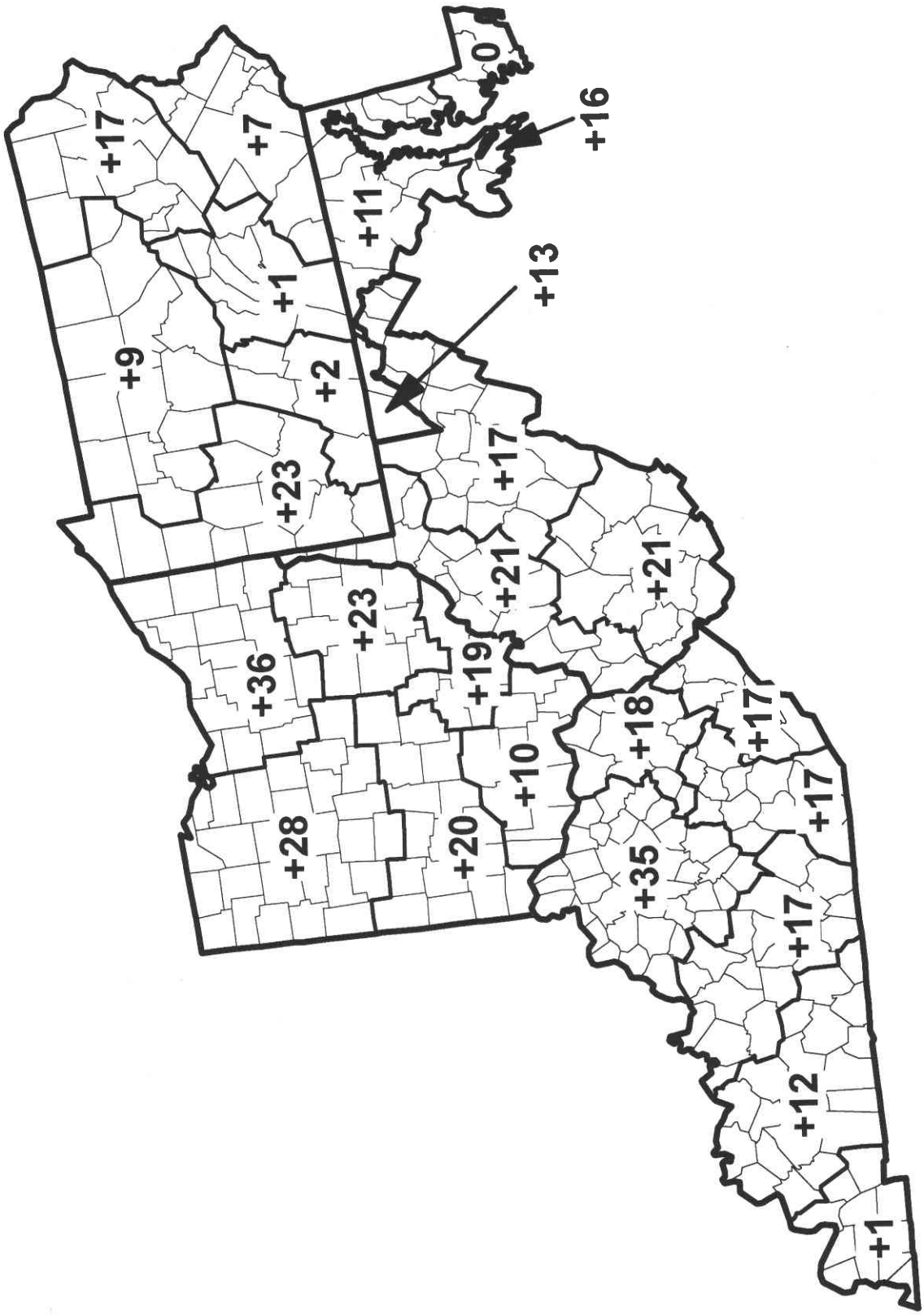

Figure 1. Percentage of change in stocking between inventories, by subunit.

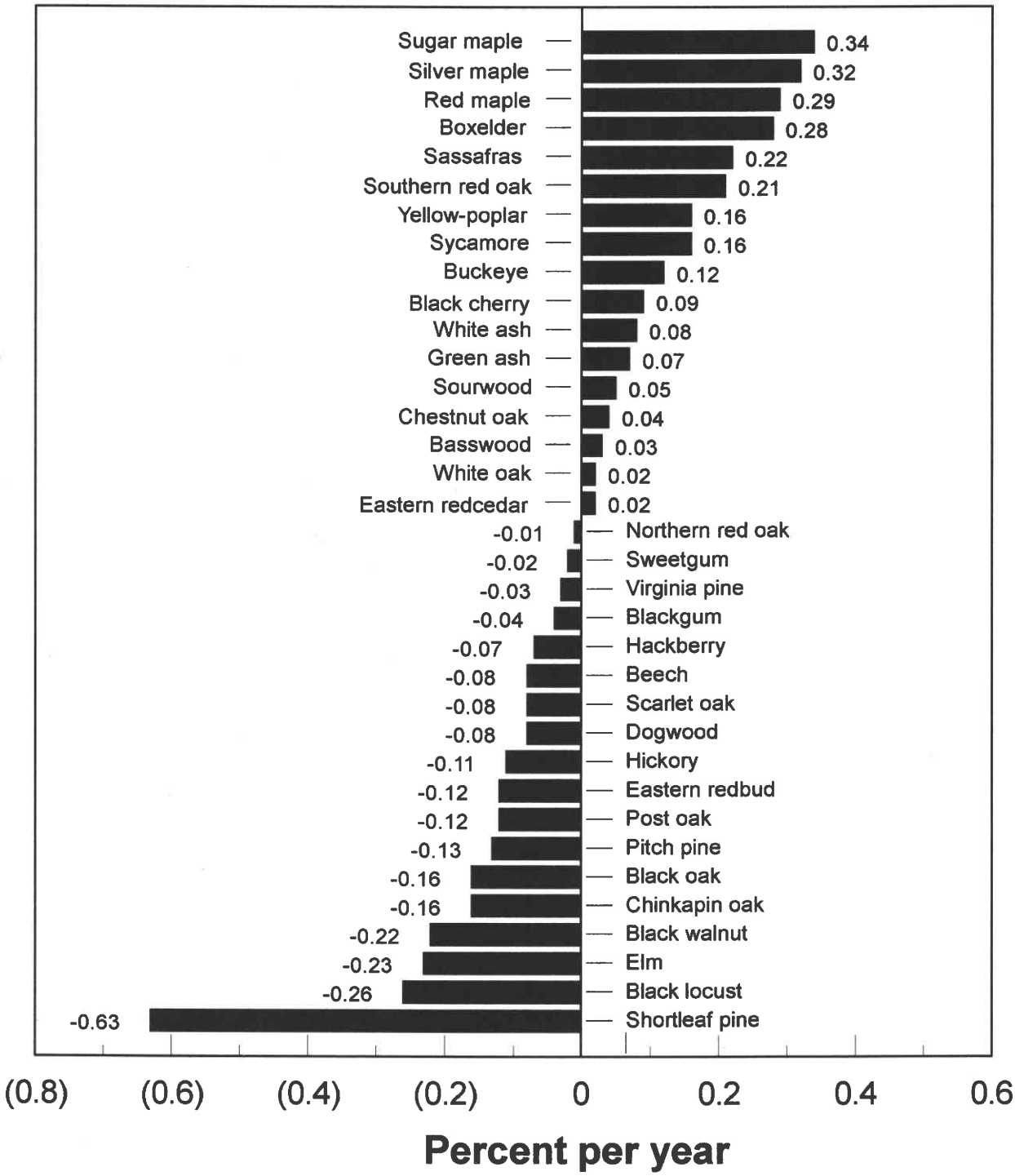

Figure 2. Average annual change in the relative stocking of some common forest tree species in Kentucky, 1975-1988.

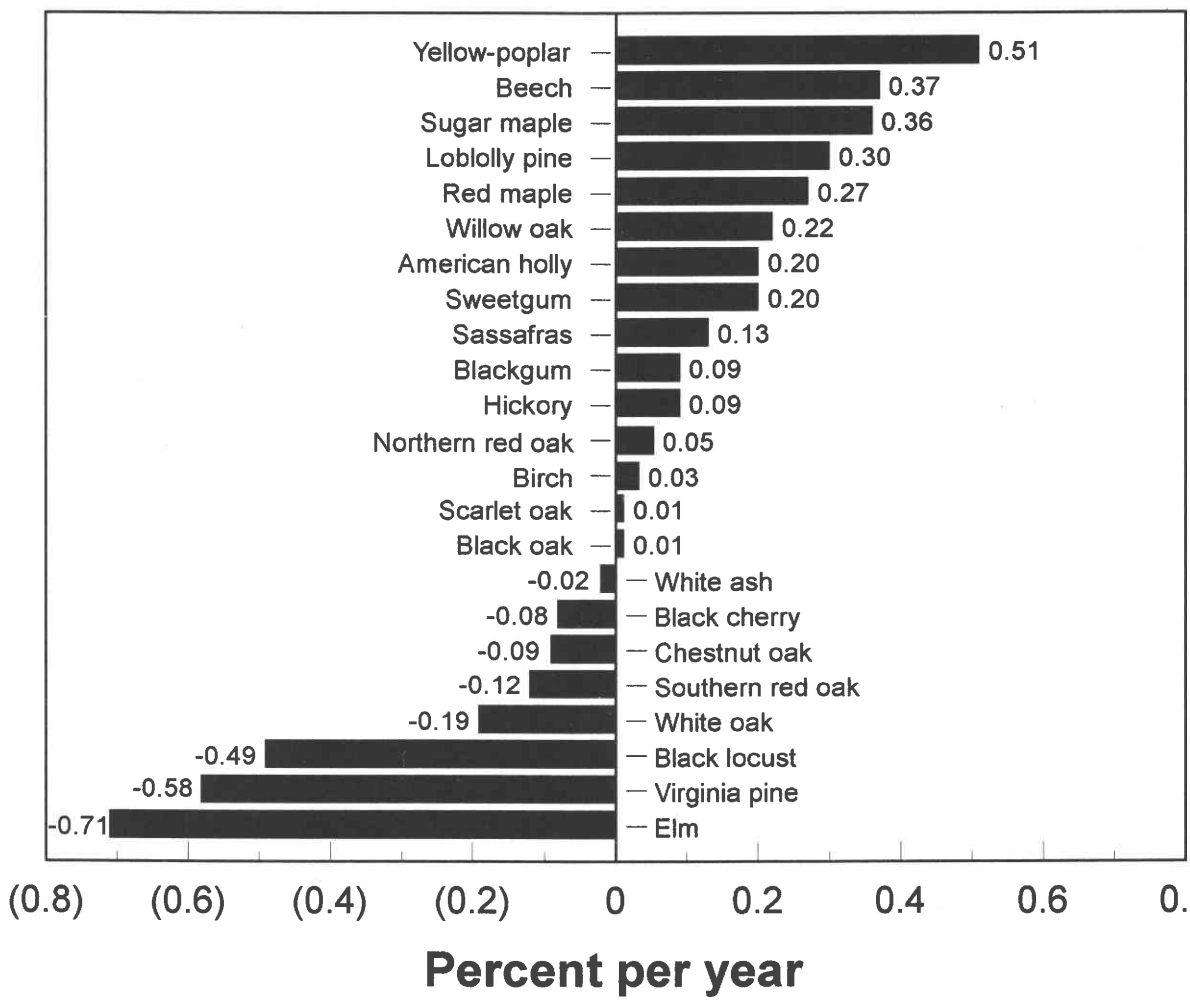

Figure 3. Average annual change in the relative stocking of some common forest tree species in Maryland, 1976-1986.

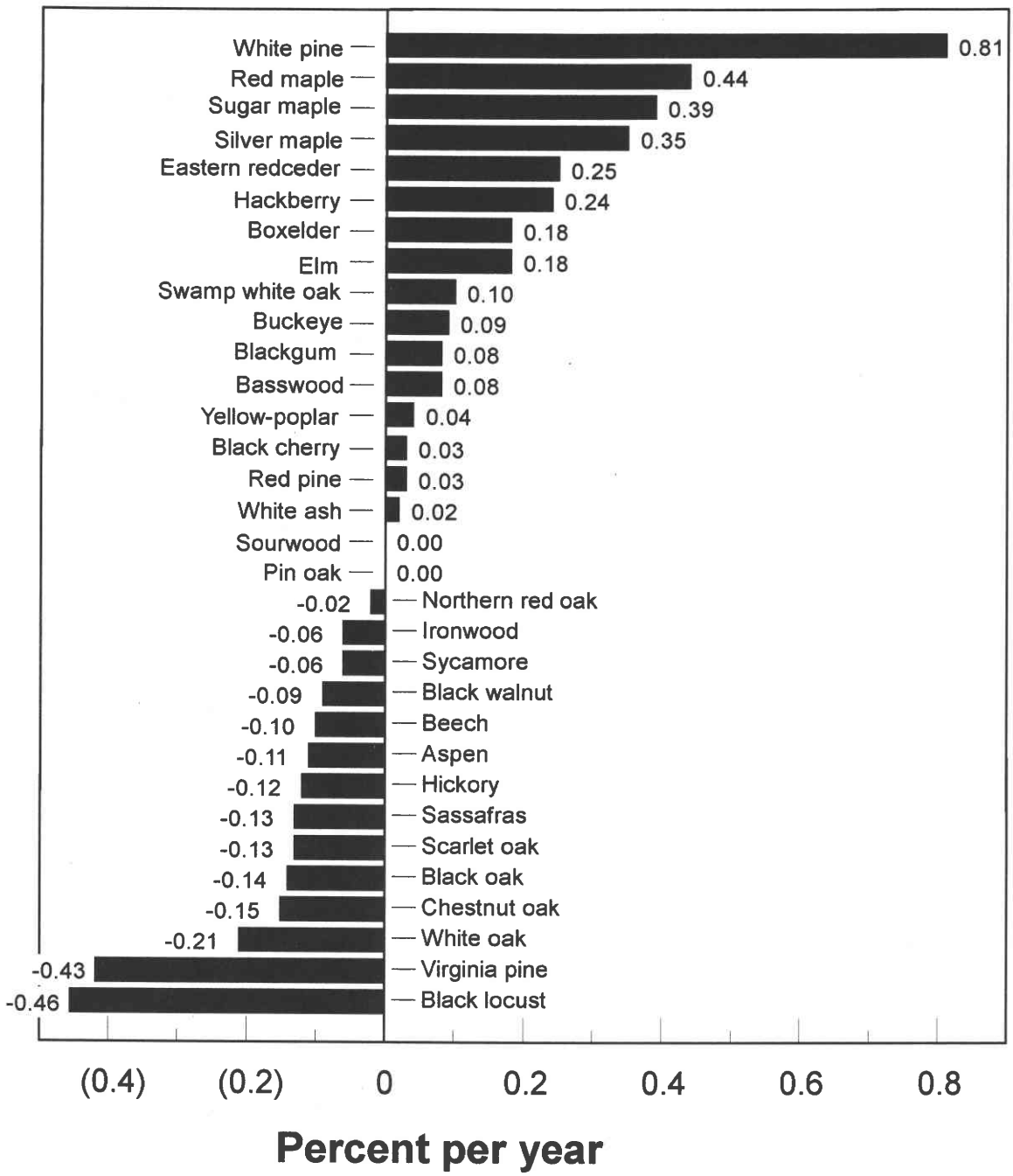

Figure 4. Average annual change in the relative stocking of some common forest tree species in Ohio, 1979-1991.

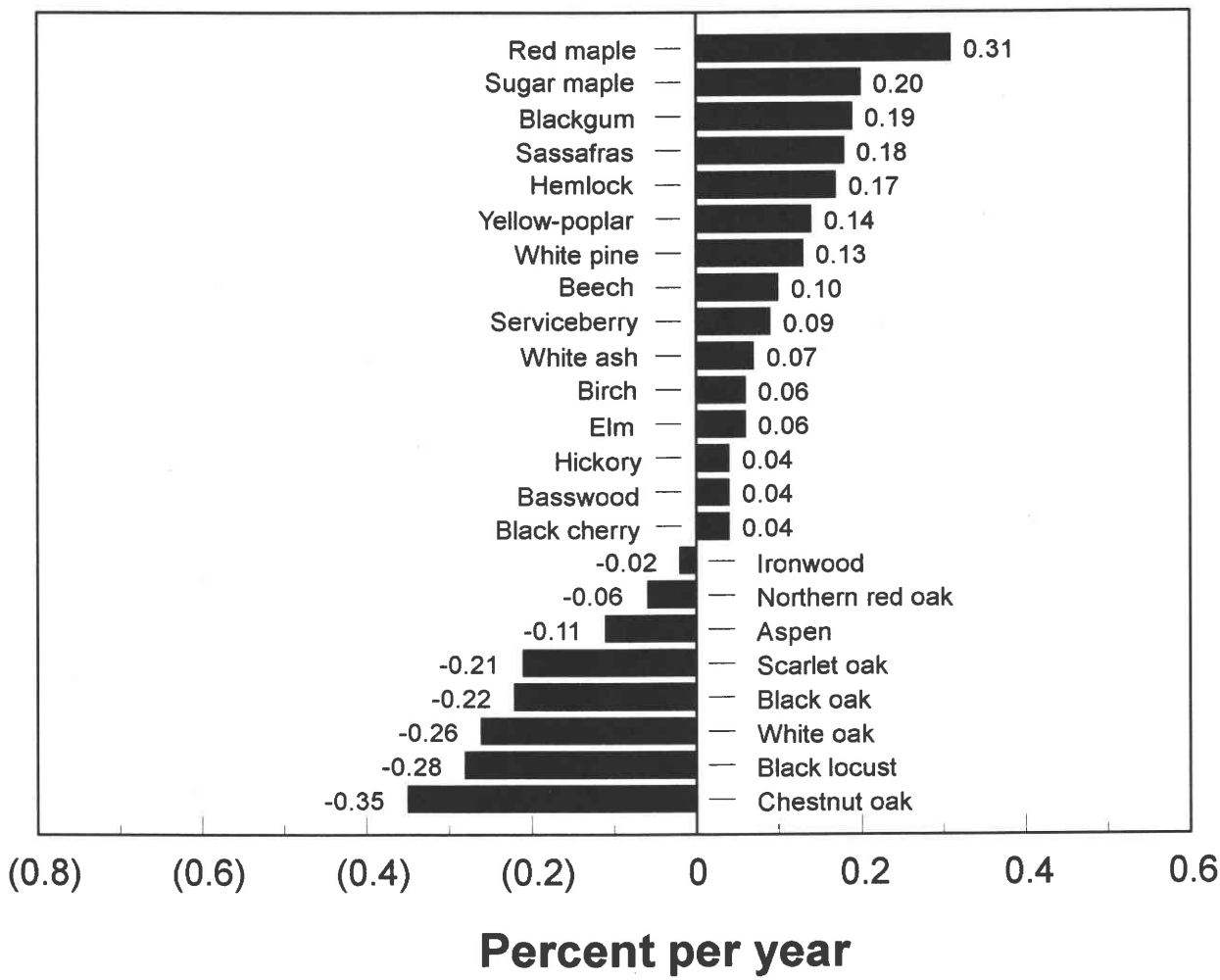

Figure 5. Average annual change in the relative stocking of some common forest tree species in Pennsylvania, 1978-1989.

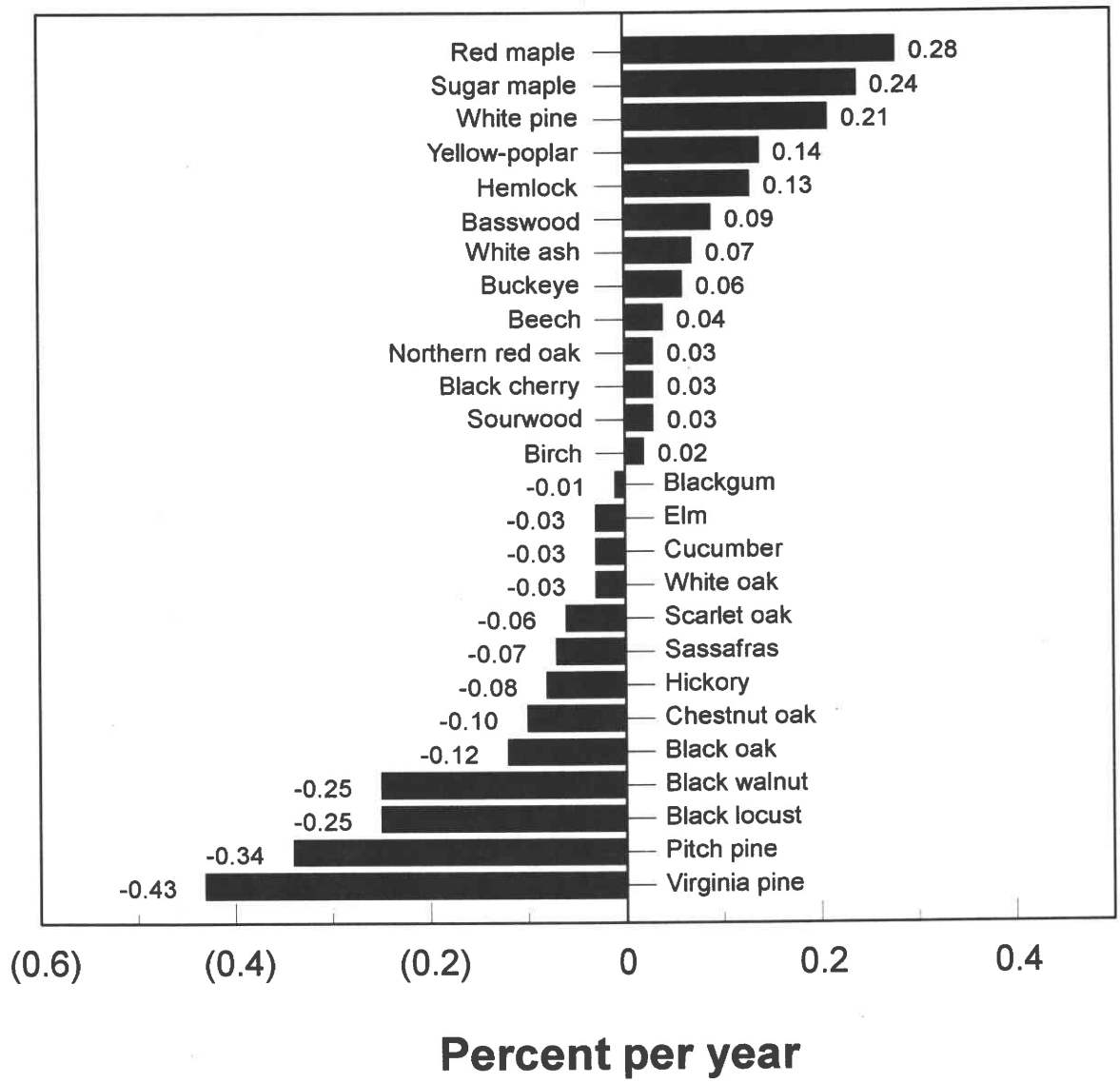

Figure 6. Average annual change in the relative stocking of some common forest tree species in West Virginia, 1975-1989.

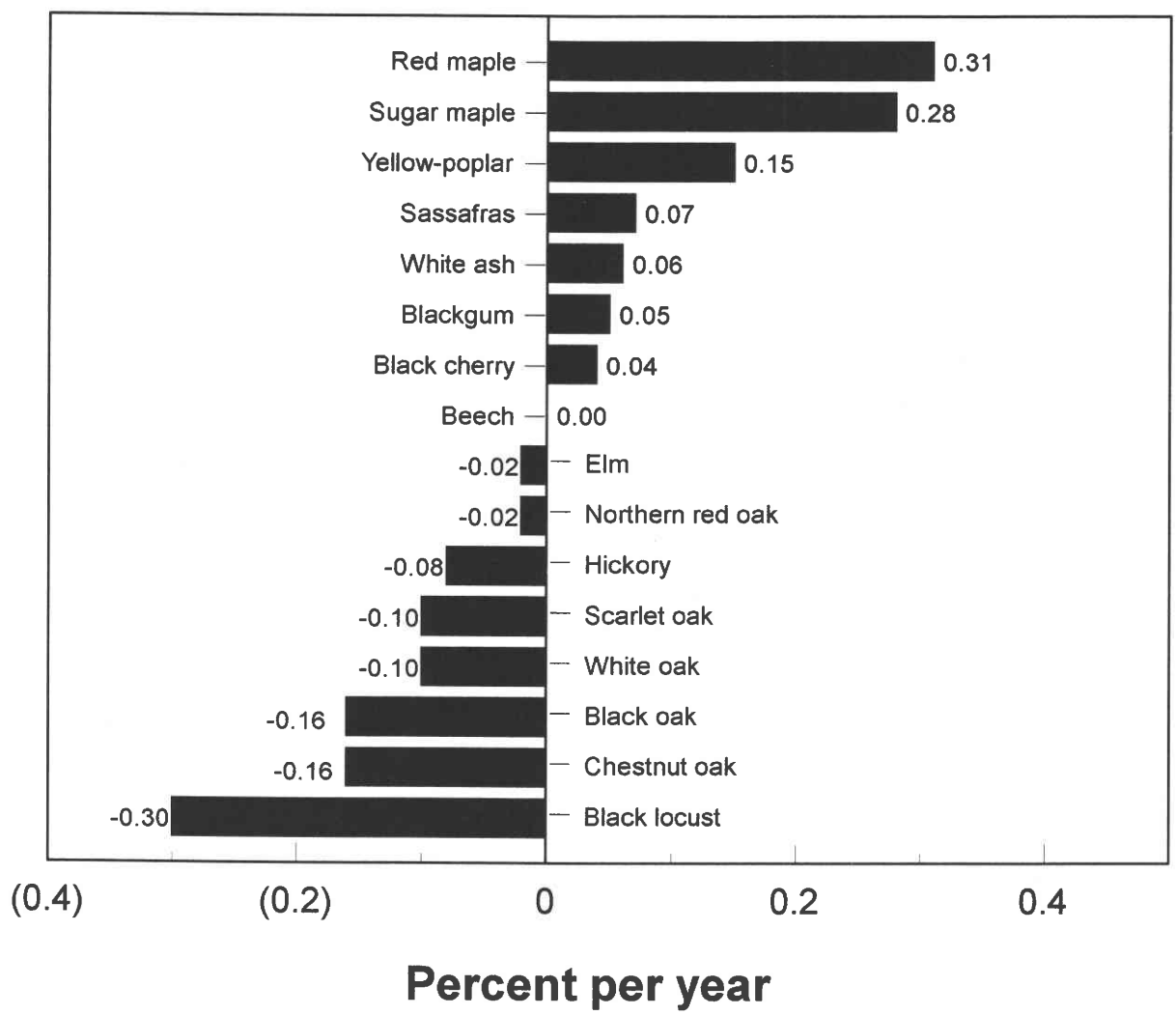

Figure 7. Average annual change in the relative stocking of forest tree species common in the five-state area.

TRACKING TRENDS AND SPOTTING PROBLEMS

Since each inventory plot represents a proportional share of the forest area in a county, appropriate weights can be assigned to plot data to derive average trends in relative stocking for individual species in each county. County averages can then be mapped to provide a closer look at where a species is gaining or losing ground. Maps showing county trends for three of the area's top gainers (red maple, sugar maple, and yellow-poplar) and three of its losers (black locust, chestnut oak, and black oak) are presented here (Figures 8-13).

Using county averages to gauge trends in relative stocking has its limitations. Some county averages, especially those for sparsely forested counties, are based on very few ground plots and may be subject to high sampling errors. Locations of individual ground plots have been digitized, so trends for each plot also can be mapped. These maps provide a more specific view of spatial shifts in relative stocking (Figures 14-15).

Together, maps showing trends for counties and individual plots can be used to detect areas where species are gaining or losing ground and, thus, help us locate potential problems in tree health. For example, figure 12 shows that chestnut oak has been losing ground in several of the region's counties. Figure 14 tells us that declines were especially noticeable in Pennsylvania where chestnut oak occurred on one-third of the remeasured plots and was losing ground on two-thirds of them. Bedford County in south-central Pennsylvania epitomizes the worst case. Here, change in the relative stocking of chestnut oak averaged
-0.86 percent per year. Dead trees and stumps recorded on Bedford County plots provide clear evidence of why. For a first-hand view of the situation, take a ride on the Pennsylvania turnpike and check out the landscape around Everett, Pennsylvania. Some good news here is that most of the oak stands that were devastated by gypsy moth, drought, cutting, and other agents during the 1980's have regenerated to a more diverse mix of species such as maple, birch, cherry, ash, and yellow-poplar that are less susceptible to the gypsy moth (Gansner et al. in press).

For a completely different scenario, take a look at sugar maple. Figures 9 and 15 indicate that this species recorded large gains in relative stocking in all five states. But the maps also show that while sugar maple was performing well regionwide, it was losing ground in some places such as the Allegheny Plateau in Elk, Forest, McKean, and Warren Counties, in northwestern Pennsylvania. Annual change in the relative stocking of sugar maple in these four counties averaged -0.12. The reasons for this change are not apparent.

IMPLICATIONS

The procedure used here to track shifts in relative stocking can be used for virtually any species anywhere. Results provide an extensive look at where each species is gaining or losing ground and, thus, a means for locating current and potential problems in tree health.

Findings of our analysis raise more questions than they answer. Will red maple continue its rapid gains in relative stocking and, if so, how will that affect the integrity of the region's forested ecosystems? Will oaks continue to lose ground or will they overcome the effects of gypsy moth and other stress that hit them hard during the last couple of decades? Why is sugar maple making significant gains regionwide but losing ground on the Allegheny Plateau of Pennsylvania--is it pear thrips, drought, overstocking, a combination of these factors, or none of the above? Many are hypothesizing that sugar maple just does not do well in dense maturing stands. Will black locust continue to disappear from the scene as overall stocking continues to improve? Is white pine on the verge of regaining the status it once held? What is happening to the hard pines? And what about hickory, especially in Kentucky, Ohio, and West Virginia?

Information on shifts in relative stocking can provide a symptomatic guide to recognizing problems of forest health. It definitely gives us a better understanding of the complex workings of a dynamic ecosystem. This information can help us make better decisions about the management and use of our precious forest resource.

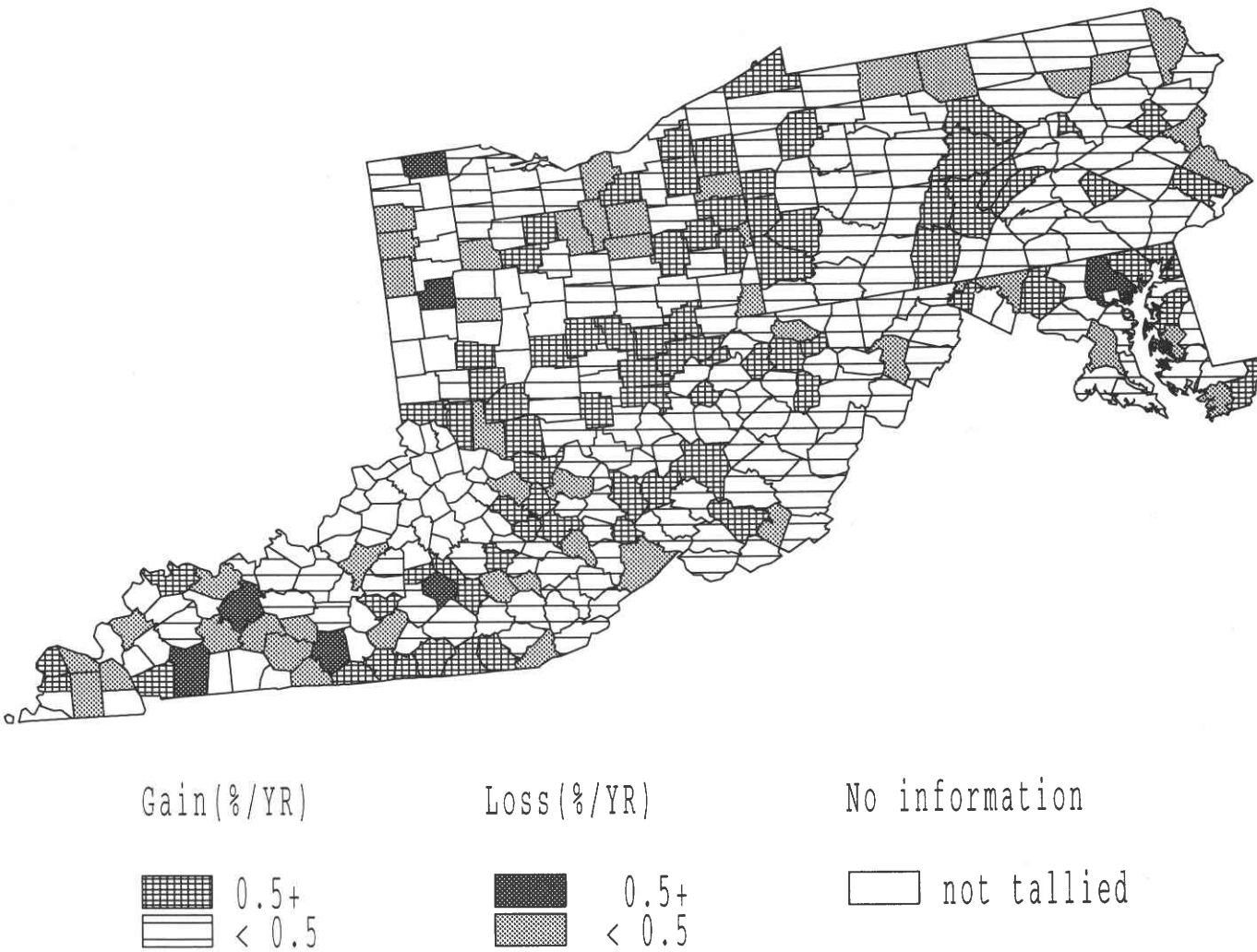

Figure 8. Average annual change in the relative stocking of red maple, by county.

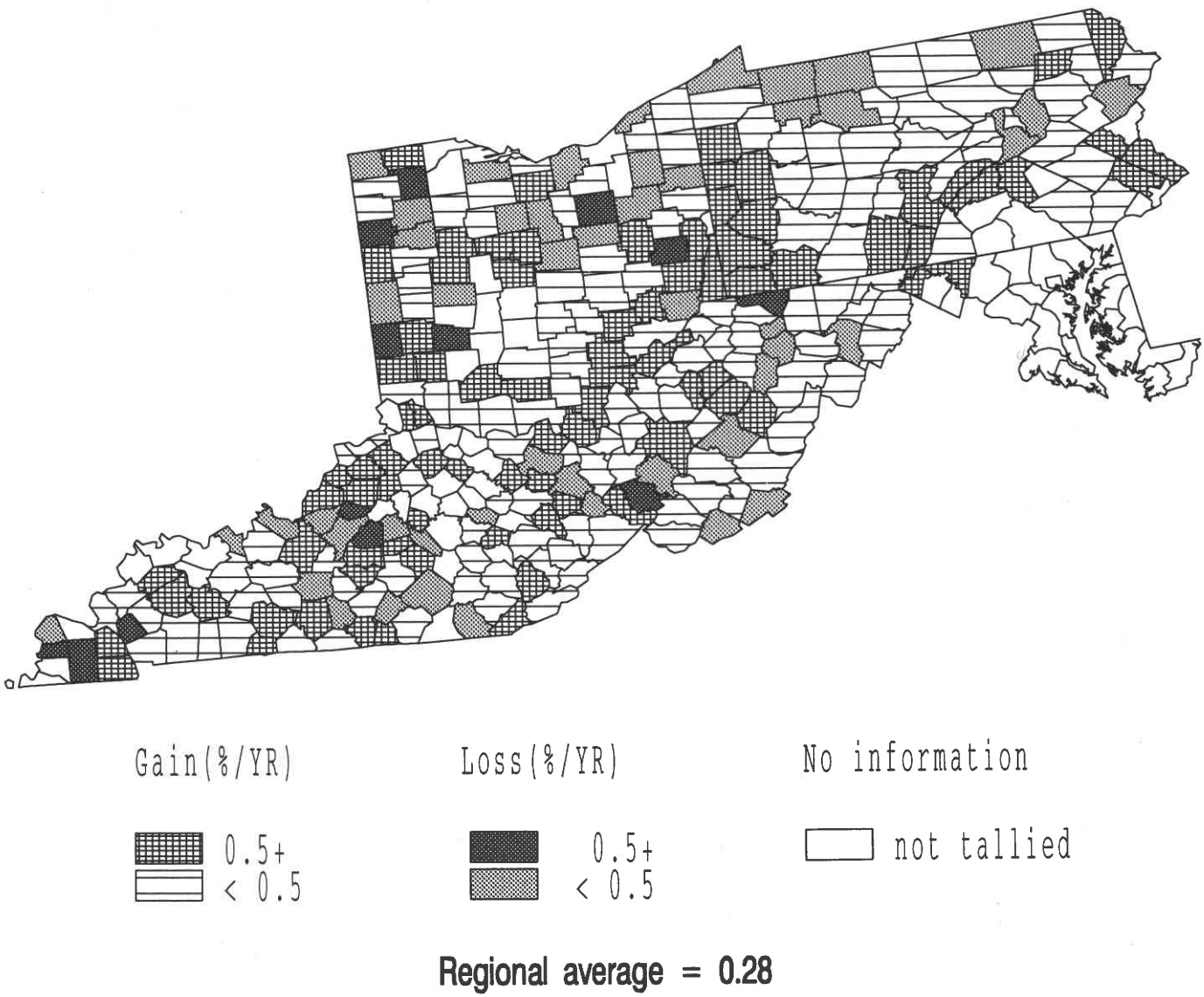

Figure 9. Average annual change in the relative stocking of sugar maple, by county.

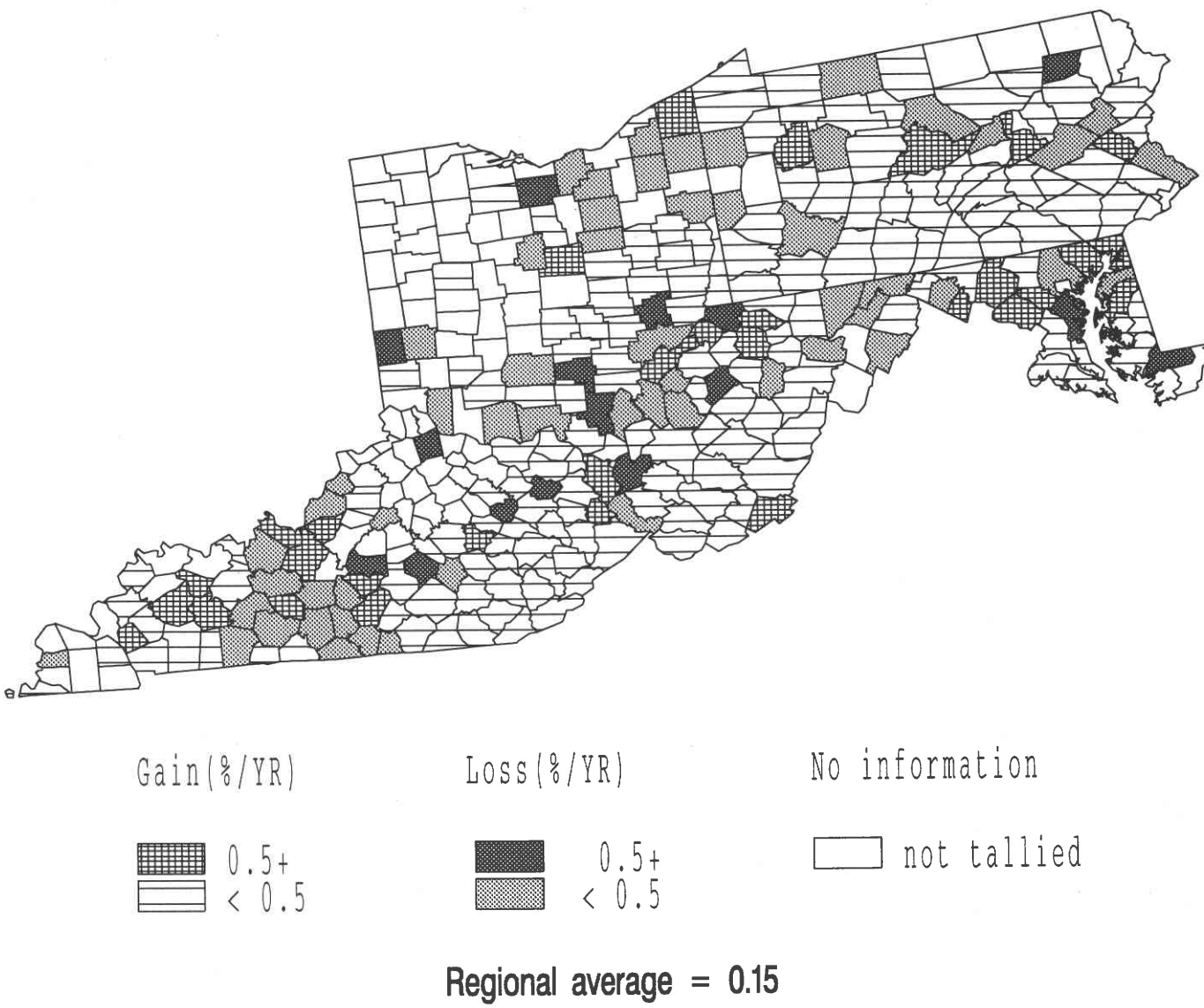

Figure 10. Average annual change in the relative stocking of yellow-poplar, by county.

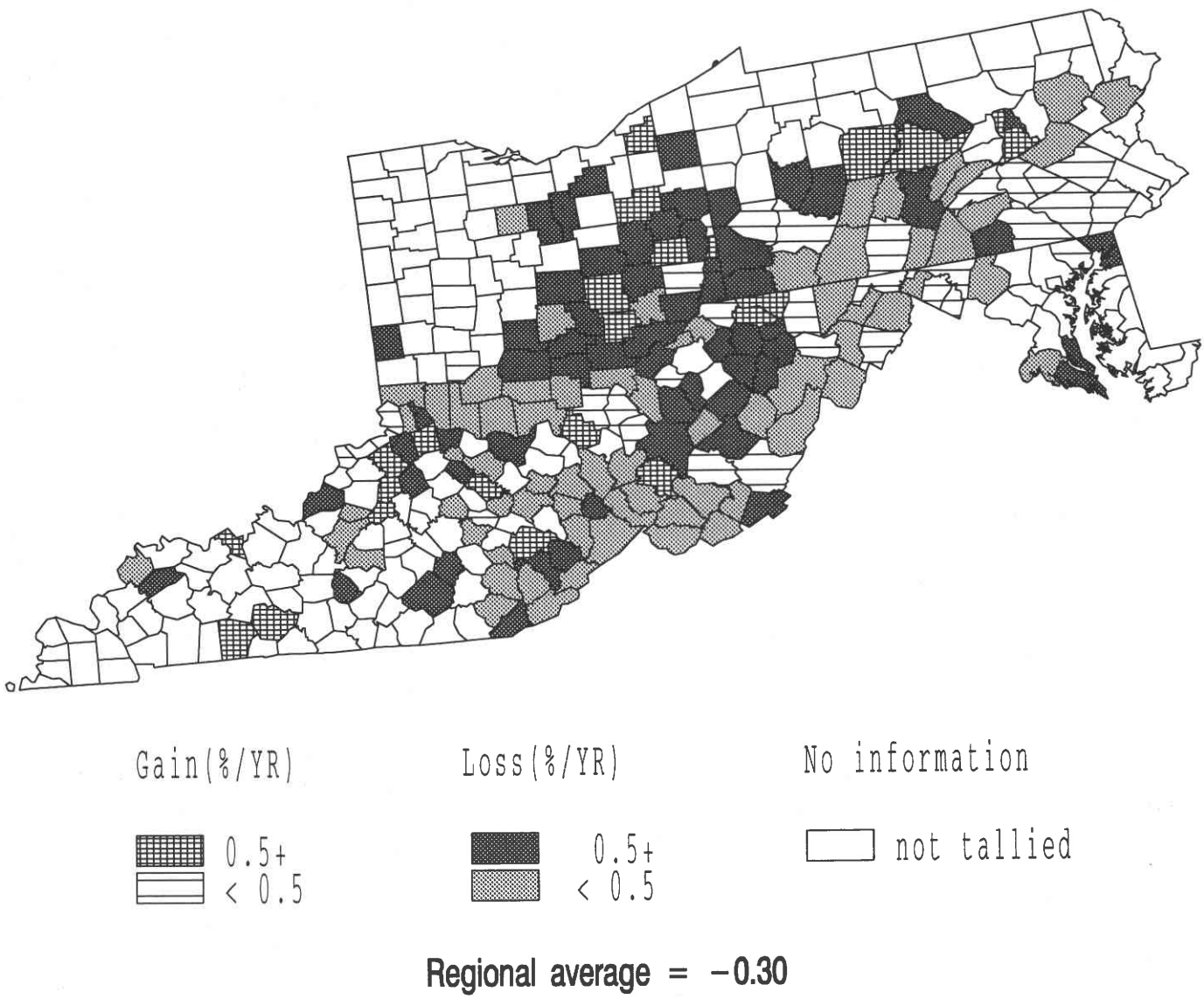

Figure 11. Average annual change in the relative stocking of black locust, by county.

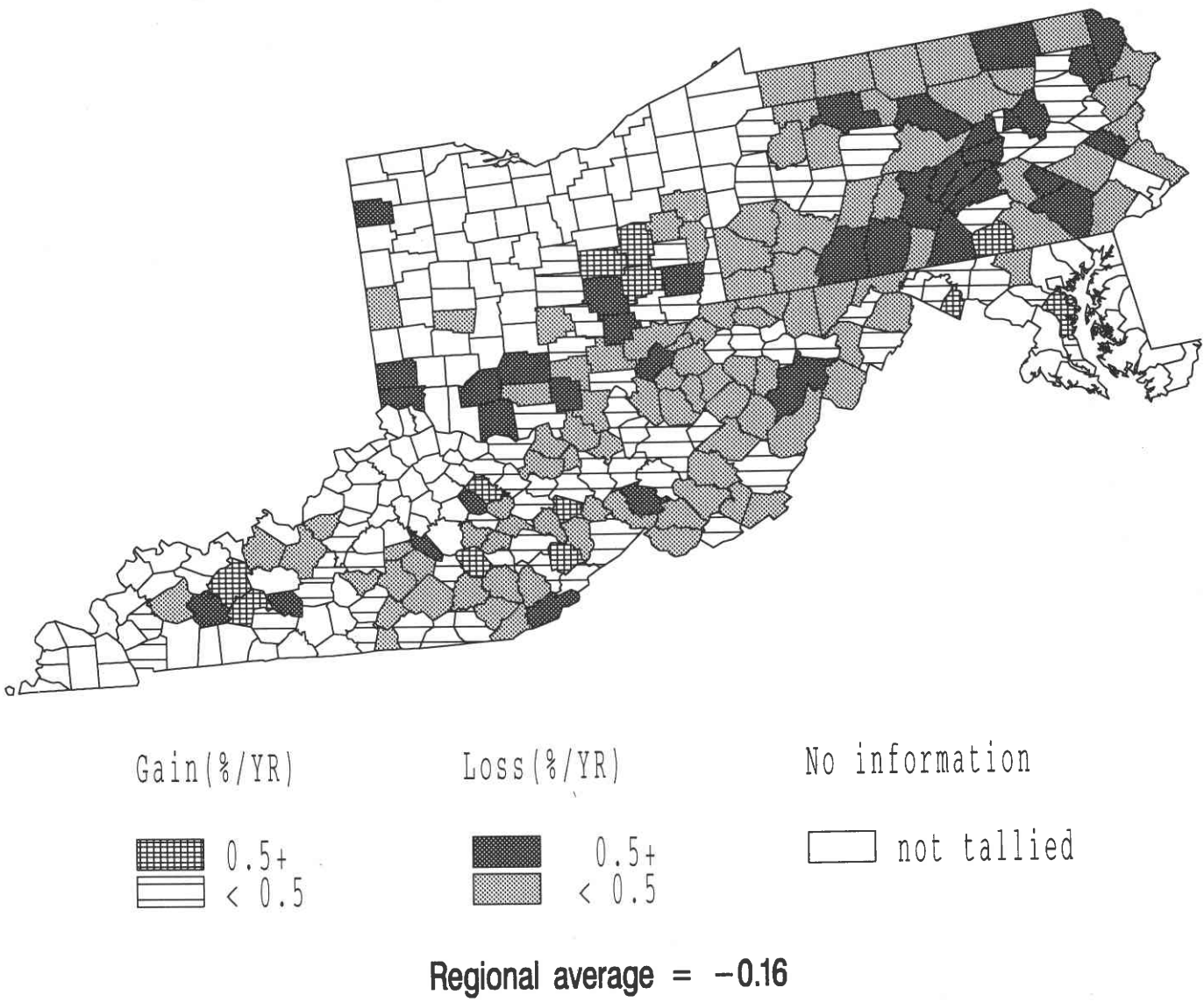

Figure 12. Average annual change in the relative stocking of chestnut oak, by county.

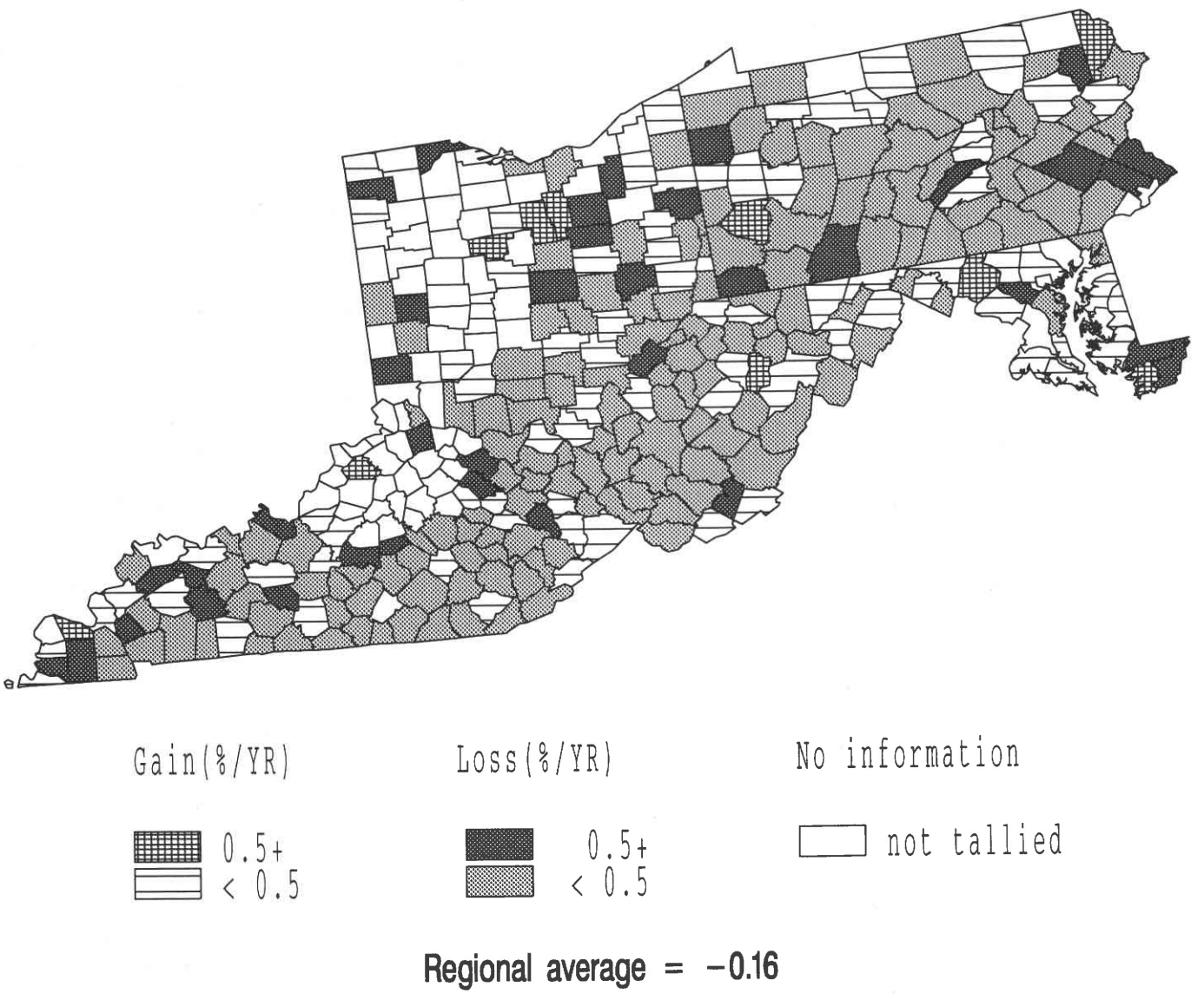

Figure 13. Average annual change in the relative stocking of black oak, by county.

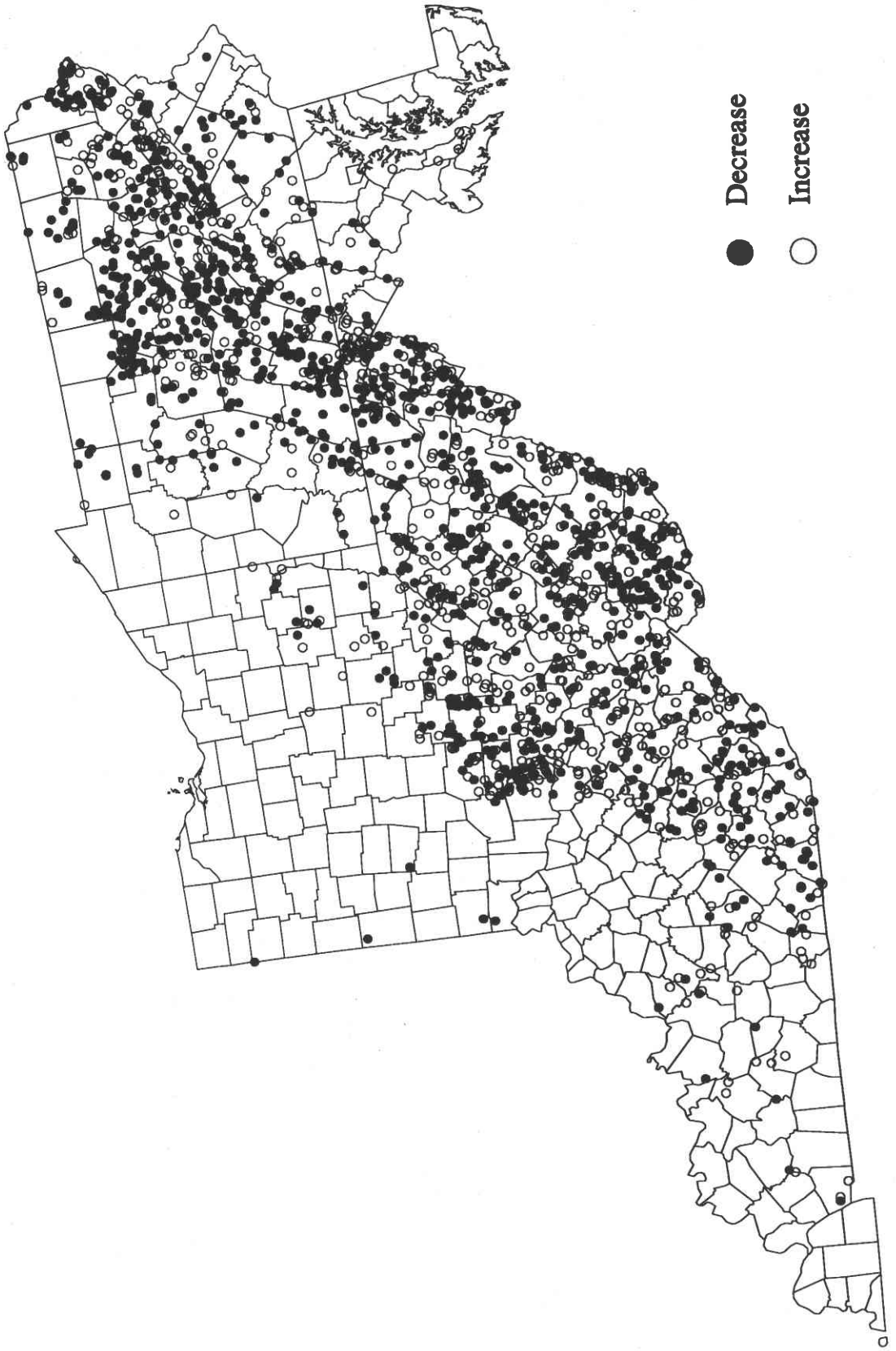

Figure 14. Change in the relative stocking of chestnut oak on plots between inventories.

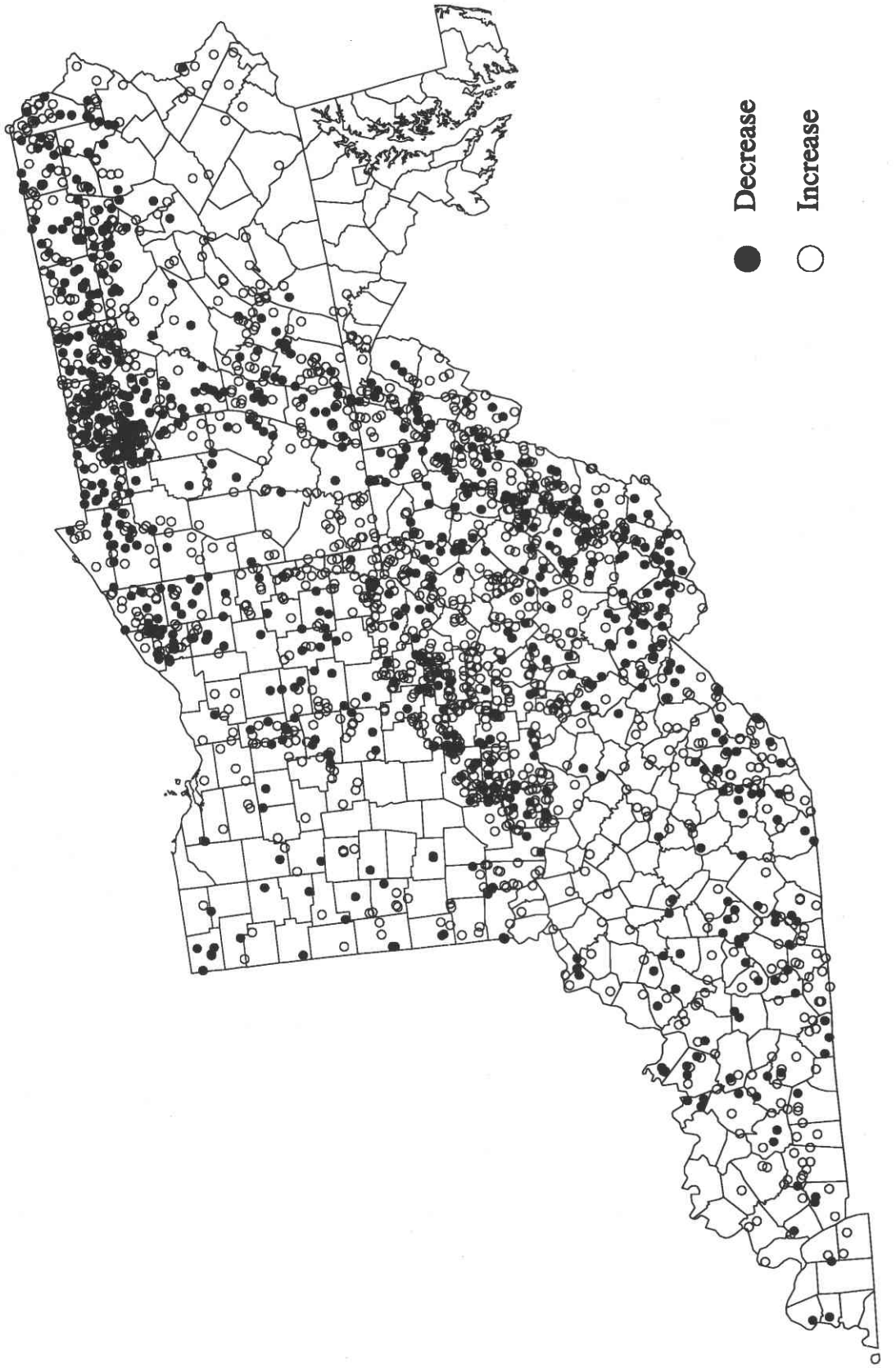

Figure 15. Change in the relative stocking of sugar maple on plots between inventories.

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DIMILIN EFFECTS ON LEAF-DECOMPOSING AQUATIC FUNGI ON THE FERNOW EXPERIMENTAL FOREST, WEST VIRGINIA

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Abstract: Dimilin was applied to two watersheds on the Fernow Experimental Forest on May 16, 1992, as part of a study to evaluate its effect on non-target organisms. Data were obtained on the occurrence, conidial production, and leaf litter colonization of aquatic hyphomycetes 5 days prior to and 2, 10, 25, and 55 days following application in the two treated watersheds and two control watersheds. Two days after application, conidial numbers were higher in one treated watershed, while they remained relatively constant in the other treated watershed. Northern red oak (*Quercus rubra* L.) and sugar maple (*Acer saccharum* Marsh.) leaf bags placed in the weir ponds and streams of the treated watersheds prior to Dimilin application and retrieved 2 days after application were colonized by greater numbers of fungal taxa than bags retrieved later. This pattern suggests the possibility of an increasing influence of this pesticide on the occurrence and the litter decomposition activities of this group of aquatic fungi.

INTRODUCTION

Rapid deterioration of the quality of oak forests due to invasion of the gypsy moth (*Lymantria dispar* L.) has led to the widespread use of Dimilin (Diflubenzuron) as a method to control this introduced insect pest. Dimilin is a chitin synthetase inhibitor and affects immature insects during molting. It is ingested as the insect larvae feed on treated foliage. However, other non-target organisms also may be affected negatively.

Aquatic hyphomycetes dominate the assemblages of aquatic fungi associated with decaying leaves in lotic systems (Bärlocher and Kendrick 1974, Suberkropp and Klug 1976, Trisca 1970). Decomposition of leaf detritus and its exploitation by other members of stream detrital communities are largely dependent upon the activities of aquatic hyphomycetes, since hyphomycetes have the enzymatic capability to digest the structural polymers that comprise most dead plant tissue (Bjarnov 1972, Monk 1976). Thus, any change in the physicochemical or biological environment of a stream will influence aquatic hyphomycete activity. Consequently, concerns exist about how Dimilin might affect aquatic hyphomycetes.

The present paper compares aquatic fungal occurrence data in two watersheds treated with Dimilin and two other watersheds maintained as controls. Conidia in stream water samples were enumerated and fungal species colonizing northern red oak (*Quercus rubra* L.) and sugar maple (*Acer saccharum* Marsh.) leaves placed in the four watersheds were identified.

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METHODS

Site Descriptions

Four small watersheds (1, 4, 7, and 13) on the USDA Forest Service's Fernow Experimental Forest, near Parsons in Tucker County, West Virginia, were used for this study. The Fernow Experimental Forest is located in the Allegheny Mountain section of the unglaciated Allegheny Plateau (79° 0' 11" W, 39° 0' 05" N). Elevations range from 530 to 1100 m; slopes range from 10 to 60 percent. Mean annual precipitation is 147 cm, and the mean annual temperature is 9°C. Soils are derived from sandstone and shales of the Hampshire formation. All four watersheds were logged heavily between 1905 and 1910. However, since then their disturbance histories have been quite different.

All the trees on watershed 1 (30.1 ha), except for cull trees and trees less than 2.4-cm dbh, were harvested from May 1957-June 1958. A "logger's choice" method was used such that no specific road construction, harvesting, or skidding guidelines were required. Seventy-four percent of the original basal area was cut, with 24,527 bd ft ha^{-1} removed (Kochenderfer and others 1990). The current stand is 34 years old and is dominated by yellow-poplar (*Liriodendron tulipifera* L.), chestnut oak (*Q. prinus* L.), and red maple (*A. rubrum* L.). In 1986, watershed 1 was treated with Dimilin at a rate of 0.067 kg active ingredient ha^{-1} with a fixed-wing aircraft. This treatment was imposed to measure the persistence of Dimilin in stream water, sediment, organic matter, and throughfall (Jones and Kochenderfer 1987).

Watershed 4 (38.7 ha) has had negligible disturbance since 1910, except for salvage removal of dead American chestnut (*Castanea dentata* (Marsh.) Borkh.) during the 1940s following the chestnut blight (*Cryphonectria parasitica* (Murrill) Barr). Currently, the dominant stand is about 90 years old, though scattered residual trees left from the turn-of-the century logging are estimated to be 175 to 200 years old. The dominant overstory species are sugar maple, red maple, and American beech (*Fagus grandifolia* Ehrh.).

Watershed 7 (24.2 ha) was harvested in two halves in the 1960s. The upper 12.1 ha, comprising 49 percent of the basal area (80.2 m ha^{-1}), were clearcut from November 1963 to March 1964. This half was maintained barren of vegetation with herbicides, principally 2,4,5-T, until October 1969. The lower half of watershed 7 was clearcut (76.9 m ha^{-1}) from October 1966 to March 1967 and also maintained barren until October 1969 (Patric and Reinhart 1971; Kochenderfer and others 1990). Today, the overstory vegetation is dominated by black birch (*Betula lenta* L.), red maple, and sugar maple.

Less is known about the history of watershed 13 (14.2 ha) than of the other watersheds because it became an active research watershed only in 1984. A light selection cut is believed to have been performed in the 1960s, although the volume of wood or percent basal area removed is not known. The dominant overstory vegetation is yellow-poplar, northern red oak, and sugar maple. Stand age is approximately 65 years.

Dimilin Application

For the current study, watersheds 1 and 13 were treated with Dimilin and watersheds 4 and 7 were maintained as untreated controls. The Dimilin was applied at a rate of 0.03 kg active ingredient ha^{-1} on the morning of May 16, 1992. The application was made using a Bell 206 Jet Ranger helicopter. The spraying was done using protocols to ensure that the Dimilin application was confined to the treated watersheds (USDA Forest Service 1991).

Sampling Methods

Aquatic hyphomycete distributions in the four watersheds were studied by membrance filtration of stream water samples (Iqbal and Webster, 1973) and by leaf bag incubation in both streams and weir ponds (Musil and Shearer, 1982). For membrane filtration, 250-ml water samples were collected from each stream 5 days prior to and 2, 10, 25, and 55 days following Dimilin application. Ten samples were collected from each stream on the first three sampling dates; because of low water levels, only five samples were collected from each stream on the last two sampling dates.

Samples were filtered through Millipore 0.45-μm membrane filters in the field at the time of collection. In the laboratory, each filter was treated with 0.1 percent cotton blue and heated in lactic acid at 50-60°C to render it sufficiently transparent for conidial enumeration and identification. Filters were examined microscopically at 100x, and conidia were counted. Taxa were recorded for 40 noncontiguous fields-of-view (36.2 mm^2) on each filter.

For leaf bag incubation, mesh bags containing leaves from northern red oak and sugar maple were prepared in the manner described by Musil and Shearer (1982) and Chamier and others (1984). The leaves were collected in October 1991 shortly after leaf abscission, air-dried, and stored until leaf bags were prepared. Each bag contained approximately 2.0 g of dried leaves of either red oak or sugar maple. One set of four bags of each leaf type was tied to a perforated brick. These were placed in the streams and weir ponds of the four watersheds on May 11, 1992. Bags of each leaf type were retrieved on each of four visits following application of Dimilin (i.e., after 2, 10, 25, and 55 days). Upon retrieval, leaf bags were placed in plastic zipper-lock storage bags, partially filled with stream water, and returned to the laboratory. Leaves were removed from the mesh bags, gently washed in sterile distilled water, and cut into 1 x 1-cm squares. A few squares were examined microscopically to determine the presence of hyphomycetes. The remaining squares were incubated in aerated chambers filled with sterile distilled water, as described by Shearer and Webster (1991). After 24 hours, the water was filtered through a membrane filter, and conidia were identified by microscopic examination of the filter paper after treatment with cotton blue and lactic acid, described previously. Fungal specimens recovered from leaf bags were studied as living material and/or fixed and stained with cotton blue and mounted in lactophenol. Fungal species were identified using keys and description provided in Ainsworth and others (1973), Barnett and Hunter (1972), Ellis (1971), Nilsson (1964), Ingold (1975), and Subramanian (1983). Nomenclature used in this paper follows that given by these authors.

Data Analysis

Separate data sets were compiled for aquatic hyphomycetes recorded as conidia filtered from water samples and for those occurring on leaves. The four streams were compared using coefficient of community (CC) indices (Mueller-Dombois and Ellenberg 1974). The equation for this index, which is based solely on the presence or absence of taxa, is

$$CC = 2c/(a+b) \quad (1)$$

where a = total number of taxa in the first stream being considered, b = total number of taxa in the second stream, and c = number of taxa common to both streams. The CC value ranges from 0.0 (when no taxa are common to both streams) to 1.0 (when all taxa are common to both streams).

RESULTS

Water Filtration

Two days after the Dimilin application (May 18), the number of conidia in samples from watershed 1 was approximately twice that recorded 5 days prior to treatment (May 11) (Table 1). However, the number of conidia recorded in samples from the other treated watershed decreased from May 11 to May 18. In both treated watersheds, conidial numbers generally increased on subsequent sampling dates. Conidial numbers recorded from control watersheds 4 and 7 also declined from May 11 to May 18 and remained below the pretreatment levels throughout the subsequent sampling dates.

Numbers of fungal taxa recorded on May 18 did not differ significantly between the treated and control watersheds (Table 2). Fewer taxa generally were observed in June and July when compared to the May sampling periods for both treatment and control watersheds. This decline may be the result of the lower water levels or higher water temperatures common in these headwater streams during the growing season.

Table 1. Occurrence of aquatic hyphomycetes as indicated by the presence of conidia filtered from water samples prior to and after the Dimilin application to watersheds 1 and 13, with watersheds 4 and 7 serving as controls. Data are numbers of conidia per 1000 ml of water.

	Sampling Date				
Watershed	May 11	May 18	May 27	June 11	July 12
1	797	1660	359	304	407
13	684	485	262	314	347
4	860	498	414	213	543
7	1834	584	231	110	343

Table 2. Numbers of aquatic hyphomycete taxa recorded from filtered water samples prior to and after the Dimilin application to watersheds 1 and 13, with watersheds 4 and 7 serving as controls.

	Sampling Date				
Watershed	May 11	May 18	May 27	June 11	July 12
1	30	29	19	17	15
13	18	16	20	14	16
4	27	24	24	14	19
7	23	25	14	14	14

Overall, 108 fungal taxa were recorded by means of membrane filtration of water samples from all four watersheds on the five sampling dates. This total included 79 fresh water forms, or Ingoldian hyphomycetes; 24 terrestrial geofungi; and 5 aeroaquatic fungi, which typically occur in marshy environments. Only 25 taxa were common to all four watersheds.

The CC indices calculated from pooled filtration data for all sampling dates (Table 3) indicate that watersheds 1 and 4 were the most similar streams (CC = 0.687), even though watershed 1 was treated with Dimilin and watershed 4 was not treated. Watersheds 7 and 13 were the least similar (CC = 0.556). The average CC value for all possible combinations of streams was 0.638.

Table 3. Community coefficient indices calculated from water filtration data.

Watershed Comparisons	Coefficient of Community
1-13	0.672
1-4	0.687
1-7	0.631
4-7	0.630
4-13	0.654
7-13	0.556

Leaf Bag Colonization

Like the filtration data, leaf bag colonization data (Table 4) are quite variable. In general, more aquatic hyphomycete taxa were associated with red oak leaves than with sugar maple leaves in both the streams and the weir ponds. In most instances, the highest numbers of taxa for both leaf types were recorded on the first sampling date. The maximum number (28) was recorded for red oak in the weir pond of watershed 7 on June 11, whereas the minimum number (4) was recorded for the same leaf type in the weir pond of watershed 13, also on June 11. Overall, numbers of taxa recorded from the two types of ecological situations (i.e., weir ponds and streams) were remarkably similar.

A total of 64 fungal taxa colonized red oak leaves. Forty-six of these taxa were freshwater hyphomycetes. Fifteen species of terrestrial geofungi and 3 species of aeroaquatic hyphomycetes also were recorded. Nine taxa were present only in the treated watersheds, whereas 12 species were restricted to the control watersheds.

Colonization of sugar maple leaves was similar to that of red oak leaves. Sixty-five fungal taxa were recorded. Fifty-one of these taxa were fresh water hyphomycetes, 11 were terrestrial geofungi, and 3 were aeroaquatic hyphomycetes. Nineteen taxa were found only in treated watersheds, while 7 were restricted to the control watersheds. *Anguillospora crassa* and *Flagellospora curvula* were the only taxa recorded from both filtered water samples and leaf bags, with 100 percent consistancy.

Coefficient of community indices calculated from leafbag colonization data are summarized in Table 5. The highest CC value was 0.666, which was recorded for pairwise combinations of watersheds 1 and 4, 1 and 7, 4 and 7, and 1 and 13. Four of these involved red oak and three involved sugar maple. The lowest CC value (0.303) was recorded for sugar maple in watersheds 1 and 13.

Table 4. Aquatic hyphomycete colonization of red oak and sugar maple leaf bags.

Watershed	Leaf type	Stream						Weir pond				
		May 11[a]	May 18	May 27	June 11	July 12	Mean	May 18	May 27	June 11	July 12	Mean
		Number of taxa										
1	Sugar maple		6	8	15	3	8.0	13	6	15	7	10.3
	Red oak		17	8	6	14	11.3	16	5	7	8	9.0
		19										
13	Sugar maple		23	12	6	8	12.3	27	6	11	15	14.8
	Red oak		19	19	12	17	16.8	21	9	4	18	13.0
		14										
4	Sugar maple		15	10	14	16	13.8	10	10	10	9	9.8
	Red oak		19	5	10	12	11.5	12	14	13	15	13.5
		13										
7	Sugar maple		16	13	15	9	13.3	16	13	11	12	13.0
	Red oak		14	16	21	12	15.8	15	5	28	9	14.3
		14										

[a]Present on natural substrates prior to Dimilin application.

Table 5. Coefficient of community (CC) indices calculated from red oak (RO) and sugar maple (SM) leaf bag colonization data obtained on four dates following Dimilin application.

	Sampling Date							
	May 18		May 27		June 11		July 12	
Watershed Comparisons	RO	SM	RO	SM	RO	SM	RO	SM
1-13	0.622	0.451	0.387	0.583	0.545	0.514	0.514	0.303
1-4	0.666	0.555	0.416	0.518	0.451	0.486	0.500	0.416
1-7	0.666	0.611	0.666	0.551	0.380	0.500	0.600	0.384
4-7	0.666	0.500	0.533	0.666	0.509	0.540	0.562	0.666
4-13	0.625	0.528	0.540	0.528	0.571	0.437	0.594	0.594
7-13	0.577	0.490	0.540	0.592	0.434	0.514	0.564	0.410
Average CC Value	0.637	0.522	0.513	0.573	0.481	0.498	0.555	0.462

DISCUSSION

Filtration data indicate that conidial numbers in watershed 1 increased following the Dimilin application, but a similar increase did not occur in watershed 13. In fact, numbers of conidia decreased from May 11 to May 18 for all watersheds except watershed 1. Although the decrease in watershed 13 was less than that observed in the two control watersheds, there is not enough of a difference to suggest that the application of Dimilin caused any sudden increase or decline in fungal occurrence in the treated watersheds.

Generally higher numbers of fungal taxa were observed in leaf bags from treated watersheds than in those from control watersheds. This difference was most obvious on May 18 and was more apparent in the weir ponds of treated watersheds than in streams of the same watersheds. Dimilin may have a greater opportunity to accumulate in weir ponds and thus influence growth and sporulation of litter-decomposing hyphomycetes. Ongoing investigations on residual analysis of Dimilin do indicate persistence of Dimilin in litter and soil within the treated area (Wimmer 1994).

Overall, numbers of fungal taxa recorded from filtration and leaf bag colonization in treated watersheds on May 18 were comparable to or even higher than those recorded on subsequent sampling dates. This pattern demonstrates the tolerance of these fungi to biochemical change. It seems likely that decreases in fungal occurrence and numbers of conidia during June and July may be related to the low streamflow and resultant elevated temperatures.

The possible effects of Dimilin on fungal occurrence can be assessed in two ways. The first way is by examining the direct utilization of residual Dimilin in litter and soil (Wimmer 1994). In an earlier study (Dubey 1992), five species of aquatic hyphomycetes (*Clavariopsis aquatica, Heliscus lugdunensis, Lemonniera aquatica, Lunulospora curvula* and *Tetracladium marchalianum*) showed increased growth rates with increased Dimilin concentrations. However, direct effects of insecticides, such as Dimilin, on aquatic microorganisms probably are modified by a number of factors, including the extent to which the insecticide is water soluble, the contact time between the insecticide and the

fungal mycelium, the nutritional status of the environment in which the fungus interacts with the insecticide, the amount of fungal cell material present, and the initial insecticide concentration applied.

The second way to examine the effects of Dimilin involves monitoring changes that may occur in the biochemical environment of treated watersheds as a result of reduced defoliation as gypsy moth larvae are killed after the application of the insecticide. For example, reduced defoliation may result in increases in the watershed's buffering capacity, pH, and Ca and Mg status, and decreases in streamflow and NO_3 and NH_4 compared to conditions of greater defoliation (Downey and others 1994). Low streamflow rates also favor the production of allochthonous coarse particulate organic matter (CPOM) and fine particulate organic matter (FPOM), which are basically produced by the enzymatic leaf processing action of fresh water hyphomycetes. CPOM and FPOM serve as major food sources for many aquatic macroinvertebrates. Downey and others (1994) recorded a relatively high density of aquatic macroinvertebrates from mountain streams in Virginia where an abrupt crash in gypsy moth populations due to disease prevented the occurrence of a significant defoliation during the 1992-93 growing seasons. This finding illustrates the relationship between aquatic hyphomycetes and Dimilin -- Dimilin kills gypsy moth larvae, thereby minimizing defoliation, permitting CPOM and FPOM production, and contributing to increased macroinvertebrate activity and success.

CONCLUSIONS

In the present study, Dimilin application was not shown to exhibit any clear evidence of a direct influence on conidial production in treated watersheds two days after treatment. Fungal colonization data for northern red oak and sugar maple leaves suggest that fungal growth and decomposition activities in the treated watersheds were similar to or slightly greater than those occurring in the control watersheds. If aquatic hyphomycetes are affected by Dimilin, these effects are manifested in an indirect rather than a direct manner.

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THE EFFECT OF ACORN INSECTS ON THE ESTABLISHMENT AND VIGOR OF NORTHERN RED OAK SEEDLINGS IN NORTH-CENTRAL WEST VIRGINIA

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Abstract: During a 2-year investigation into the effect of small mammals on northern red oak (*Quercus rubra*) acorn survival and germination, widespread germination failure and lack of seedling vigor was apparent in control quadrats on one of two watersheds under study. Insects were present in and on the failed acorns but it was unknown whether they were primary invaders or merely scavengers. The purpose of the present study was to determine: 1) the proportion of acorn failure that could have been attributed to insect infestation, and 2) the effect of cotyledon damage due to insects on the vigor of northern red oak seedlings. Northern red oak acorns were collected from good and excellent sites on the two watersheds in October 1993. All acorns located under the crowns of 40 study trees were collected and float-tested for viability. Samples of 18 apparently sound acorns were placed in mammal exclusion cages under each of the study trees; nine acorns were buried and nine were placed on the soil surface. In late spring of 1994, any failed acorns and seedlings with cotyledons attached were dissected. Insect presence or damage, percent cotyledon damage, and overall vigor were recorded. Anaylses of variance (ANOVAs) were used to compare: 1) mean seedling vigor by watershed, year, and treatment, 2) insect infestation rates on the two sites and between treatments in 1994, and 3) the effect of cotyledon damage on seedling vigor in 1994. Important results include the identification of three insect pests in germinating northern red oak acorns in north-central West Virginia and the finding that insect infestation may decrease vigor but does not necessarily preclude germination or seedling establishment.

INTRODUCTION

During the course of a 2-year study to determine the effects of small mammals on the survival of northern red oak acorns (*Quercus rubra*), widespread germination failure and lack of seedling vigor were noted on control plots that were protected from rodent damage with screening (Gribko and Hix 1993). Surface-sown acorns fared poorly in both years on one of two watersheds under study. Insects were present in and on the failed acorns but no detailed investigation of the cause of failure was made. Since insect damage was suspected to play a role in the lack of seedling success, a study was initiated in the fall of 1993 to investigate the impact of insects on germinating northern red oak acorns.

In most studies of acorn insects, *Curculio* weevils (Coleoptera: Curculionidae) are found responsible for much of the damage to acorn crops (Brezner 1960, Tryon and Carvell 1962a and 1962b, Marquis *et al.* 1976, Gibson 1982, Weckerly *et al.* 1989, Oak 1992). Infestations of an acorn moth, *Melissopus latiferreanus* Walsingham (Lepidoptera: Oleuthreutidae), and gall-forming wasps of the genus *Callirhytis* (Hymenoptera: Cynipidae) may also be severe in certain years (Gibson 1982, Oak 1992). All of these insects have been established as primary pests of northern red oak acorns (Gibson 1982). As such, they are capable of breeding in otherwise sound acorns as they develop on the tree. In the small mammal study initiated in 1990, an intense effort was made to eliminate acorns that were infested with primary pests. Therefore, if insects did contribute to the observed seedling failure, it is highly probable that species other than these were responsible.

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Other insects, although present in collections of northern red oak acorns, have often been disregarded due to their low numbers or to the perception that they act primarily as scavengers of the frass and detritus left in acorns vacated by primary pests. Unlike primary invaders, these insects cannot breach the acorn shell and are limited to damaged, previously infested, or germinating acorns for oviposition sites. The most important insects in this group are: 1) weevils of the genus *Conotrachelus* (Coleoptera: Curculionidae) 2) an acorn moth, *Valentinia glandulella* Riley (Lepidoptera: Blastobasidae), and 3) a nitidulid sap beetle, *Stelidota octomaculata* Say (Coleoptera: Nitidulidae). Three species of *Conotrachelus* weevils have been observed breeding in acorns in the United States: *C. posticatus* Boheman, *C. naso* Leconte, and *C. carinifer* Casey. All three are capable of breeding in captivity in northern red oak acorns. (Gibson 1964). However, *C. carinifer* is primarily a coastal species more commonly found infesting bottomland oaks. It has appeared in collections of northern red, black (*Quercus velutina)*, and scarlet oak (*Quercus coccinea*) from Georgia (Gibson 1964), but Brooks (1910) makes no mention of this species in West Virginia and it will not be discussed further here. Of the remaining two species, *C. posticatus* appears to be the most serious pest of northern red oak (Winston 1956, Gibson 1964, Galford *et al.* 1988).

Gibson (1964) reports that adults of *C. posticatus* appear from June to August and begin mating and ovipositing within days of appearing. Rather than ovipositing in acorns as they mature on the tree, *Conotrachelus* weevils wait until they fall to the ground and then deposit eggs through cracks or other openings in the shell. The eggs hatch within days of being laid and the larvae pass through 5 instars before emerging from the acorn in 2-4 weeks. The larvae then burrow down into the soil and construct pupation cells by twisting their bodies until a sealed chamber has formed. It has been suggested that under natural conditions *C. posticatus* will pass the first winter in the larval stage and will not emerge as adults until late spring or early summer of the following year (Gibson 1964). The weevils will then pass their second winter in the adult form. However, both Brezner (1960) and Gibson (1964) report that under laboratory conditions *C. posticatus* will pupate immediately and overwinter as adults.

Conotrachelus weevils were historically considered to be pests of oaks in only the white oak group. However, this conclusion was reached based only on observations made in autumn when the weevils were found to infest germinating white and chestnut oak acorns (Gibson 1964, Gibson 1971, Gibson 1972). Most authors fail to mention the presence of *Conotrachelus* weevils in northern red oak or indicate that they are so rare as to be incidental (Brooks 1910, Brezner 1960, Dorsey *et al.* 1962). However, very meticulous examination of acorn crops has revealed that small numbers of *Conotrachelus* weevils can be found in damaged northern red oak acorns in the fall (Gibson 1964, Gibson 1982, Galford *et al.* 1988). They followed infestation by other insects or attacked acorns with cracked or mammal-damaged shells. Research conducted in the spring on germinating northern red oak acorns has indicated that *Conotrachelus* weevils, particularly *C. posticatus*, can be serious pests (Galford *et al.* 1988, Galford *et al.* 1991a). In addition to their habit of breeding in the cotyledons of acorns, these weevils cause considerable damage to the shoots and radicles of germinating acorns.

The acorn moth, *Valentinia glandulella*, has been reported by many authors to be a scavenger of acorns vacated by primary pests such as *Curculio* weevils (Winston 1956, Brezner 1960, Gibson 1971, Gibson 1972). It was originally assumed that the larvae of this species fed only on the frass and detritus left behind in acorns by other insects. However, Gibson (1972) suggested that *Valentinia* larvae could cause additional damage to the acorn if during their foraging they consumed the remainder of the cotyledons. Subsequently, this species has variously been described as a secondary invader of acorns (Gibson 1982), a primary pest of germinating acorns (Galford 1986), and a primary invader of sound immature acorns (Galford *et al.* 1991a). The life cycle of this species has not been widely reported; however, some basic information is available. In the fall, the adult moth seeks out damaged acorns that have fallen to the ground and deposits eggs on or in cracks and other openings in the acorn shell (Williams 1989). When the larvae hatch, they enter the acorn, spin a web over the opening, and then feed on the remainder of the embryo, the frass of the previous occupant, and other materials, such as fungi, that may be present (Winston 1956, Williams 1989). It has been suggested that the larvae overwinter as early instars (Galford *et al.* 1991a). Winston (1956) reports that maturation may take up to a year but indicates that this may be partially controlled by the availability of food. Williams (1989) and Gibson (1972) both report that *Valentinia glandulella* pupates in the nut but no indication of the duration of the pupal period is given.

Valentinia glandulella has not been found to be a particularly important pest of acorns in the fall; however, because this species has until recently been considered primarily a scavenger, some researchers may have neglected to report their findings of it. Other authors have lumped *Valentinia* and *Melissopus* together in reports of infestations by lepidopterous larvae (Myers 1978, Gibson 1982). In one of the few studies in which lepidopterous larvae were separated by species, *Valentinia glandulella* was found in 38 of 283 collections of upland oak acorns made in West Virginia and was found to infest 0.5-11% of each collection (Dorsey *et al.* 1962).

Recently, Galford (1986) found *Valentinia glandulella* to be a serious pest of sound chestnut, white, and northern red oak acorns as they germinated. The larvae consumed emerging radicle tips as the acorns germinated or gained entrance to acorns as the seed coats split and fed on the outer surfaces of the cotyledons (Galford 1986, Galford *et al.* 1988). Since the larvae did not destroy the hypocotyl, which links the developing radicle with the cotyledon, it was suggested that some of the acorns may have retained their viability. In fact, many apparently healthy seedlings were produced in this study from *Valentinia*-infested acorns. The authors suggested that the only significant impact of this moth on northern red oak regeneration might be the attraction of more serious pests such as nitidulid beetles and *Conotrachelus* weevils.

Of the three species of sap beetles in the genus *Stelidota* found in the United States, only *Stelidota octomaculata* is considered a serious pest of northern red oak acorns (Galford *et al.* 1991b). Adults of this species become active in early spring when northern red oak acorns begin to germinate. They feed extensively on the emerging radicles and then breed in the cotyledons of the acorns as the shells split (Galford *et al.* 1988). This beetle can completely destroy an acorn and, in Ohio, was found to be one of the most serious pests of germinating northern red oak acorns. Krajicek (1960) reports that sap beetles (presumably *S. octomaculata*) were found in Iowa associated with northern red oak acorns that began germination but did not produce seedlings

Only limited information is available on the effects of these insects on northern red oak regeneration. Their most serious impacts can be expected during germination; however, most investigations of acorn insects have been conducted in the fall before red oak acorns begin to germinate. Recently, some work has been completed by Galford *et al.* (1991a) in Pennsylvania on germinating acorns. They found that 92% of surface-sown and 8% of buried northern red oak acorns that were protected from mammal predation were damaged by this group of insects. Buried acorns were destroyed by weevils and sap beetles that burrowed down through loose soil. In a study conducted the following year, insects damaged or destroyed 64% of northern red oak seedlings and 63% of germinating acorns (Galford *et al.* 1991a). Of the 1800 acorns used in that study, only 13% produced healthy seedlings.

The purpose of the present study was to determine if the same degree of damage could be expected on highly productive sites in West Virginia and to further determine if insect infestation was the cause of diminished seedling vigor on the less successful site. The effect of acorn burial on insect infestation was also examined.

METHODS

Study Site

This study was conducted on the West Virginia University Forest located in north-central West Virginia along the westernmost range of the Allegheny Mountains. This 7600-acre experimental forest is part of the Coopers Rock State Forest which straddles Interstate 68 in Monongalia and Preston Counties.

Two study sites were selected. The first site was located in the Lick Run watershed of the Forest where cove hardwood stands were selected on a northeast-facing slope. This study area was characterized by very high site indices for northern red oak (81 to 97), an abundance of mature yellow-poplar and northern red oak, and a moderate ground cover of herbaceous and woody vegetation. The SAF cover type was yellow-poplar--white oak--northern red oak (Eyre 1980). This site was located on a mid-slope position with an average slope of 12% (McNeel 1993). The soils are Dekalb stony sandy loams (Baur 1959). This study area will henceforth be referred to as the Lick Run site.

The second site was located in the Glade Run watershed of the Forest. The stands selected in this area were located on a drier northwest-facing slope and were characterized by lower site indices for northern red oak (68 to 73), a relative absence of yellow-poplar, a significant component of mature black cherry, and sparse ground cover. The SAF cover type on this site was also yellow-poplar--white oak--northern red oak (Eyre 1980). This site was located on a higher mid-slope position with an average slope of 17% (McNeel 1993). Soils were Dekalb stony sandy loams; however, the A horizons were thinner and contained more stone than those on the Lick Run site. This study area will henceforth be referred to as the Glade Run site.

Plot Establishment

Paired square plots of 0.5 acre were established on each site in May and June of 1990. The plots were isolated by a buffer strip on each side. Due to constraints imposed by a concurrent study, three pairs of plots were located on the Lick Run site and only two pairs were located on the Glade Run site. Each plot was divided into quarters and a mature (> 12 inches dbh), mast-producing, dominant or codominant northern red oak was located within each quarter. Therefore, a total of 40 trees (4 on each plot) were selected for study.

A cluster of three 1-foot square quadrats was randomly located within the area under the crown of each study tree. One of the quadrats in each cluster was fully enclosed with 0.5-inch mesh hardware cloth to exclude all potential mammalian and avian predators. These exclosures were constructed with mesh bottoms to prevent predators from entering from below and served as controls in the small mammal study. The tops of the exclosures were hinged so that they could be opened for observation. The remaining two quadrats were fully or partially accessible to small rodents and other mammalian predators and were not used in the present study.

Acorn Collection and Preparation

In 1990 and 1991, northern red oak acorns were collected off- site from a variety of locations in West Virginia and western Maryland from mid-September to late October of each year. The acorns were thoroughly mixed and then tested for viability at least 3 times over a period of 2 weeks using the float method (Korstian 1927, Schopmeyer 1974). Acorns that floated in water, had caps attached (indicating premature absiccsion), were damaged or discolored, or had insect exit holes or ovipositor scars were discarded. All acorns were stored in an unheated root cellar until use.

In the final year of study, acorns were collected on the study area. In mid-October 1993, all acorns, regardless of apparent condition, were manually collected from the ground under the crown of each of the 40 study trees. Acorns collected under the 4 study trees on each plot were then combined to provide a representative sample of acorns produced on each plot. These acorns were floated at least 3 times during a 2-week period and rejected acorns were set aside for use in another portion of the study.

Field Survival and Germination

In early November of 1990, 1991, and 1993, 9 apparently sound, undamaged acorns were placed on each quadrat in a square grid pattern. Nine acorns per quadrat was determined to be a reasonable approximation of that naturally occurring on a 1-foot square area (Gysel 1957). The acorns were pressed slightly into the humus to prevent them from rolling but were not buried. To assess the effect of burial on acorn survival, additional acorns were planted one inch deep in mineral soil on each quadrat. In 1990, only 3 acorns were buried; however, in 1991 and 1993, 9 acorns were buried. Therefore, there were a total of 12 acorns sown on each quadrat in 1990, 9 surface-sown and 3 buried, for a total of 60 acorns on each plot. In 1991 and 1993, a total of 18 acorns were sown on each quadrat, half on the surface and half buried, for a total of 72 acorns per plot. Leaf litter was placed over the acorns and the exclosures were wired shut and left undisturbed until the following spring. Around May 31 of each year, the exclosures were opened and the vigor of each seedling was rated 0 to 4 (Table 1).

Table 1. Seedling vigor rating descriptions

Rating	Description
0	Did not germinate
1	Germinated but weak root system, root or shoot dieback; not likely to survive
2	Strong root system, shoot small or just forming, generally earlier in development than seedlings with a rating of 3 or 4
3	Strong root system, no true leaves but strong healthy shoot
4	Same as a rating 3 seedling but at least one set of true leaves present

In 1994, any failed acorns and the seedlings with attached cotyledons were dissected with anvil shears. Insect damage or presence, disease, and apparent desiccation were recorded. In addition, the cotyledons were quartered and percent damage was estimated. All larvae present were collected and preserved in 80% ethanol for later identification.

RESULTS

Seedling Vigor in All Three Years of Study

Mean seedling vigor on each quadrat was calculated by year and treatment. Overall means by watershed are shown in Table 2. Analysis of variance revealed that mean vigor was significantly higher on the Lick Run site in all 3 years of study ($F = 22.60$ $Pr > F = 0.0001$),. In addition, year ($F = 21.16$, $Pr > F = 0.0001$) and treatment ($F = 29.21$, $Pr > F = 0.0001$) were found by the main effects model to significantly affect seedling vigor; however, there was also a significant year by treatment interaction ($F=14.18$, $Pr > F = 0.0001$). Examination of the interaction using additional ANOVA's and t-tests indicated that treatment was only significant in 1992 ($T = 6.16$, $Pr > |T| = 0.0001$). In that year, seedlings produced by buried acorns on both sites were more vigorous than those produced by surface-sown acorns. Among just the buried acorns, there was no significant difference between seedling vigor in 1992 and 1994, although vigor in both of these years was significantly higher than that observed in 1991 ($F = 15.04$, $Pr > F = 0.0001$). Among just surface-sown acorns, there was significantly higher vigor in 1994 than in either 1991 or 1992 ($F=16.77$, $Pr > F = 0.0001$). This was apparent on both sites but was more important on the Glade Run watershed. Average vigor in 1991 and 1992 on this site was 1.3 and 1.7, respectively, indicating that very few acorns were able to produce viable seedlings in either year. Seedling vigor rose to 2.6 in 1994, suggesting that the average seedling was healthy, well-developed, and capable of surviving. This was quite apparent in the field, as quadrats and entire plots that did not produce a single seedling in the previous years of study were found to be highly successful in 1994. On the Lick Run site, average vigor in 1991 was 2.3, in 1992 was 2.0, and in 1994 was 3.0. Healthy, viable seedlings were produced in all 3 years on this site; those produced in 1994 were exceptionally vigorous.

Table 2. Mean seedling vigor by year, watershed, and treatment.

Year	Site	Treatment	No. quadrats	Mean Vigor	Standard Error
1991	Lick Run	Surface	24	2.3	0.18
		Buried	24	2.3	0.17
	Glade Run	Surface	16	1.3	0.12
		Buried	16	2.0	0.24
1992	Lick Run	Surface	24	2.0	0.18
		Buried	24	3.3	0.15
	Glade Run	Surface	16	1.7	0.24
		Buried	16	2.8	0.25
1994	Lick Run	Surface	24	3.0	0.18
		Buried	24	3.0	0.11
	Glade Run	Surface	16	2.6	0.11
		Buried	16	2.7	0.15

Overall Insect-Infestation Rates on the Two Sites in 1994

In 1994, 56.9% of the 216 surface-sown acorns on the Lick Run site were sound. Twenty-seven percent had been at least partially damaged by insects, 5.6% were diseased, and the remainder exhibited damage due to root or shoot dieback (3.4%), desiccation (0.4%), surface mold (0.4%), or unknown causes (0.4%) (Figure 1). Most of the insect damage was caused by *Conotrachelus* weevils, although *Valentinia* moth larvae, and very small numbers of nitidulid sap beetles and fly maggots were identified (Figure 1). In contrast, 49.1% of the surface-sown acorns on the Glade Run site were sound, 24.8% were insect damaged, 10.3% were affected by root and shoot dieback, 8.5% were diseased and 1.8% were desiccated. Again, *Conotrachelus* weevil larvae were responsible for most of the insect damage (Figure 1). In general, buried acorns fared much better. On both sites, approximately 76% of the buried acorns were undamaged and approximately 10% were insect damaged. Root and shoot dieback affected 1.7% of the germinating acorns on the Lick Run site and 2.1% on the Glade Run site. Just under 9% of the acorns on the Glade Run site and 5.3% of those on the Lick Run site appeared to be diseased; however, the isolation of pathogens was not attempted (Figure 1).

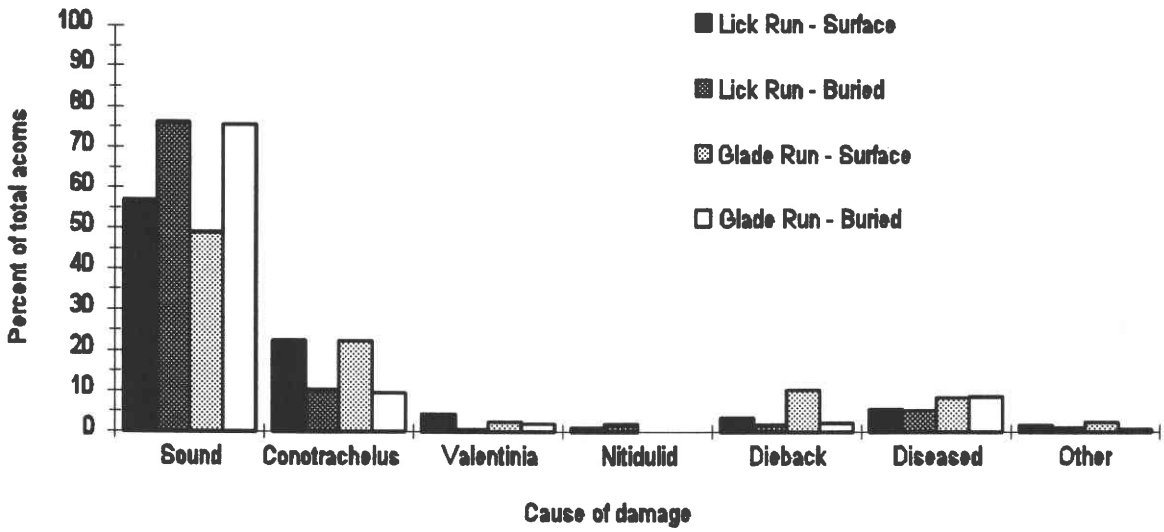

Figure 1. Total percentages of sound and damaged acorns by site, treatment, and damaging agent. Note: these are pooled percentages and are not based on the quadrat-level means used in the analysis of variance.

Comparison of Insect Infestation Between Sites in 1994

To allow comparison of insect infestation between the sites, percent infestation by any insect or combination of insects was calculated by treatment on each quadrat. Acorns infested by more than one insect were counted only once in this analysis and those infested by fly maggots were excluded as they appeared to have succumbed to disease before infestation. Analysis of variance on these data revealed that insect infestation did not vary by site ($F = 0.31$, $Pr > F = 0.5824$); however, there was a significant treatment effect ($F = 4.69$, $Pr > F = 0.0335$) with no site by treatment interaction. On average, surface-sown acorns were about twice as likely to be infested regardless of site (Table 3). Infestation rates on individual quadrats within the sites were quite variable.

Table 3. Mean percent of acorns infested by insects on each plot in 1994 by site and treatment.

Site	Teatment	No. plots	Mean infestation rate (%)	Range on individual quadrats (%)	Standard error. (%)
Lick Run	Buried	6	11.6	0-44	1.2
	Surface-sown	6	24.1	0-100	7.4
Glade Run	Buried	4	10.4	0-44	2.3
	Surface-sown	4	19.4	0-89	7.4

Cotyledon Damage and Seedling Vigor in 1994

Each acorn was classified as sound, insect-infested, or damaged by other agents. Mean vigor and mean percent cotyledon damage were then calculated by group and treatment on each quadrat. Analysis of variance indicated that there were significant differences in vigor among the groups ($F = 139.58$, $Pr > F = 0.0001$). Seedlings from sound acorns were found by Fisher's LSD comparisons to be significantly more vigorous than those produced by damaged acorns, regardless of damaging agent, site, or treatment. In addition, acorns damaged by insects produced significantly more vigorous seedlings than those damaged by other agents such as desiccation and disease. Differences in seedling vigor are obvious in the pooled data (Figure 2).

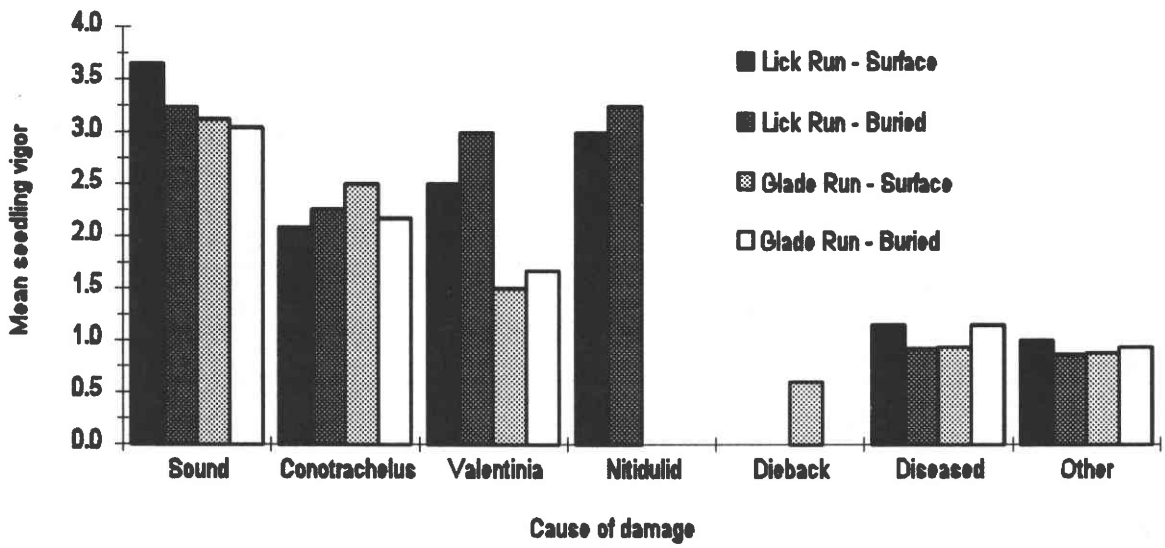

Figure 2. Average vigor of sound and damaged acorns by site, treatment, and damaging agent. Note: these means are calculated using all of the data pooled by site and treatment and are not based on the quadrat-level means used in the analysis of variance.

An additional ANOVA was conducted to determine if desiccation and disease resulted in more extensive cotyledon damage than did insects. This analysis, which excluded sound acorns, revealed that percent cotyledon damage was significantly higher in acorns damaged by agents other than insects ($F = 19.24$, $Pr > F = 0.0001$). Cotyledon damage due to insects averaged 14.8% (SE=2.5%) while that caused by other agents averaged 38.9% (SE=5.0%). Site and treatment had no significant impact on the degree of cotyledon damage observed in either group of acorns.

These results suggested that percent cotyledon damage could be used to improve the original analysis of variance on vigor. Therefore, percent cotyledon damage by all agents was used as a covariate with site and treatment in an analysis of covariance. This analysis demonstrated that percent cotyledon damage was significantly related to seedling vigor ($F = 8.06$, $Pr > F = 0.0060$) and that it explained some of the variation within site (Figure 3). There was no significant treatment effect.

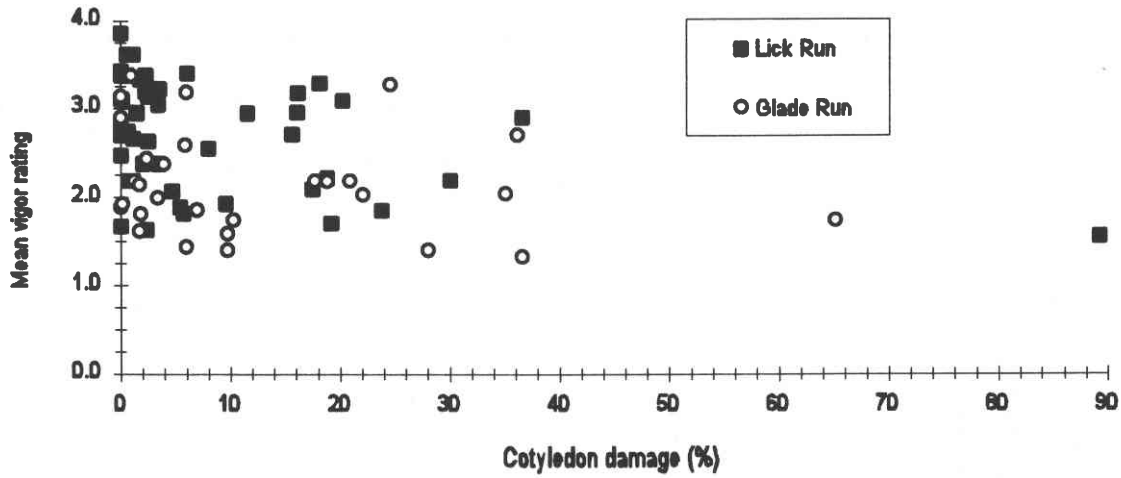

Figure 3. Mean seedling vigor versus mean cotyledon damage by site. Each observation is based on a single quadrat.

DISCUSSION

Seedling vigor was consistently better on the Lick Run site regardless of the year of study, indicating that the additional data pertaining to acorn condition and damage collected in 1994 could be expected to have at least some relevance to the results from the previous two years. However, vigor was found to be significantly higher among surface-sown acorns on both sites in 1994, indicating that growing conditions were not consistent across the 3 years. This annual variation, likely weather related, cannot be accounted for with a single year of study. Therefore, it is unlikely that the cause of the widespread failure observed on the Glade Run site in 1991 and 1992 can be determined here. However, initial data on the incidence of insects in germinating northern red oak acorns can be reported for north-central West Virginia. In addition, the effects of these insects on seedling vigor in a year of high reproductive success can be examined.

The results of the 1994 study indicate that all three of the insects noted to be serious pests of germinating northern red oak acorns in Ohio (Galford *et al.* 1988) and Pennsylvania (Galford *et al.* 1991a) were also present in north-central West Virginia. In concurrence with the results of both of these studies, *Conotrachelus* weevils were the most serious pest of germinating acorns. The *Valentinia* moth was much less prevalent. In addition, the incidence of moth larvae was not uniform across the study area. This species appeared sporadically in individual quadrats and usually infested several acorns in each affected quadrat. *Valentinia* moth larvae have not been previously reported to infest buried acorns; however, in this study, small numbers were found feeding in buried acorns on both sites (Figure 1).

The nitidulid sap beetle was almost absent from the study area. Cotyledon damage due to the feeding and breeding activities of adult sap beetles was extremely minor in the few acorns infested on the Lick Run site. As suggested by Galford et al. (1991a), sap beetles may not have been active by late May when the acorns and seedlings were collected from the field. Their possible impact later in the growing season deserves more study, as sap beetles were found capable of widespread seedling damage in Ohio (Galford *et al.* 1988).

Overall, the incidence of acorn pests in germinating northern red oak acorns was much lower than that reported in the Ohio and Pennsylvania studies. However, weather conditions were quite unusual in West Virginia during the winter of 1993-94. A relatively wet fall was followed by an extremely cold winter with much greater than normal snowfall. A hard crusty snowpack remained on the study area almost constantly throughout the winter. Spring came late and hard frosts were still occurring in mid- to late May. However, by May 30-31, when the exclosures were opened, daytime temperatures had exceeded 70◦F for several days and the ground was beginning to warm. In contrast, the winters of 1991 and 1992 were very mild with sporadic snowfall and only patchy accumulations of snow on the study area. In both of these years there was unseasonably warm weather in late February when temperatures ranged from 60◦F to 70◦F for a week to 10 days.

In 1991 and 1992, acorns in the study area began germinating by late February; however, in 1994, germination did not begin until mid-April. Galford *et al.* (1991a) found larval acorn moths in Pennsylvania to become active in germinating acorns by mid-February, weevil larvae to commence feeding by mid-March, and sap beetles to begin activity in late May. The hard winter and late spring in West Virginia in 1994 may have delayed emergence of the insects and it is possible that insects had not begun to emerge until shortly before the exclosures were opened. In addition, the late germination of the acorn crop may have resulted in a lack of suitable habitat for insects that did emerge early. In the case of the sap beetle, it appears that breeding was delayed. The few acorns that were infested by nitidulids showed only minimal oviposition damage and no larvae were present. Had the study been continued into the summer, sap beetles, and perhaps late-emerging weevils and acorn moths, may have had more of an impact.

In addition, Galford *et al.* (1991a) indicate that most acorns infested by insects in central Pennsylvania were completely destroyed and were incapable of producing viable seedlings. This was not the case in the present study. Although insect infestation and resultant cotyledon damage did reduce seedling vigor, they were not severe enough in most cases to cause lack of seedling success. The fact that insect infestation on average resulted in only 14.8% cotyledon damage and that larvae were still present in most infested cotyledons indicates that the adult insects may have emerged too late to have more than a minimal impact on the rapidly growing seedlings.

Of even more significance is the lack of difference in percent insect infestation between the sites, even though the seedlings on the Glade Run site were again found to be less vigorous. It was initially hypothesized that the lack of seedling success and vigor on the Glade Run site was due in part to insect infestation; however, at least in 1994, this was not the case. As mentioned previously, the widespread reproductive failure observed on the Glade Run site in 1991 and 1992 was absent in 1994. Glade Run acorns produced healthy seedlings in 1994, although they were in general smaller and less well-developed than those produced on the Lick Run site. This appeared to be a microsite effect, perhaps caused by the warmer, drier northwest aspect or more exposed slope position of the Glade Run site.

Several corollaries can now be put forth to explain the acorn failure on the Glade Run site. First, if insect infestation and effect is assumed to have been similar on both sites in the previous two years of study, then some other factor must have been responsible for the lack of seedling production. This factor may have been desiccation as the Glade Run site is drier and more exposed than the Lick Run site. Or the acorns may have become diseased. Winston (1956) and Dorsey *et al.*(1962) identified several bacterial and fungal agents that may affect acorn viability even in the absence of insect infestation. Overall, desiccation and disease were found to be slightly more prevalent on the Glade Run site (Figure 1), although there were not sufficient numbers of desiccated or diseased acorns to allow statistical analysis. Both were found in this study to cause almost complete cotyledon destruction and loss of acorn viability. No seedlings were produced by desiccated acorns and the seedlings produced by diseased acorns exhibited a notable lack of vigor (Figure 2). Since the winters of 1991 and 1992 were relatively dry with a lack of snowpack, widespread desiccation on the Glade Run site is a particularly good possibility.

Conversely, the relatively minimal effects of insects in 1994 may have been an anomaly. Populations may usually be higher on one or both sites. If this was the case in 1991 and 1992, feeding in the cotyledons may have been more extensive and in itself may have caused more notable lack of vigor or may have predisposed acorns to failure due to drying or disease. Several more years of study on a greater number of sites would provide additional insight.

SUMMARY

It has been established that *Conotrachelus* weevils, the acorn moth,*Valentinia glandulella*, and the nitidulid sap beetle, *Stelidota octomaculata*, infest germinating northern red oak acorns in north-central West Virginia. The data from a single year of study indicate that these insects were much less prevalent and much less damaging on this study site than they were on sites in Ohio and Pennsylvania. However, the severity of the winter of 1994 and the late spring frosts may have resulted in delayed emergence.

It is suspected that agents other than insects were responsible for the acorn failure and lack of vigor noted in 1991 and 1992. Disease and desiccation, when they did occur, almost invariably caused acorn failure in 1994. These factors may have had more widespread effects in the previous years of study and warrant additional study in future years.

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LOGGING SAFETY IN FOREST MANAGEMENT EDUCATION

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Abstract: Forest management degree programs prepare students for careers in forestry by teaching a combination of biological sciences (e.g., silvics and genetics) and business management (e.g., forest policy and timber valuation). During a 4-year degree program, students learn the impact of interest rates, equipment costs, and environmental policies on forest management and silvicultural decisions. However, little consideration is given to worker safety and health and its impact on forest management. This paper illustrates the ethical and economic importance of logging safety to forest managers and advocates incorporating safety issues into existing forestry courses. An appendix provides examples of safety and health information that can be integrated into forest policy and forest economics courses.

INTRODUCTION

Four-year forest management programs are designed to prepare students for professional careers in management of natural resources, including the business aspects of finance and personnel management. Students are exposed to a diverse curricula, ranging from the biological sciences to accounting principles. This broad program emphasis is encouraged through the *Society of American Foresters (SAF) Accreditation Standard for Forestry Program Mission, Goals and Objectives* (Society of American Foresters 1994a), in which the interdisciplinary nature and service orientation of the forestry profession are emphasized. The SAF places importance on management training -- two of the four areas of study that must be adequately represented in the forestry curriculum are Management of Forest Resources (including forest engineering, harvesting and utilization) and Forest Resource Policy and Administration (including policy development, administration, and personnel management). Forestry curricula must "provide students with an understanding of the social, cultural, political, legal, economic, institutional, and historical influences on forestry" (Society of American Foresters 1994a).

Forestry programs traditionally meet this goal of providing students with an understanding of socioeconomic influences on forestry through policy and management courses. External influences on forestry typically addressed in these courses include organizations, cultural resources, environmental concerns, pollution control, military policy, taxation, and international trade policy (see tables 1 and 2). One important topic, occupational safety and health, is not typically addressed in these courses.

WHY TEACH SAFETY AND HEALTH IN FORESTRY PROGRAMS?

Worker safety concepts should be integrated into forestry education for several reasons. First, logging is the primary tool for accomplishing forest management objectives on the ground. Logging has long been considered a dangerous industry, and death certificate data indicate that the rate of work-related fatalities in the logging industry is extremely high (Myers and Fosbroke 1994), with regional rates ranging from a low of 87.0 deaths per 100,000 workers in the southeast to 394.3 deaths per 100,000 in the central hardwood region. For comparison, the fatality rate across all industries is 7.0 per 100,000 workers, and the rate in the manufacturing sector (of which logging is a part) is 4.4 per 100,000 (Jenkins et al. 1993). The logging industry has also been identified as having a high rate of nonfatal injuries with a lost workday case rate of 10.7 per 100 full-time workers in 1990 compared to a private industry average of

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Table 1. Examples of Forest Management Influencing Legislation Taught in Forest Policy Courses[1].

Legislation	Date	Notes
Magna Carta	1215	King's ownership of wildlife in trust for all people
King's Broad Arrow	1691	Reserved mast trees for the British navy
Lacey Act	1900	Prohibited interstate trade of wildlife illegally taken
Antiquities Act	1906	Preservation of cultural artifacts
Week's Law	1911	Watershed protection; Fire protection
Forest Cons. and Taxation	1949	Tax incentives for certain forest management practices
Land and Water Cons. Fund	1964	Public purchase of private land for recreational use
Wilderness Act	1964	Established the Wilderness Area system
Nat'l Historic Preservation Act	1966	Assessment of activity impact on historic sites
Clean Air Act and Amendment	1970, 1973	Designation of air quality classes, scenic visibility
Fed. Water Pollution Control Act	1972	Stop discharges, attain fishable/swimmable waters
Endangered Species Act	1973	Protection of endangered plants/animals, critical habitat
Eastern Wilderness Act	1974	Extended 1964 act to eastern U.S.; added 16 new areas

[1] Sources: Cubbage, O'Laughlin, and Bullock (1993); Dana and Fairfax (1980)

Table 2. Examples of Organizations and Events that Have Influenced Forest Policy[1].

American Assoc. for the Advancement of Science	Monongahela National Forest Controversy
American Forestry Association	National Academy of Sciences
Bitterroot National Forest Controversy	National Trail System
Civilian Conservation Corps	Native Alaskan Lands Settlement
Council on Environmental Quality	Office of Management and Budget
Environmental Protection Agency	Sand County Almanac publication
Farm Abandonment & Westward Expansion	Sierra Club
Hetch Hetchy	Soil Conservation Service
Friends of the Earth	Society to Protect New Hampshire Forests
Izzak Walton League	Tellico Dam

[1] Sources: Cubbage, O'Laughlin, and Bullock (1993); Dana and Fairfax (1980)

3.9 and a manufacturing industry rate of 5.3 per 100 full-time workers (U.S. Department of Labor 1992). Though there have been tremendous advances in harvesting technology over the past 30 years, the logging industry has seen little improvement in injury statistics (McCormack 1963, Myers and Fosbroke 1994).

The second reason for teaching forestry students about worker safety is that forest management decisions directly affect logging safety. There are many examples. The decision to leave den trees may raise logger's risk of being hit by a falling snag. The selection of a residual stand density influences the probability of a logger being injured in the hit-tree reaction type of incident described by Peters (1991). Decisions about the length of a cutting contract may influence the amount of time loggers have to complete the job safely. Even forestry graduates that do not make harvesting decisions may be employed by industry in supervisory positions where they are responsible for other employees. Under the provisions of the 1970 Occupational Safety and Health (OSH) Act (Public Law 91-596), employers are responsible for assuring safe and healthful working conditions for all employees (see Appendix).

Graduating forestry students need to be aware of their responsibility for understanding and following regulations that apply to their jobs.

Finally, teaching safety to forestry students is an ethical obligation of any forestry program. The "Code of Ethics" set forth by the Society of American Foresters (Society of American Foresters 1992), states in its preamble that compliance with the SAF canons "ensures just and honorable professional and human relationships, mutual confidence and respect, and competent service to society." Professional foresters are morally and ethically bound to ensure that their decisions and actions do not jeopardize the safety of others. Graduating forestry students need to be aware of how their future management decisions can affect the safety and health of other workers.

CURRENT STATUS OF SAFETY IN FORESTRY PROGRAMS

Programs that train students to become professional foresters often devote less program time to safety than do programs that train forestry technicians. This is partially exhibited by differences in SAF requirements for accreditation of professional programs and recognition of technical programs. Though current *Guidelines for Accreditation of Educational Programs in Professional Forestry* (Society of American Foresters 1994a) do require coverage of personnel management and do require that facilities provide an environment that is safe, healthful, and conducive to learning, the guidelines make no other mention of safety. In contrast, the *Standards and Procedures for Recognizing Educational Programs in Forest Technology* (Society of American Foresters 1994b) have a specific curriculum requirement for "Woods Safety," which is to include basic first aid, identification of hazards, hand and power tool safety, and pesticide safety.

In May of 1993, the authors sent a request to the 45 professional degree programs accredited by the Society of American Foresters. Each program was asked to provide a college catalog describing their program, course syllabi of management and policy related courses, and a copy of the most recent self-evaluation report. These materials were reviewed to determine how prominent a role safety topics played in the way courses and programs are described to prospective students. Thirty-one forestry programs provided materials (26 schools sent catalogs, 22 sent course syllabi, and 18 sent self-evaluation reports). Only seventeen programs referenced any safety concept within the reviewed material and these references were generally limited to the harvesting course. Based on this review, most professional forestry degree programs do not currently emphasize safety issues in their descriptions of course content.

Though the numbers above suggest that safety is not an emphasis area in forest management programs, most programs do have some safety components. Frequently. the strongest safety component of 4-year programs is in the summer forestry camps. Here, a few days are spent teaching the technical aspects of harvesting, road construction and chain saw use. Most 4-year programs also touch briefly on safety issues in a variety of courses. For example, logging hazards are discussed in harvesting classes, though the impact of these hazards in terms of lives lost and lost work time injuries is typically not. Pesticide labelling is covered in planting, silviculture, and entomology courses. Ecology and wildlife management courses also discuss pesticide safety, though usually from the environmental contamination standpoint. Each year, a few students gain practical experience in safe woods operations from forest managers while working for the school at the university woodlot or forest. Other attempts are made to instill a consciousness of safety among forestry students, including the requirement that all students wear hard hats and sturdy boots during field labs and industry tours.

What is Missing?

Professional foresters need to make rational decisions based on an understanding of safety principles and an awareness of safety issues that affect forest (and any other management) operations. Forestry programs should prepare students for challenges they will face by at least making them aware of: the safety record of the logging, sawmilling, and paper manufacturing industries; the existence of safety regulations, standards, and policies; the identification and control of hazards; the management of safety; the methods of preventing injuries; the impact of safety programs on insurance premiums; the concepts of insurance and shared risk; the impact of injuries on operating

costs and employee morale; the responsibility and liability of decision makers and managers; and the influence of management decisions on worker safety.

When the National Research Council (NRC) surveyed academic and nonacademic forestry groups to identify needs in forestry education (National Research Council 1990), respondents suggested a need to "move from technician-type courses to more analytical, decision-making, conflict resolution types of courses." Expansion beyond technical safety topics (e.g., first aid and tool safety) toward the legal, economic, and moral aspects of safety and health fits well with this NRC recommendation. This expanded coverage of safety and health issues will provide future foresters with the analytical tools they will need to evaluate management decisions.

INTEGRATING SAFETY CONCEPTS INTO EXISTING COURSES

Integration of safety and health concepts into forestry education **will not** require a new course, "Safety in Forest Operations." Forestry curricula already contain a full complement of course requirements; any new course would require elimination of an existing course. The ultimate goal should be to teach future foresters to consider the safety implication of all decisions. To do this, safety thinking needs to be incorporated into many courses, wherever a safety issue is relevant. This need not be difficult because most safety issues relate in some way to concepts that are taught in existing courses.

For example, forest policy courses traditionally include tax laws, environmental regulations, recreational development programs, and resource funding programs because of the influence of these regulations and programs on forest management decisions. The inclusion of safety and health regulations, the regulatory process, and safety research organizations fits into the concept of teaching policies and programs which influence forestry (see tables 3 and 4 for selected examples). Additional topics that could be incorporated into forest policy courses include the regulatory process (including the OSH Act's provision for public comment during standards development), owner liability, third party tort, and specific Occupational Safety and Health Administration (OSHA) and national consensus standards related to forestry operations.

In addition to the above forest policy examples, other topics can be integrated into the existing curricula. Introductory forestry courses should cover sources of injury and illness information, injury statistics specific to forest industries and occupations, and the importance of safety. Timber harvesting courses should describe specific hazards associated with logging and recommended ways of minimizing hazards, especially machine guarding, rollover protection, and the identification and safe felling of hazardous trees. Forest Management courses should emphasize the role of managers in ensuring safe and healthful working conditions. Other safety topics for forest management include safety language in logging contracts, hazard communication, recordkeeping, accident investigation, and safety management programs. Silviculture courses should explain the effect of cutting method, leave tree selection, and cutting cycle on logging safety.

Specific information on the Occupational Safety and Health Act, the history of workers' compensation, and the incorporation of workers' compensation costs in machine cost calculations are provided in the appendix. This material provides details of the type of information that could be integrated into forest policy and forest economics courses. Other examples of incorporating safety issues into an existing curriculum are described by Taylor et al. (1994).

Table 3. Examples of Safety Legislation, Standards, and Policy Impacting Forest Management[1].

Legislation	Date	Notes
Federal Workers' Compensation	1908	Federal law providing compensation for federal workers injured on the job
State Workers' Compensation	1911	Wisconsin and Maryland pass laws similar to federal workers' compensation law
Walsh-Healy Act	1938	Specified working conditions for employees working for private companies under contract to the U.S. government
Fair Labor Standards Act	1938	Prohibited children under 18 years old from working in hazardous occupations (including logging)
Occupational Safety and Health Act	1970	Created regulatory and research agencies and a process for development and enforcement of OSH standards
Safety Requirements for Pulpwood Log.	1971	ANSI national consensus standard
OSHA Pulpwood Log. Standard 1910.266	1971	OSHA adopted ANSI consensus standard
NIOSH criteria document on logging	1976	Recommendation for an occupational standard on logging from felling to first haul
Safety Requirements in Logging	1978	ANSI consensus standard extending prior pulpwood requirements to all logging
ANSI withdraws Safety Req. in Logging	1984	ANSI committee fails to renew standard
OSHA Logging Standard	1994	Proposed in 1989, OSHA promulgated a single standard for the logging industry on October 12, 1994.

[1]Sources: DeReamer 1980; Hammer 1989; Kavianian and Wentz (1990); Public Law 91-596 (1970); Readers' Digest (1975); U.S. Department of Labor (1989, 1994)

Table 4. Organizations Influential in the Establishment of Safety Policy Affecting Forest Management[1].

Organization	Notes
American National Standards Institute	Develops national consensus standards
American Pulpwood Association	Produces logging safety materials; facilitates safety
American Society of Agricultural Engineers	Forest Operations Safety Committee provides a forum for harvesting safety research
American Society of Safety Engineers	Established an independent board to certify safety professionals
International Labor Organization	Established as part of League of Nations to improve working conditions around the world; major contributor to woods worker training and international recommendations
National Fire Protection Association	Develops national consensus fire protection standards
Nat'l Institute for Occupational Safety and Health	Conducts occupational safety and health research, trains safety and health professionals
National Safety Council	Private, non-profit organization working in traffic, home, recreational, and occupational safety
Occupational Safety and Health Administration	Administers the OSH Act, promulgates regulations, and provides state training grants

[1]Sources: DeReamer 1980; Hammer 1989; Kavianian and Wentz (1990); Readers' Digest (1975)

SUMMARY

Logging is well known as a hazardous industry, both in terms of fatal and nonfatal outcomes. Recent statistics, when compared to statistics from 30 years ago, indicate that the industry's injury record is not improving (Myers and Fosbroke 1994). These facts led OSHA to promulgate a new standard for logging operations (U.S. Department of Labor 1994). The new mandatory standard has direct implications on the way logging is conducted. Since logging is the primary tool for accomplishing natural resource management objectives, this standard also impacts forest management decisions. Therefore, graduate foresters increasingly will need to be aware of occupational safety and health concepts. Professional forestry degree programs have traditionally provided students with a wide range of background; however, worker safety and health typically is not a major focus of class instruction. Integrating important occupational safety and health issues into existing curricula can be done, but will take a concerted effort on the part of faculty to expand existing courses into new subject areas. This paper demonstrates a few examples of how safety topics can be covered in forestry courses. The appendix contains additional examples. It is hoped that faculty will consider these examples, look at their existing courses, and ask, "Are there important safety issues that I should incorporate into my courses?"

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APPENDIX

SELECTED SAFETY TOPICS FOR FOREST POLICY AND FOREST ECONOMICS COURSES

The Occupational Safety and Health Act

The most sweeping U.S. law regarding worker safety and health was passed by Congress on December 17, 1970. The preamble to the Occupational Safety and Health (OSH) Act (Public Law 91-596) includes a promise to "assure safe and healthful working conditions for working men and women." Section 5 of the Act defines (in broad terms) the duties of employers and employees with respect to occupational safety and health. An employer has the duty under Section 5(a)(1) "to furnish to each of his employees employment and a place of employment which are free from recognized hazards that are causing, or are likely to cause death or serious physical harm to his employees." An employer also has the duty to "comply with occupational safety and health standards promulgated under this Act." This section of the OSH Act **clearly** places on forest managers the responsibility of knowing what standards are applicable to forestry operations. An employee also has a duty under Section 5(b) to "comply with occupational and safety and health standards and all rules, regulations, and orders pursuant to this Act which are applicable to his own actions and conduct."

Within 2 years of the effective date of the Act, the Secretary of Labor was to "promulgate as an occupational safety and health standard any national consensus standard, and any Federal standard, unless he felt that the promulgation of such a standard would not result in improved safety and health for specifically designated employees." Several existing Occupational Safety and Health Administration (OSHA) standards resulted from adoption of national consensus standards developed by organizations such as the National Fire Protection Association and the American National Standards Institute (ANSI) (Kavianian and Wentz 1990). The original pulpwood standard (U.S. Department of Labor 1993a) is an example of a national consensus standard that was developed by ANSI and promulgated by OSHA into the new OSHA standards in 1971 (U.S. Department of Labor 1989). OSHA continues to review consensus standards during regulatory review and development.

OSHA has promulgated general industry standards and industry-specific standards. General industry standards apply to all industries. Examples include electrical, machinery and machine guarding, hazardous materials communication, fire protection (Kavianian and Wentz 1990), and lockout/tagout standards (U.S. Department of Labor 1993b), and recordkeeping regulations (U.S. Department of Labor 1986). Other standards apply to a specific industry. The logging standard that OSHA recently promulgated[2] (U.S. Department of Labor 1994a) is an example of an industry-specific standard.

Several aspects of the Occupational Safety and Health Act and the standards and regulations promulgated under the Act influence forestry operations[3]. The new logging standard, and the pulpwood standard that it replaced, obviously apply to the harvesting of trees. However, these are not the only factors of which forest managers need to be aware. The Act clearly was intended to hold employers responsible for the safety of their employees. OSHA has used the

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2 For a discussion of the new OSHA logging standard and what it means for forest management, see Myers and Fosbroke (1995). For a copy of the text of the new standard, contact: U.S Department of Labor, Occupational Safety and Health Administration, Office of Publications, Room N-3101, 200 Constitution Avenue, NW., Washington, D.C. 20210.

3 Examples of OSHA requirements are included to apprise the reader of the range of regulatory issues affecting forestry and logging employers. This is not a complete list, and a full description of the specifics of these requirements is beyond the scope of this paper. The reader should also be aware that OSHA is continually revising its standards and regulations; therefore, affected readers should remain cognizant of the status of applicable OSHA regulations by contacting their Regional OSHA Area Office.

provisions of the General Duty Clause [Section 5(a)(1)] to cite employers that allowed a recognized hazard to persist at a site of employment, even when no specific OSHA standard or regulation had been violated. Additionally, there are many general industry standards that forestry employers are also responsible for meeting, even though logging and forestry are not specifically mentioned in these standards. For example, a sawmill owner may be held responsible for failing to maintain protective guarding on machinery.

Any company employing 11 or more employees is required to maintain an annual log of employees' occupational injuries and illnesses. These logs must be posted annually for public disclosure and provided to OSHA upon request (U.S. Department of Labor 1986). OSHA must also be notified of any fatality or incident resulting in the hospitalization of 3 or more employees within 8 hours of the incident, or when the employer became aware of the incident (U.S. Department of Labor 1994b).

Workers' Compensation

The development of the Workers' Compensation System in the United States is another topic relevant to courses in forest policy. Workers' compensation costs are a significant portion of overall labor costs (DeReamer 1980). Additionally, discussion of workers' compensation provides the opportunity to introduce students to the concepts of shared risk, of occupational injury and illness costs, and of risk management.

Workers' compensation is a system of insurance, initiated in the early 1900's, that is funded by employers to compensate workers who receive an injury or illness while working. Employer premiums are based on an experience-rating mechanism, such that employers with fewer occupational injuries and illnesses pay lower rates than employers with poor safety records (U.S. Chamber of Commerce 1984). Currently, all 50 states and the District of Columbia have workers' compensation laws. Each state administers its own law and these laws vary in terms of the employers covered, the definition of a compensable injury or illness, and the type of system (U.S. Chamber of Commerce 1984).

Several premises underlie the workers' compensation system (U.S. Chamber of Commerce 1984). First, industrial employers should assume costs of occupational disabilities without regard to any fault involved. Second, economic losses resulting from occupational disability are considered costs of production. Third, employers are relieved from lawsuits under common law involving negligence. The goal is that adherence to these principles promotes honest study of injury and illness causes rather than concealment of fault. As a result, future incidents and human suffering can be prevented. The experience-rating mechanism is intended to encourage employers' interest in safety and rehabilitation as a means of reducing costs.

Adding Workers' Compensation Cost to Machine Cost Calculation. Calculating the hourly machine cost of harvesting equipment entails a combination of time study and financial analysis, accounting for a projected production rate (e.g., cords/day), fixed costs (e.g., investment and tax costs), and variable costs (e.g., fuel and maintenance costs). Inclusion of workers' compensation premiums as a cost factor in machine cost calculations provides an appreciation for the effect of workers' compensation insurance on total operating expense. Because workers' compensation insurance in the logging industry is expensive and increasing rapidly (Lansky 1990, Longwell and Lynch 1990, Williams 1991, and Stevens and Giannetti 1992), premium costs should be considered in the financial analysis methods taught in forest economics classes.

An example of how to calculate the impact of workers' compensation on operating costs is provided on the next page. In this example, the hourly machine cost for a medium-sized cable-skidder is determined using a workers' compensation rate of $35.62 per $100.00 of payroll. Most of the calculations are based on the methodology laid out in a U.S. Department of Agriculture Forest Service pamphlet entitled, "How to Calculate Costs of Operating Logging Equipment" (Miyata date unk.). The exception is that a workers' compensation premium was added to the labor costs. All cost items have been updated to approximate realistic costs in 1994.

Example of Machine-cost Calculation for a Cable Skidder Using the Methodology of Miyata

BASIC ASSUMPTIONS		
Purchase Cost	$90,000	
Less: Tire	-9,500	
Initial Investment (P):	$80,500	
Salvage Value (S): 20% of P	$16,100	
Economic Life (N):	3yrs	
Schedule operating hours (SH):	2,000 hrs/yr	
Utilization	65 percent	
Productive Hours (PH):	1,340 hrs/yr	
FIXED COSTS		
Depreciation (D): (P-S)/N		$21,466/hr
Interest, Insurance Taxes (ITT):		9,445/yr
ITT=16%((P - S) (3+1)/(2)(3))=9445.33		
Yearly Fixed Cost (YFC):D+ITT		$30,911/yr
Hourly Fixed Cost (HFC):YFC/PH		$23.07/hr
OPERATING COSTS		
Maintenance and Repair (MR): 0.5(D)/PH		$8.01/hr
Fuel (F): 4 gal./hr at $1.00/gal.		$4.00/hr
Lubricants (L):		$1.90/hr
L=(((12 gal.)($4/hgal.)+138)/100 hurs) + ((50#)($1.03/#)/1340 hrs)=1.86+.04=$1.90/hr		
Tire (T): T=(1+0.15)($9,500)/3,000 hrs= $3.64/hr.		$3.64/hr
Hourly Operating Cost (HOC):		$17.55/hr
Hourly Machine Cost (HMC):HFC+HOC		$40.62/hr
LABOR COSTS		
Hourly Wage		$7.00/hr
Social Security @7.57%		$ .53/hr
Workers' Compensation 35.62%		$2.49/hr
Total Labor Cost (TLC):		$10.02/hr
Hourly Machine Cost (HMCL):HMC+TLC		$50.64/hr

A comparison of the proportion of the hourly machine cost for three different levels of workers' compensation premiums are shown in Table 5. The premiums are the low, mean, and high rates for the calendar year 1989 from a survey of state workers' compensation systems (American Pulpwood Association 1990). The cost a company pays for insuring a single logger against work-related injury or illness accounted for as little as 2.4 to as much as 10.1 percent of the hourly owning and operating cost of a cable skidder. Even at the mean compensation premium rate of $35.62 per $100.00 of payroll, workers' compensation is an important cost factor.

Table 5. The Effect of Different Workers' Compensation Premiums on the Hourly Machine Cost of a Cable-Skidder.

Premium Rate ($/$100 payroll)	Hourly Machine Cost ($/hr)	Percent (%)
17.08	49.35	2.4
35.62	50.64	4.9
76.90	53.53	10.1

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FOREST MANAGEMENT PRACTICES AND THE OCCUPATIONAL SAFETY AND HEALTH ADMINISTRATION LOGGING STANDARD

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Abstract: The Occupational Safety and Health Administration (OSHA) has established safety and health regulations for the logging industry. These new regulations move beyond the prior OSHA pulpwood harvesting standard by including sawtimber harvesting operations. Because logging is a major tool used by forest managers to meet silvicultural goals, managers must be aware of what the OSHA standard would mean to them. Many aspects of the new standard pertain to the training of logging employees, safe equipment operation, and safe harvesting techniques, but there are sections of the OSHA standard that impact forest management practices, especially for hardwood forests, in much the same way as environmental regulations. There is the potential for forest management practices to influence the safety of logging operations by considering how the stand will be harvested when selecting silvicultural treatments for a stand. Forest managers will also be responsible for balancing conflicts between environmental issues and OSHA regulations, such as the removal of hazardous trees from a stand, the placement of skid trails, or where logging is initiated in a stand. These and other issues are discussed to give forest managers a better appreciation of their role in making logging a safer industry.

INTRODUCTION

Logging is one of the most flexible tools available to the forest manager to meet the silvicultural, biological diversification, economic, and recreational goals developed for a forest stand. In reviewing such classical forestry texts as "The Practice of Silviculture" (Smith 1962) and "Regional Silviculture of the United States" (Barrett 1980), the majority of the topics directly, or indirectly involve harvesting methods that can be applied to a forest stand and the expected future results that these harvesting methods will produce in the forest stand. The use of silvicultural systems and logging to promote wildlife habitats is also a common practice (Yoakum and Dasmann 1969), as well as the use of specific silvicultural treatments and logging to enhance recreational uses of forest stands (Knudson 1980). Thus, natural resource management, especially silviculture, is highly dependent on logging.

Given this close relationship between forest management and logging, forest managers must be aware of what regulations impact the logging industry. Environmental regulations and public concerns on environmental issues related to logging are now commonly included in the decisions a forest manager makes in practicing silviculture in any forest stand (Cubbage et al. 1993). A new issue, which until recently has not been a concern for forest managers in the central hardwood region, is worker safety in the logging industry. This will assume more significance with the final promulgation of the Occupational Safety and Health Administration's (OSHA) Logging Operations Standard (U.S. Department of Labor 1989).

As with any new set of regulations, forest managers will need to know and understand the OSHA logging standards and how these regulations impact upon their management decisions. Equally important, forest managers should consider and assess how their management decisions impact upon logging safety.

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THE OSHA LOGGING OPERATIONS STANDARD

Background

Logging is one of the most hazardous industries in the United States from both a fatal and nonfatal injury perspective (U.S. Department of Labor 1984, Paulozzi 1987, Leigh 1988, Myers and Fosbroke 1994). In 1989, the logging industry had a work-related fatality rate of 193 deaths per 100,000 workers (U.S. Department of Health and Human Services 1994), a rate more than 34 times higher than the total private sector fatality rate of 5.6 deaths per 100,000 workers (Jenkins et al. 1993). The greatest risk for fatal injuries in the logging industry were in those regions that were associated predominantly with hardwood sawtimber harvesting (e.g., Central Hardwood and Eastern Mixed-Hardwood regions), with fatality rates of over 300 deaths per 100,000 workers (Myers and Fosbroke 1994). In 1989, the logging industry had an estimated injury rate of 19.2 work-related injuries per 100 full-time workers compared to a general private sector rate of 8.2 work-related injuries per 100 full-time workers (U.S. Department of Labor 1991). Data suggest that these fatal and nonfatal injury rates have remained well above the national average for more than 30 years (McCormack 1963, Myers and Fosbroke 1994).

Based on these types of injury statistics, OSHA acted in 1989 to address the major occupational hazards associated with logging through a new proposed standard specific to the logging industry. Prior to this proposed rule, OSHA maintained a standard which was only applicable to pulpwood logging (U.S. Department of Labor 1988a). The 1989 proposed rule, which replaces the pulpwood standard, covers the entire logging industry, including sawtimber operations. The final OSHA logging standard was signed on October 4, 1994, with an effective date of January, 1995.

Impact On The Logging Industry

Unlike the prior pulpwood standard, the OSHA logging standard covers both pulpwood and sawtimber harvesting operations. As such, the standard impacts mostly those parts of the country which do not harvest large quantities of pulpwood and do not have existing state-specific regulations on worker safety and health for sawtimber logging operations. In general, logging operators in the eastern hardwoods, central hardwoods, and mountain regions of the United States are for the first time required to plan for and meet worker safety and health standards set by OSHA. Logging operators in the South and Lake States that harvest primarily pulpwood will see less of an impact on their operations because of their compliance with the prior OSHA pulpwood standard, while those engaged with sawtimber harvesting will be affected. Logging operators in the Pacific Northwest will see little impact because of existing state regulations that cover sawtimber harvesting. The main focus here is on the probable impacts the OSHA standard has on hardwood sawtimber operations that primarily use manual felling with log skidders as the timber harvesting method.

The OSHA logging standard contains four areas: Training; general requirements; equipment protective devices; and tree harvesting. The intent of these sections is to provide, as much as possible, performance requirements for employers to reduce the risk of work injuries. A performance requirement is one that states the objective of a rule, but allows employers broad latitude in how they meet the objective (U.S. Department of Labor 1989). Specification requirements, a rule which sets forth a specific requirement that employers must adhere to, also exist in the standard.

The new OSHA standard requires employers to be more responsive to the training of their employees for the specific tasks they are to perform, including providing them with sufficient knowledge to identify and avoid hazardous conditions on the logging site. The new standard requires employers to maintain the proper maintenance of tools and equipment used on the logging site, and requires a greater use of safety devices on such equipment as skidders and yarders to protect the operator and other workers. The standard also requires the employer to be more conscious of work assignments and work activities to prevent bystanders from being exposed to hazards related to felling trees or rolling logs on the logging site.

These general changes may require hardwood sawtimber logging operations to be modified, not only in how employers prepare their employees and equipment to do the logging job, but also in how the logging operation is laid out and conducted. Such methods as having the bucking crew and skidder crew located within the same area where trees are actively being felled is not be permitted under the OSHA standard. The location of skid trails and landings may have to be modified in such a way as to reduce the exposure of skidder operators or workers in the landing area from hazards caused by tree felling.

Impact On Forest Management

While the OSHA logging standards are the primary responsibility of the employer, forest managers need to understand these standards to reduce, as much as possible, conflicts between management objectives and logging safety requirements. The impact of the new OSHA regulations could result in changing the requirements of a hardwood logging job to the extent that forest management decisions may need to be modified to facilitate the logging of a stand in accordance with safety regulations. Being aware of the OSHA requirements before finalizing the marking and layout of a logging operation can avoid such potential conflicts.

Some aspects of the OSHA regulations that forest managers should consider are:

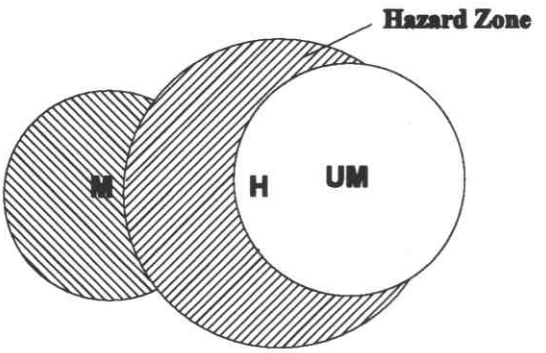
Snags. One specific provision of the standard for hazard abatement requires the identification and removal of hazardous dead, broken, or rotted trees or limbs from the area before felling marked trees. If the hazardous tree, or limb, is not removed, then it must be avoided. No work can occur in the hazard area around the hazardous tree or limb except to make the area safe. In the case of hazardous trees, since the concern by OSHA is that the tree may fall unexpectedly into a work area, the hazard zone defined by the hazardous tree would be expected to be the same as that for a tree which is being manually felled--two tree-lengths around the hazardous tree (Figure 1). Table 1 provides some examples of the acreage associated with the hazard zone from hazardous trees left in the stand.

Table 1. Acreage* for a hazard zone associated with hazardous trees in a stand of timber.

Height of Hazardous Tree (feet)	Acreage of Hazard Zone
50	0.72
60	1.04
70	1.41
80	1.85
90	2.34
100	2.88

* Acreage=[π {2 (Height of Tree in Feet)}^{2}]/ 43,560

Forest managers who intend to leave snags, or other poor quality trees, for the purpose of wildlife habitat should determine if the tree will pose a hazard to a feller. If so, the manager should adjust the marking of timber to ensure that the hazardous tree will not be within two tree-lengths of any marked tree, or mark the hazardous tree for removal and select an alternative, nonhazardous tree for wildlife purposes. The manager should also consider the placement of skid trails, landings, or other similar work areas to ensure they are all two tree-lengths away from the hazardous trees. Managers may also consider flagging the area around hazardous trees to ensure that loggers working in the stand are aware of the hazard.



H-Location of Hazardous Tree
M-Location of Markable Tree
UM-Location of Unmarkable Tree

Figure 1. Diagram of the OSHA "Two Tree-Length" Rule and its impact on marking trees.

Skid Trails and Roads. The requirements for road construction are not included in the OSHA logging standard because they are covered under the OSHA Construction Standard (U.S. Department of Labor 1988b). The placement of skid trails, however, are affected by the standard. The placement and construction of trails are not permitted to exceed the stability limitations of the machine.

The standards would require the forest manager to give more consideration to the placement of skid trails such that they do not involve crossing steep grades, which are common in hardwood stands. Where such grades are unavoidable, other options may need to be considered for the moving of logs, such as cutting roads to reduce the grade. At the same time, the manager would need to make decisions during the layout of the trails on whether to remove hazardous trees (which if left standing, might meet management goals for wildlife habitat) to allow for the placement of skid trails on acceptable grades, or whether to relocate skid trails in less acceptable areas to retain specific hazardous trees.

Locating trails to meet safety considerations also has the potential to conflict with trail locations that would meet environmental considerations, especially with respect to the crossing of streams, or other sensitive areas in the stand. The running of skid trails perpendicular up steep slopes, which may be feasible without exceeding the stability limitations of a skidder, is a limited option due to current environmental regulations that, in many instances, restrict or ban such trail locations because of the environmental concerns. Thus, the objective of reducing as much as possible the running of skidders on steep slopes is a recommended practice from both a safety and environmental standpoint. However, the use of the most level terrain for the placement of skid trails from a safety perspective, may conflict with buffer zone locations, or require the crossing of streams that would be unacceptable from an environmental perspective.

Location of Landings or Yards. One provision of the OSHA standard is the requirement that work ongoing in a stand be spaced such that fellers actively felling trees be at least two tree-lengths away from other logging activities such as bucking, limbing, skidding, yarding, or landing activities. This requirement, while not necessarily impacting the placement of landings and yards, would require the recognition of where the landings and yards are located with respect to ongoing felling activities. The forest manager must decide during the marking of the stand on whether to refrain from marking any tree within two tree-lengths of the

proposed landing or yard, or whether to mark timber in this area knowing that no landing or yarding activities would be permitted while those trees are felled. The location of hazardous trees within two tree-lengths of the landing or yarding area must also be resolved, as stated previously.

Sequence of Harvesting. The OSHA standard requires that harvesting of the stand be conducted such that the manual felling be conducted up-slope of, or on the same level as, previously felled trees (Figure 2). The standard would, from a practical standpoint, encourage the logging operation to proceed from the bottom of the slope to the top of the slope. In many instances, skid trails would need to be constructed prior to initiating logging, especially if the landing areas and access roads for the logging job are located at the top of the slope. If the access roads are located at the bottom of the slope, skid trails would require constant maintenance to remove slash because the skid trails would go through previously timbered areas. Concerns for soil erosion and run-off from the logging site also need to be considered. This may require planning for the placement and maintenance of water bars, or other soil stabilization techniques, during the logging operation rather than at the end of the operation to reduce erosion. Initiating logging at the top of the slope and using the lower slope as a buffer zone during the logging operation is still feasible, but would require logging to be conducted laterally, and would not allow for felling trees up-slope into previously logged areas.

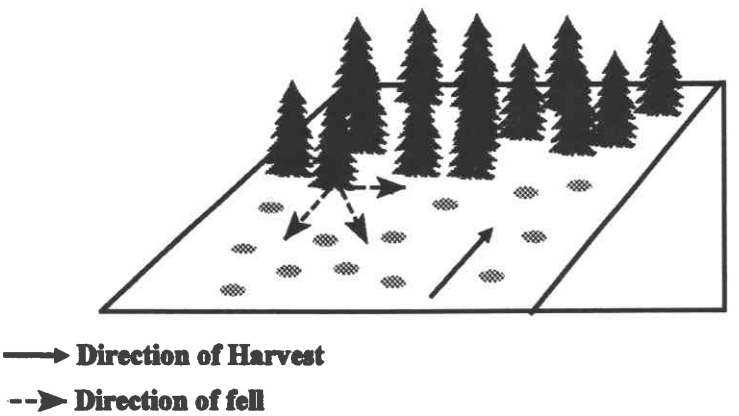


Figure 2. OSHA requirement for felling trees on sloped terrain.

The issues presented here are not all-inclusive of the potential impact the OSHA logging standard may have on a forest manager. They reflect the more obvious issues forest managers dealing with hardwood stands will need to consider when planning a harvest.

The forest manager should also consider that more time may need to be allotted to complete some logging jobs. Time requirements will be affected by such factors as: The type of harvest proposed for the stand; the temporary suspension of work due to felling occurring within two tree-lengths of other activities; the requirement that fellers not be located within two tree-lengths of each other while felling trees; the required suspension of logging operations due to hazardous weather; the suspension of work in an area until a hazardous tree, or other hazardous condition has been

removed; the requirement that hung-up trees be removed by the use of skidders, or other similar mechanical system that isolates the worker from the hung-up tree; and the requirement that trees felled on slopes be blocked prior to limbing or bucking if they cannot be limbed and bucked from the up-hill side of the tree. Failure by the forest manager to consider these new work requirements for the logging crew when determining the amount of time needed to safely log a stand of timber could result in poor time estimates. Taking such factors into account will prevent unnecessary conflicts with the logging operator over how quickly the logging job is being conducted.

The new OSHA standard may also create a new level of responsibility for the forest manager. With the adoption of the logging standard by OSHA, the standard has become a common industry practice and establishes a standard of care for third party, non-direct employers (Smith 1991). The forest manager may not cause a logging operator to knowingly violate the OSHA standard, which could be a consequence of the manner in which the forest manager marks the timber and lays out skid trails, landings, and other aspects of the harvest. If a forest manager knowingly sets forth requirements in violation of the OSHA standard, the forest manager may be found to be a liable party under third-party tort law for any injury, fatality, or property loss that is incurred by the logging operator, or worker, because of that requirement. Furthermore, OSHA is considering a requirement in construction that would place responsibility upon primary contractors for maintaining injury and illness records for all employees working on a work site, including subcontractors (Newell 1994). This precedent of making primary contractors accountable for injuries of subcontract employees could extend to other industries, including logging. Thus, while not directly responsible for ensuring that a logging operation is conducted in compliance with the OSHA logging standard, forest managers are indirectly responsible because they are the individuals planning and stating the conditions under which the logging operations will be conducted.

FOREST MANAGER'S IMPACT ON LOGGING SAFETY

Compliance with the OSHA logging standard will be the primary responsibility of the logging operator. However, this does not mean that forest managers do not have a professional or moral obligation to consider the safety of workers performing logging in a stand for which the forest manager is responsible. The Society of American Foresters' code of ethics makes it clear that forest managers are to perform their work with the utmost honesty, integrity, and professionalism (Society of American Foresters 1992). Forest managers have traditionally applied these professional standards to the land owner, or other employer of their services. Clearly, the forest manager has a professional obligation to the land owner, or other employer, to ensure that all logging operations performed on their land meet existing regional, state, or federal laws, including those related to safety. Furthermore, forest managers should apply these same professional standards to all constituencies they interact with. Logging operators are clearly a constituent of the forest manager--one that forest managers need if they are to meet most, if not all the goals outlined to them by a land owner, or other employer. Therefore, the forest manager has a clear professional obligation to assist the logging operator in making any logging operation as safe as possible whether one views this from the professional obligation to the owner of the timber, or from the professional obligation to society in general.

There are several ways that the forest manager can become involved in promoting the safety of workers logging a stand under the forest manager's care. The single most effective means would be through the contract for the logging job. It is common practice to include conditions in logging contracts for environmental requirements in the logging operation (e.g., placement of water bars, culverts, seeding skid trails). The incorporation of safety into the contract can be simply done by including a provision that the operator will perform the logging operation in compliance with the OSHA logging standard. Provisions for fines, or halting work on the site for violations of this provision, would be handled in the same manner as with environmental requirements. The forest manager will need to have a sound working knowledge of the OSHA standard, but this is no different than the need of the forest manager to have similar working knowledge of environmental or other regulations that are part of conducting a timber harvest.

The forest manager can also influence the safety of the logging operation beyond the provisions of the OSHA standard. Decisions on what types of silvicultural treatment will be used in a particular stand can greatly influence the types of hazards the logging workers will face when harvesting the timber. Many of the hazards faced by loggers are

associated with felled trees striking other standing timber, which can result in the felled tree becoming hung up, breaking a top, or changing its direction while falling (Peters, 1991). The use of silvicultural treatments that will minimize the likelihood of felled trees striking standing trees would reduce worker risk.

Where the use of potentially lower-risk silvicultural treatments is not possible, the forest manger can still reduce the risk to the logging worker through the layout of the logging operation and through the marking of the timber to be harvested. The issues of the placement of landings, skid trails, and roads has been previously mentioned. The marking of timber around hazardous trees has also been noted, but there is more the forest manager can do, especially in marking the stand.

The felling of a tree is the greatest hazard in logging (Peters 1991, Myers and Fosbroke 1994). The forest manager should be aware of this and attempt to visualize how each marked tree will be felled by a logger, noting possible hazards the logger will face in the process (e.g., high likelihood of the marked tree becoming hung up in an adjacent tree, limited options for the placement of an escape path for the feller). A system for assessing the likelihood of hanging a felled tree into a standing tree, and other related hazards, has been proposed by Peters et al. (1993), and may be adaptable for use by forest managers for field use. If the hazard can be reduced by removing another tree in the felling area, the forest manager may consider marking that tree along with the original tree marked for removal. If there is no obvious way to reduce the hazard to the logger, the forest manager should consider marking an alternative tree which poses less risk.

The forest manager should also consider leaving den trees that are sound and pose little or no hazard to the logger. As many hazardous trees as possible should be removed in the stand, leaving only those that are needed to meet specific management goals. While this may require leaving merchantable trees for wildlife purposes, it does give the forest manger more flexibility in marking the stand by not losing large areas of the stand due to the OSHA hazard zone requirement for hazardous trees.

The use of directional felling when logging the stand is another requirement the forest manager may consider. This will encourage the proper use of undercuts and backcuts when felling a tree, improve the likelihood that hazards identified by the forest manager when marking the stand are noted by the feller when dropping the tree, and allow for safer limbing, bucking, and skidding of the felled timber because trees would be felled in a logical, rather than a random direction. The use of directional felling also has the benefit of reducing the damage to the residual standing timber, improving the long-term timber value of the stand (Simmons 1979).

While forest managers cannot ensure that all aspects of a logging operation are conducted in a safe manner, the more emphasis they place on worker safety in those aspects of the logging operation where they are involved, the more likely the logging operator is to log the stand in a safe manner. Taking such a position on logging safety is no different than the forest manager's role in actively enforcing environmental requirements, or placing penalties on a logger who harvests non-marked trees. By setting the goal of having all logging operations they oversee being done safely, forest managers can have an impact, an impact which they should actively pursue as part of their professional work ethic.

THE FUTURE

With the establishment of the OSHA logging standard, logging safety is an area that many forest managers managing hardwood forests will be asked to incorporate into their management decisions for the first time. This will require forest managers to not only understand what parts of the OSHA standard will affect their management decisions for a stand of timber, but also place an ethical obligation on them to see that they do all in their power to ensure that a logging operation is conducted in a safe manner. This will require forest managers to view their decisions in a new light, similar to their taking new views on management decisions based on environmental regulations and constraints. To do this job well, forest managers must have a better understanding of the basic concepts of logging safety and how to apply these concepts in a stand of timber. Education on safety concepts will be a part of this, not only for forest

managers presently working in the field, but as part of the formal educational system for the forest managers of the future. The more forest managers understand safety issues, the better they will be able to incorporate these issues into their everyday work.

The need to balance safety issues with other factors such as environmental regulations, economic goals, bio-diversity goals, or the many other constraints a forest manager must consider, is unavoidable. Forest managers cannot ignore the OSHA standard any more than they can ignore environmental regulations. This will necessitate compromise and ingenuity on the part of the forest manager. Still, in most instances, solutions do exist and it will be the forest manager who will need to find these solutions.

In many instances, the concepts of forest management and forest harvesting have not worked closely together. The forest manager makes the decisions on what is to be harvested with little concern on how the harvesting is done. New harvesting methods are only viewed as an improvement to an existing silvicultural tool. This "removed" attitude toward harvesting systems ignores how important a tool harvesting is to forest management. As with any tool, the forest manager can only use it effectively if he completely understands it. Forest managers need to know how harvesting systems work, how economical harvesting will be in any given stand of timber, and not remove themselves from the role of making harvesting systems better. Forest managers also need to become more involved in the development of safer harvesting systems (National Research Council 1990). By tying together the goals of forest management with the goals of developing safer harvesting systems, the forest manager gains additional flexibility in balancing logging safety with all the other issues that must be considered in managing a forest. Research into the development of management-harvesting models could provide many solutions, not only for logging safety issues, but for environmental and other concerns as well (National Research Council 1990). Without this interaction between forest management and forest harvesting, the forest manager will always be asked to adapt to, rather than influence, harvesting technology--asked to adapt to a tool, rather than shape the tool for the intended purpose.

SUMMARY

The new OSHA logging standard will have an impact on the forest manager managing hardwood stands. Areas for which the forest manager will see the greatest impacts are the marking of stands, the placement of roads and skid trails, the location and development of landings or yards, and possibly the sequence in which the stand is logged. The forest manager can also play a major role in decreasing the hazards in logging timber through taking a professional interest in logging safety. Of the many influences that the forest manger can effect, the greatest safety influences can be effected in the logging contract, and in how the stand is marked for harvesting. Finally, the forest manager should take a more active role in the development of new harvesting technology, especially since it is the forest manager who is a major end user of these new harvesting techniques.

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HARDWOOD SILVICULTURE AND SKYLINE YARDING ON STEEP SLOPES:

ECONOMIC AND ENVIRONMENTAL IMPACTS

John E. Baumgras[1] and Chris B. LeDoux[2]

Abstract: Ameliorating the visual and environmental impact associated with harvesting hardwoods on steep slopes will require the efficient use of skyline yarding along with silvicultural alternatives to clearcutting. In evaluating the effects of these alternatives on harvesting revenue, results of field studies and computer simulations were used to estimate costs and revenue for skyline yarding operations. The methods evaluated include group selection and three treatments each for conventional shelterwood, thinning, diameter limit, and irregular shelterwood to initiate two-age management. Harvesting costs ranged from \$15.97 to \$42.22/100 ft^3, gross revenue from \$59 to \$131/100 ft^3, and net revenue from \$58 to \$2,809/acre. Environmental impacts observed at field sites and those reported in the literature indicate that relatively low levels of soil disturbance and residual stand damage can be achieved with skyline yarding.

INTRODUCTION

Public concern about the clearcutting of eastern hardwoods, combined with contemporary forest management issues such as ecosystem management have encouraged forest-land managers to consider alternative silvicultural practices and harvesting systems. Alternatives for even-age management include shelterwood and group selection, while irregular shelterwood (deferment) cuts can be used to initiate two-age management. In sawtimber stands, thinning can yield the volumes of sawlogs required for commercial operations. While often criticized, diameter-limit cuts are perhaps the most popular option on private forestland.

The adverse visual impact of large clearcuts has been a major source of public concern. Forested landscapes are affected by the size of clearcut units and the percentage of the viewing area that has been harvested (Palmer and others 1993). Interior views of harvested stands were evaluated for six silvicultural methods ranging from clearcutting to single-tree selection. Scenic beauty increased with residual basal area and years elapsed since harvesting, and decreased with amounts of logging slash (Pings and Hollenhorst 1993). Modifying the size and spatial relationships of harvest units and/or using silvicultural practices other than clearcutting can ease public concerns about timber harvesting.

The growing emphasis on maintaining the productivity and sustainability of forested ecosystems makes protection of soil and water resources an increasingly important issue. Efforts to reduce environmental impacts are primarly responsible for the reintroduction of cable yarding to the eastern hardwood region (Patrick 1980). Using a skyline yarder versus rubber-tired skidders can greatly reduce the need for truck roads and skid trails. (Kochenderfer and Wendel 1978) and significantly reduce the amount of bare or compacted soil within harvest units (Swanston and Dyrness 1973).

Until recently, commercial application of skyline yarding in the Appalachian hardwood region was confined largely to relatively large clearcut units. However, case studies in Pennsylvania (Fairweather 1991) and West Virginia (Wendel and Kochenderfer 1978) indicate that light to moderate residual stand damage is possible with cable yarding in partial cuts of hardwood stands on steep slopes.

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To reduce the environmental and visual impacts of harvesting timber on steep slopes in the Appalachian hardwood region, unconventional applications of cable yarding technology will be required to implement the silvicultural practices that the public now demands. Although harvesting revenue is not the sole consideration in forest management decisions, commercial timber sales are needed to meet goals related to ecosystems management objectives on public lands. And because cash flows remain critical to harvesting decisions on private lands, harvesting economics is an important consideration in selecting harvesting systems and silvicultural practices for steep-slope hardwood sites.

METHODS

Computer simulation was used in a sensitivity analysis of harvesting costs and revenues for 13 silvicultural treatments. Simulation allowed standardized production assumptions to be applied to all treatments and provided estimates of cost and revenue for hypothetical treatments.

The information required to simulate specific harvesting operations was obtained through field studies of skyline yarding operations. A conventional shelterwood harvest, an irregular shelterwood harvest (deferment cut), and a thinning were monitored at a study site on the Nantahala National Forest near Franklin, North Carolina. The three 6- to 8-acre units were located in a yellow-poplar, red oak, and white oak stand, with a site index of 80 (50-year basis for northern red oak) on slopes of 30 to 50 percent. All units were yarded with an inexpensive shop-built yarder rigged with a 30-foot tower and a gravity outhaul carriage. Also studied was the use of a commercial skyline yarder in four group-selection units at a site on the Pisgah National Forest near Asheville, North Carolina (LeDoux and others 1991). Loggers at both study sites harvested pulpwood and sawlogs from trees 8 inches and larger in dbh. Cycle-time equations were developed and additional data obtained on yarding delay times and causes and times to rig skylines and change landings. We also developed stand tables for both the initial and cut stands.

Because of important differences between study sites and among harvesting units at each site with respect to equipment, unit dimensions, crew size, and delay times unrelated to silvicultural treatments, observed production rates and costs were not directly comparable. As a result, comparison of yarding production and costs for silvicultural alternatives was standardized using simulation models calibrated on observed production rates (using cycle-time equations) and delay times obtained from field data.

Silvicultural Treatments

The 13 treatments evaluated include the three treatments observed at the Franklin site: conventional shelterwood (stand 1), irregular shelterwood (stand 2), and thinning (stand 3). The initial attributes of these stands were similar (Table 1). The scope of the sensitivity analysis was broadened to include treatments that were not observed at the Franklin site but could be implemented to satisfy alternative ownership or ecosystem management objectives. These hypothetical treatments were developed using initial stand tables from the Franklin site and modifying the cut-stand tables from the observed treatments.

The conventional shelterwood harvest observed at the Franklin site removed approximately 70 percent of the initial basal area of trees 7.5 inches and larger in dbh. Alternatives that removed approximately 30 and 50 percent of initial basal area from the smaller diameter classes represent treatments required to obtain oak regeneration on medium to good sites (Schlesinger and others 1993). For the three conventional shelterwood treatments, the mean dbh of harvested trees ranged from 11.9 to 17.1 inches (Table 2).

The irregular shelterwood or deferment cut was applied to initiate two-age management. The observed harvest of stand 2 (Table 1) removed all but 20 ft^2/acre of basal area, leaving mostly trees that were 8 to 18 inches in dbh. Two variations of the observed treatment were developed by modifying the diameter distribution of the residual basal area, leaving trees 12 to 22 and 20 to 30 inches in dbh, respectively (Table 2). For these three treatments, the mean dbh of trees harvested ranged from 15.3 to 17.1 inches. The different treatments could affect the visual quality of the residual

stand, or reflect the scheduled harvest of residual trees. If the residual stand were to remain until the next crop matures, residual trees would be smaller than trees left to be harvested in the next 20 to 40 years.

Table 1.--Initial attributes of timber stands at the Franklin study site

Stand	No. trees/ per acre[a]	Basal area[a]	Mean dbh	Merchantable volume[b]
		ft^2/acre	inches	ft^3/acre
1	76	117	16.8	3,669
2	77	111	16.3	3,378
3	73	119	17.2	3,773

[a]Trees ≥ 7.5 inches in dbh.
[b]Volume of wood and bark to merchantable top ≥ 4.0 inches diameter (inside bark).

Equal amounts of basal area also were removed in all three thinnings. The observed thinning (essentially a thinning from above) removed 47 ft^2/acre of basal area in trees with an average dbh of 17.8 inches (Table 2). Modifying the diameter distribution of trees removed from stand 3 (Table 1) produced thinnings with cut trees averaging 13.7 and 15.1 inches in dbh (Table 2). These two alternatives would concentrate growth on the largest trees and help create an "old growth" or park-like appearance. Thinning trees with a mean dbh of 13.7 inches also produced a cut stand similar to that resulting from thinning in younger sawtimber stands.

Two types of silvicultural treatments not observed at the Franklin site--group-selection and diameter-limit harvests--also were evaluated using the initial stand tables for stands 1, 2, and 3. Because results were similar, only those from stand 1 are presented. The diameter-limit cuts were evaluated with dbh limits of 12, 15, and 18 inches. The group selection harvested all trees 7.5 inches and larger in dbh. Management alternatives for applying group selection include the size of openings created and the spatial relationships between units. For this analysis, openings of 0.23, 0.92, and 2.75 acres were evaluated; the 0.23- and 0.92-acre openings were located at varying distances from the yarder landing.

Economic Analysis

Production rates for cable yarding were estimated with THIN, a computer simulation model (LeDoux and Butler 1981). The simulations applied the yarder cycle-time equation developed for the shop-built yarder observed at the Franklin site, as well as delay times that included a standardized component for all silvicultural treatments and a delay component unique to each treatment. The yarding corridor was standardized at 150 ft by 800 feet (2.75 acres). In addition, group-selection cuts also were simulated for openings of 100 by 100 feet and 200 by 200 feet. These openings were located 100 to 700 feet from the yarder landing. The times to move the yarder to a landing and to change corridors at each landing were standardized using the average of times recorded on all units at the Franklin site. The log populations required for each THIN simulation were obtained from cut-stand tables unique to each treatment; it was assumed that stems were bucked to a maximum log length of 32 feet as observed at the Franklin site.

Felling and limbing production rates, log populations for the THIN simulations, and yields of roundwood products were estimated with GB-SIM, a harvesting simulation model developed for ground-based harvesting systems (Baumgras and others 1993). Because felling and limbing generally are completed before yarding begins, felling and yarding production can be simulated independently. Inputs to these simulations include the cut-stand tables specified

for each silvicultural treatment. GB-SIM uses tree-taper equations developed for Appalachian hardwoods to estimate log lengths and volumes. Equations for estimating percentages of trees by tree grades, and sawlog volumes by log grade also are used by GB-SIM to estimate sawlog yields by species and grade.

Table 2.--Attributes of cut stands resulting from conventional shelterwood, group selection, and diameter limit applied to stand 1, irregular shelterwood applied to stand 2, and thinnings applied to stand 3.

Treatment	No. trees per acre	Basal area	Mean dbh	Volume per acre	Average volume per tree
		ft^2/ac.	inches	---------ft^3----------	
Conventional Shelterwood[a]					
Cut 70%	50	80	17.1	2,519	50.4
Cut 50%	44	53	14.9	1,614	36.7
Cut 30%	36	28	11.9	817	22.7
Irregular Shelterwood[b]					
Lv. trees 8-18 inches	58	91	17.1	2,852	49.2
Lv. trees 12-22 inches	63	91	16.3	2,728	44.0
Lv. trees 20-30 inches	71	91	15.3	2,711	39.3
Thinning[c]					
17.8 inches $\bar{X}$ dbh	27	47	17.8	1,556	57.6
15.1 inches $\bar{X}$ dbh	38	47	15.1	1,493	41.5
13.7 inches $\bar{X}$ dbh	46	47	13.7	1,327	32.3
Group Selection[d]	76	117	16.8	3,669	48.3
Diameter Limit[e]					
12 inches dbh	52	99	18.6	3,311	63.7
15 inches dbh	38	86	20.2	2,911	76.6
18 inches dbh	25	67	22.2	2,299	91.9

[a]Percentage of initial basal area.
[b]Dbh of residual trees.
[c]Mean dbh of cut trees.
[d]Volume/acre from groups harvested, not entire stand.
[e]Minimum tree dbh harvested.

Production rates for log bucking and loading estimated from results of production studies (Baumgras and LeDoux 1989) indicate that these rates would greatly exceed simulated yarder production for all treatments. Since bucking and loading production would be constrained by yarding production, fixed costs and wages for yarding, bucking, and loading were allocated to production at the rate estimated for yarding. Felling and limbing costs were based on production rates simulated for these functions. Rates for wages and equipment costs were obtained from USDA Forest Service appraisal guides and published rates for chain saws, loaders, and trucking. All rates were inflated to 1994 levels using the Producer Price Index for all commodities.

Estimated logging costs represent the sum of felling, limbing, yarding, bucking, and loading costs. Net revenue represents the gross value of roundwood products delivered to the mill minus logging and hauling costs. Delivered prices for sawlogs and pulpwood represent median prices obtained from published price reports (Pa. State Univ. 1990-93; Tenn. Div. For. 1990-93) (Table 3). The estimates of haul cost assume a 40-mile haul distance.

Table 3.--Roundwood product prices applied to economic analysis

Species	Sawlogs[a]			Pulpwood[b]
	Grade 1	Grade 2	Grade 3	
	Dollars/mbf			Dollars/100 ft^3
White oak	325	185	100	45
Red oak	485	285	125	45
Yellow-poplar	175	125	85	45
Other hardwoods	190	140	85	45

[a]Factory grade sawlogs, USDA Forest Service grades 3 or better, scaling diameter ≥ 10 inches.
[b]Roundwood with diameter inside bark ≥ 4 inches, not meeting the quality or dimension requirements for sawlogs.

Environmental Impacts

Environmental impacts were monitored on the shelterwood and thinning units harvested at the Franklin site. Soil disturbance was surveyed by sampling disturbance classes (Dyrness 1965) at 5-foot intervals along 50-foot random azimuth transects. Residual stand damage was sampled by recording damage to trees located on 1/10-acre plots after felling, and again after yarding. To assess visual quality before and after harvesting, oblique aerial photographs were taken of each harvest unit.

RESULTS

Stand Attributes

Because the three initial stands in Table 1 are similar and 7 of the 13 treatments evaluated were applied to stand 1, differences between stands harvested were largely a function of the silvicultural treatments. For the diverse array of treatments estimates of volume harvested ranged from 817 ft^3/acre for the 30-percent shelterwood harvest to 3,669 ft^3/acre for the group selection (Table 2). Average volume per tree harvested ranged from 22.7 ft^3 for the 30 percent shelterwood to 91.9 ft^3 for the 18-inch diameter-limit cut.

Differences among the three treatment levels evaluated for each of four silvicultural methods were most pronounced for the conventional shelterwood cuts: volume harvested ranged from 817 to 2,519 ft^3/acre, and average volume per tree ranged from 22.7 to 50.4 ft^3 (Table 2). The three diameter-limits also had a large impact on attributes of the cut stand. Increasing the diameter limit from 12 to 18 inches dbh reduced the volume harvested by 30 percent, but increased average tree volume by 44 percent. The three irregular shelterwood cuts showed the least amount of variation. With only 20 ft^2/acre of basal area, the diameter distribution of the residual trees had minimal impact on volume harvested (2,711 to 2,852 ft^3/acre) and average volume per tree (39.3 to 49.2 ft^3).

Cost and Revenue

The results in Table 2 show how the selection and application of silvicultural methods can affect critical attributes of the harvested stand component. These attributes, especially volume per acre and volume per tree, are closely correlated with harvesting system production and costs, and with product yields and revenue.

The sensitivity of cost and revenue to silvicultural treatments is most evident from estimates of net revenue that range from $58 to $2,809/acre (Table 4). The 12-inch diameter-limit, group-selection, and 15-inch diameter-limit cuts produced the largest estimates of net revenue, $2,809/acre, $2,789/acre, and $2,712/acre, respectively. The two lowest estimates resulted from the 30-percent removal shelterwood ($58/acre), and the thinning of trees averaging 13.7 inches dbh ($382/acre).

The heavy stocking of large trees and the substantial component of high-value oaks contributed to relatively low harvesting costs and high revenue per unit of volume. Given the assumptions of the analysis, estimated revenue generally exceeded costs by such a wide margin that the economic feasibility of 11 of 13 options tested was not a critical issue. Four of the 5 silvicultural methods and 7 of the 13 specific treatments yielded more than $2,000/acre.

The key attributes of logging-machine production cycles form the link between harvesting conditions and harvesting-system production. Although machine cycle times generally increase with average tree volume and/or numbers of trees per cycle, the relative gains in volume exceed the increases in cycle times. Consequently, production rates generally increase with trees/acre, volume/acre, and volume/tree. These relationships are modeled by the THIN and GB-SIM programs.

The lowest harvesting costs are associated with the highest volumes per acre. In the case of the three conventional shelterwood cuts, reducing volume per acre from 2,519 to 817 ft^3 and average volume per tree from 50.4 ft^3 to 22.7 ft^3 (Table 2) more than doubled harvesting costs (Table 4). When only the diameter distribution of basal area removed in thinning was altered, the changes in volume/acre and volume/tree caused logging cost to increase by 46 percent, from $21.50 to $31.45/100 ft^3.

Another important element in the net revenue equation is the value per unit of production, represented by gross revenue in Table 4. Because larger diameter trees generally yield higher proportions of merchantable volume in grade 1 or 2 sawlogs, gross revenue is highly correlated with the mean dbh or average volume of trees harvested. Although the tree and log-grade estimates incorporated in this analysis are not completely site specific, they represent observed stand attributes and well-documented relationships among tree species, dbh, and sawlog quality. Estimated gross revenue ranged from $59 to $131/100 ft^3 (Table 4), reflecting the estimated grade distribution of sawlogs and the proportions of sawlog and pulpwood volume. The percentage of sawlog volume in grade 1 logs ranged from 10 to 38. The percentage of merchantable volume in pulpwood ranged from 13 to 59.

Estimates of cost and revenue for the group-selection cut (Table 4) represent a 150- by 800-foot (2.75-acre) opening adjacent to the landing. Because the cost of changing landings or rigging the skyline to yard each unit is largely independent of unit area, smaller units result in higher costs per unit volume for moving the yarder and changing corridors. Depending on the location of the unit (100 to 700 feet from the landing), yarding a 100- by 100-foot (0.23-acre) unit increased costs by $14 to $22/100 ft^3 over those for the 2.75-acre unit. In this comparison, net

Table 4.--Simulated yarding production, estimated logging costs, gross revenue, and net revenue

Treatment	Yarder production[a]	Logging cost[b]	Gross revenue[c]	Net revenue[d]
	ft^3/hr.	Dollars/100 ft^3		Dollars/acre
Conventional Shelterwood				
Cut 70%	416	20.55	113	2,069
Cut 50%	336	25.48	98	1,004
Cut 30%	193	42.22	59	58
Irregular Shelterwood				
Lv. trees 8-18 inches	465	19.01	116	2,481
Lv. trees 12-22 inches	400	21.14	114	2,245
Lv. trees 20-30 inches	347	23.37	99	1,788
Thinning				
17.8 inches $\bar{X}$ dbh	426	21.50	124	1,441
15.1 inches $\bar{X}$ dbh	332	25.72	98	927
13.7 inches $\bar{X}$ dbh	257	31.45	70	382
Group Selection	454	18.77	105	2,789
Diameter Limit				
12 inches dbh	504	16.84	111	2,809
15 inches dbh	555	16.41	120	2,712
18 inches dbh	608	15.97	131	2,407

[a]Delay-free production rate.
[b]Total cost to fell, limb, yard, buck, and load.
[c]Average value of sawlogs and pulpwood delivered to mill.
[d]Gross revenue - logging cost - haul cost x volume/acre.

revenue is reduced by $513 to $789/acre. Compared to the 2.75-acre unit, a 200- by 200-foot (0.92-acre) unit located 300 to 600 feet from the landing increased costs by $5 to $8/100 ft^3 and reduced estimated net revenue by $163 to $174/acre. These comparisons do not include the additional cost of locating harvest units, planning yarding corridors, or locating the landings required to harvest numerous small units scattered throughout a larger harvest area. Also, the estimates of net revenue reported for group-selection cuts represent dollars from each acre actually harvested. Because the entire stand would not be harvested with each entry, net revenue per acre for the entire stand would be much lower than net revenue per acre harvested. For example, given a 20-year cutting cycle and a 100-year rotation such that 20 percent of the stand is cut each entry, net revenue per entry for the entire stand would be 20 percent of that estimated for each acre actually harvested.

The values in Table 4 for the group-selection cut represent harvesting the entire 2.75 acre-corridor. With respect to estimated costs and revenue per acre harvested, this is equivalent to a conventional clearcut. Expressing net revenue as a percentage of that estimated for this group selection cut illustrates the relative impacts of silvicultural alternatives to large-opening group selections or clearcuts (Fig. 1). Five of the 12 alternatives evaluated yield more than 80 percent of the revenue available from clearcutting. However, four alternatives yield less than half of the net revenue available from clearcutting, two less than 15 percent, and one only 2 percent.

Environmental Impacts

Analysis of data on soil disturbance at the Franklin site revealed no difference between the observed shelterwood and thinning treatments with respect to percentage of area in each soil-disturbance class. Deep soil disturbance and deep disturbance combined with visible soil compaction was found on 10 percent of the area. Most of this occurred within the yarding corridors, the extraction path between the tailhold and the yarder. Seventy-one percent of the area showed no disturbance.

The results of the stand damage surveys indicate that logging damage was significantly greater on the two shelterwood units than on the thinning unit. Sixteen percent of the residual trees were destroyed on the conventional and irregular shelterwood units compared to only 5 percent on the thinning unit. On the two shelterwood units, 13 percent of the trees received bark wounds larger than 100 square inches, versus only 1 percent on the thinning unit. Other damage, small bark wounds and abrasions and broken limbs, affected 51 percent of trees on the shelterwood units versus 36 percent on the thinning units.

Trees destroyed were uprooted or broken off, generally during felling operations. Most of the large bark wounds occurred during yarding operations, with frequency increasing with proximity to yarding corridors. Much of the difference between damage on the shelterwood and thinning units can be attributed to the heavier removals on the shelterwood units (Table 2). It is important to note that trees on the thinning unit were felled by an experienced crew while the felling crew on the two shelterwood units included relatively inexperienced chain-saw operators.

From the oblique aerial photos of the three harvested units at the Franklin site it is difficult to distinguish the thinning unit from the adjacent uncut area. The observed conventional shelterwood cut and the irregular shelterwood cut were visible on the photos, though there was much less contrast between the cut and adjacent uncut areas than would be apparent with a clearcut.

DISCUSSION

Each of the 13 silvicultural treatments evaluated represent different forest management objectives or desired future conditions. Consequently, the estimated cash flows are not intended for ranking or selecting treatments. Nonetheless, efficient management requires an understanding of the relationships between silvicultural prescriptions and harvesting revenue. Whether timber sales represent an important source of revenue or a means of managing vegetation, costs and revenue from harvesting operations play an important role in forest operations.

Results of the cash-flow analysis demonstrate the sensitivity of cost and revenue to silvicultural treatments: logging costs ranged from \$15.97 to \$42.22/100 ft^3, gross revenues from \$59 to \$131/100 ft^3, and net revenues from \$58 to \$2,809/acre. Due to the composition of the initial stands, the group-selection, diameter-limit, and heavy shelterwood cuts all yielded large cash flows. However, treatments requiring significant reductions in harvested volume per acre and/or volume per tree resulted in large reductions in estimated net revenue--as much as \$2,011/acre for the conventional shelterwood cuts and \$1,059/acre for thinnings. There also were significant variations in net revenue resulting from location and dimensions of the group-selection units. The estimates of harvesting cost and net revenue reflect the relatively low cost of a shop-built yarder, which is commonly used in southern Appalachia.

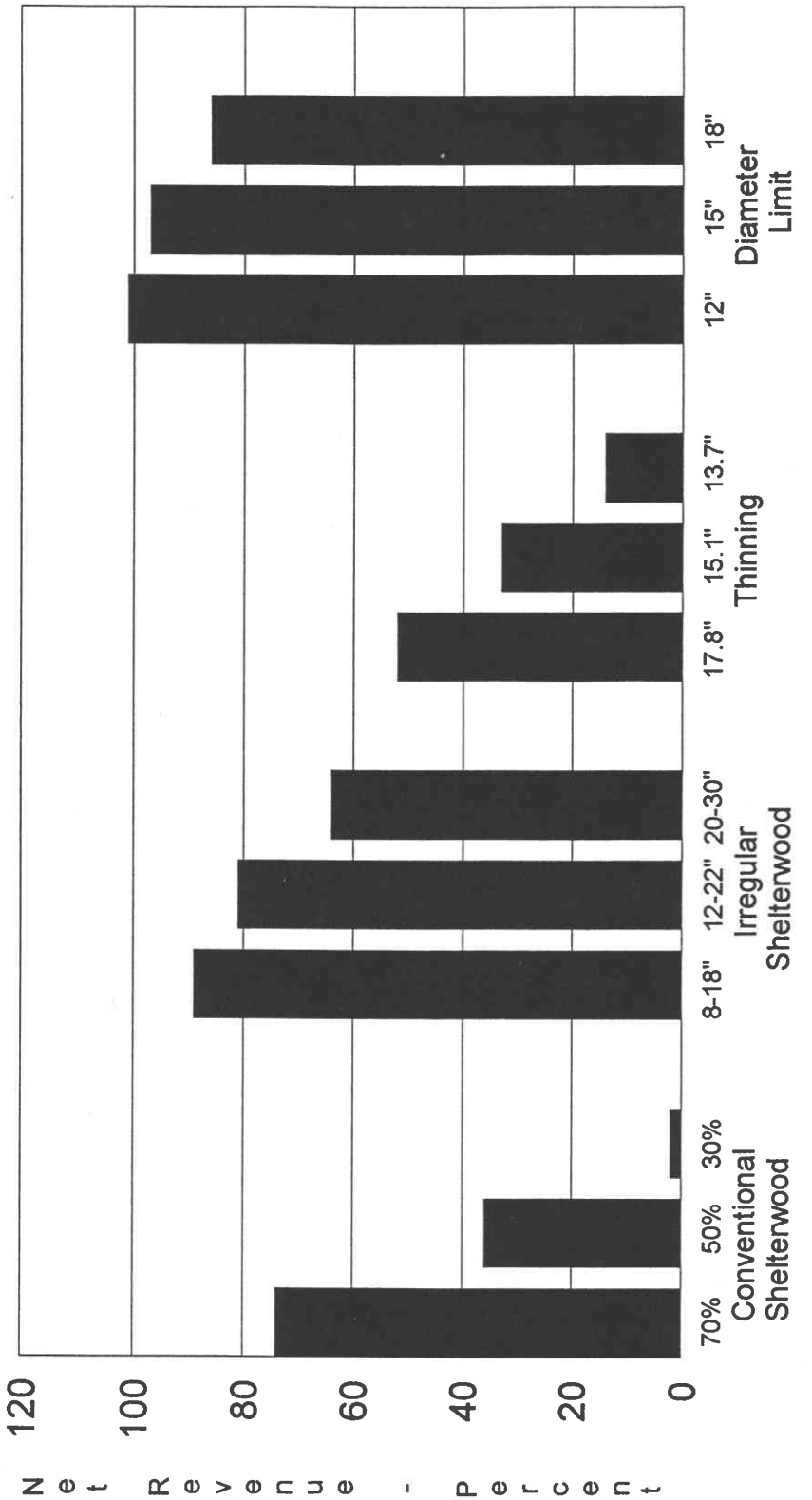

Figure 1. Percentage of net revenue available from the 2.75-acre group-selection cut; for conventional shelterwood removing 30, 50, and 70 percent of basal area; irreguar shelterwood leaving trees 8 to 18, 12 to 22, and 20 to 30 inches dbh; thinning removing trees averaging 17.8, 15.1, and 13.7 inches dbh; and diameter-limit cuts to 12, 15, and 18 inches dbh..

Compared to conventional ground-based harvesting systems operating in large clearcut units, the harvesting system and silvicultural treatments evaluated in this study provide an opportunity for significant reductions in environmental impacts associated with harvesting timber on steep slopes. Levels of soil disturbance and compaction sampled on observed shelterwood and thinning units were similar to those reported for western skyline operations (Swanston and Dryness 1973), less than those reported for ground-based systems (Hatchell and others 1970), and considerably less than those reported for mechanized whole-tree harvesting systems (Martin 1988). Observations at the Franklin site also indicated that soil disturbance on specific yarding corridors could have been reduced further by rigging the skyline for more lift and bucking large stems to lighten payloads. Effective application of cable yarding requires expert sale layout skills to locate yarder landings and yarding corridors that provide the skyline deflection required to allow the yarder to operate and to minimize soil disturbance.

Harvesting damage to residual trees can affect the health and value of future stands. The light damage on the observed thinning unit was similar to that reported for partial cuts in hardwoods harvested with rubber-tired skidders (Nyland and Gabriel 1971) and skyline yarders (Fairweather 1991). The heavier damage on both observed shelterwood units was comparable to that reported for conventional ground-based systems used in heavy shelterwood cuts in hardwoods (Nichols and others 1993).

Cable yarding in a variety of silvicultural treatments in Appalachian hardwoods (Wendel and Kochenderfer 1978) resulted in less stand damage than was observed for the thinning or shelterwood units at the Franklin site. These low levels of damage resulted from wide yarding corridors being felled before yarding, and directional felling to minimize felling damage and facilitate yarding stems through standing timber. Yarding damage also can be moderated by logging during the dormant season, or allowing for damage when marking the cut trees and then harvesting heavily damaged trees before moving the yarder to the next corridor. Radio-controlled carriages can be positioned to select the best extraction path to laterally yard logs to the skyline corridor. Forest managers also need to recognize the potential for stand damage when prescribing silvicultural treatments, especially where large-diameter trees or trees with large tops are harvested. Although residual stand damage is unavoidable, attaining acceptable levels of damage requires only that the condition of the residual stand satisfies silvicultural objectives.

Visual quality is one of the more important reasons why many forest managers are seeking alternatives to clearcutting. Compared to clearcut units, aerial photos of the Franklin site indicate an enhanced visual quality of harvested units, especially the thinning unit. Although interior views were not evaluated at the Franklin site, results reported by Pings and Hollenhorst (1993) contrast scenic values of interior views for similar silvicultural treatments. On a scale of 1 to 10, uncut areas rated 7.17 versus 5.12 for clearcuts. Ratings for other silvicultural treatments were 6.28 for thinnings, 6.22 for shelterwood, and 5.55 for irregular shelterwood. Cable yarding also improves the visual quality of harvest sites by eliminating the highly visible network of skid trails on steep hillsides.

As increasingly stringent constraints are imposed on harvesting practices, the more important it will be to recognize the implications of these constraints with respect to the economic feasibility of specific treatments and the tradeoffs among management alternatives. Skillful planning of timber sales and effective control of harvesting operations will allow forest managers to meet both environmental and silvicultural objectives.

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FOREX - AN EXPERT SYSTEM FOR MANAGING EVEN-AGED UPLAND OAK FORESTS ON STEEP TERRAIN

Chris B. LeDoux[1], B. Gopalakrishnan[2], and K. Lankalapalli[3]

Abstract: An expert system, FOREX, was developed to access the data from the database management provided by MANAGE. For a specific site index, the user of FOREX can obtain information on present net worth, optimal thinning entry timing, optimal stand rotation age, diameter at breast height, volume by grade and value of the trees harvested, and based on the cable yarder used, the average slope yarding distance, truck class, road class, log-bucking methods, and the number of thinnings desired. This paper describes FOREX, which is valuable to forest managers, planners, loggers, and landowners, planning to harvest even-aged upland oak stands located on steep terrain.

INTRODUCTION

Commercial forest management involves making decisions regarding various activities that range from regeneration efforts to commercial logging operations. A rigorous financial evaluation must be performed. Decisionmakers must know which variables affect cost and profitability and understand how these variables interact for a particular harvesting operation. They must be able to determine the total volume, species, and individual size of hardwood logs that can be removed without incurring a loss. To evaluate the long-term costs and benefits of the harvesting and silvicultural treatments that form the chain of activities and yields in the life of a stand, a computer systems simulation model called MANAGE was developed (LeDoux 1986). MANAGE, a computer program written in FORTRAN V, is a combination of discrete and stochastic subroutines. 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 the stand.

The model input is a detailed individual user-specified tree list, and the output simulates stand growth based on some user-specified silvicultural treatment, harvest of desired volume or stems with the logging system specified, sale cost of the wood, and economic analysis for the respective treatment and entry. The process is repeated for user-specified alternative mixes of entry timing, logging equipment, silvicultural treatments, and economic conditions for the life of the stand in question. The results are stored by entry and life of the stand. Users can specify numerous management scenarios during the life of the stand and evaluate the outcomes.

MANAGE, although valuable, takes substantial computing time for each execution when run on personal computers. The calculation of present net worth (PNW) of an activity is based on a number of variables such as slope yarding distance, buck type, and road class. The user has to specify the values of these variables each time the program is run. It is impractical to run MANAGE for so many combinations. Moreover, it is very difficult for a novice to understand the output of MANAGE and interpret the results. An expert system is best suited for this type of challenge. Here we detail an expert system called FOREX designed for managing even-aged upland oak forests on steep terrain.

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DOMAIN OF FOREX

The stand tables used to initiate the simulations were those published by Schnur (1937) for even-aged upland oak/hickory forests. Five sites with indices 40, 50, 60, 70, and 80 which are rated worst, bad, average, good, and excellent, respectively, are considered in this expert system. These site indices cover the majority of the range of tree growth characteristics of the development and yield potential available in the northeastern area of the United States. The management plan is to reduce each 2-inch diameter class of trees by 30 to 50 percent to simulate thinning the stand and 100 percent to simulate final harvest. This is a common approach for thinning stands. Thinning could be performed a number of times during the life of the stand before final harvest. Thinning is used as a variable; the user has the choice of performing from one to three thinnings before final harvest. Although we only highlight the results of the oak hickory database with the above mentioned treatments, the user could run MANAGE for the scenario of interest and thus expand the database.

The average slope yarding distance specified is 200 and 400 feet. In reality, any yarding distance may be used on the terrain in which the harvesting is being performed, but the forest manager could easily control the yarding distance to conform to the above mentioned values.

The grading of logs is controlled by a parameter called buck type. A value of 1 for buck type allows logs to be graded based on the length and diameter at each end regardless of where in the tree the log came from. A value of 2 for buck type, although essentially the same as buck type 1, allows only the butt logs to be graded as a grade 1 log. In this demonstration, the buck type was a user-specified variable.

The species mix is held constant throughout the simulations. The species mix included white oak, red maple, red oak, and hickory based on Schnur's (1937) tables and species commonly found in the oak/hickory upland oak range of the northeastern area of the United States. The species remained the same for each site index. Again, the user can manipulate species composition and related analysis for any scenario of choice and add to the database.

The yarders used for harvesting were Bitterroot for stands 7 to 9 inches in dbh, Koller K-300 for stands 9.1 to 16 inches in dbh, and Ecologger I for stands greater than 16 inches in diameter. The one-way mileage from the landing to the mill was assumed to be 25. The delay cost per cubic foot, the move cost per cubic foot, and the PNW discount rate were assumed to be 0.03, 0.03, and 0.03, respectively. Mill prices (in 1,000 board feet) for the logs by grade and by species were:

	Grade			Cordwood
Species	1	2	3	Cord volume
Red Maple	200	125	50	40
White Oaks	450	250	100	40
Red Oak	450	250	100	40
Hickory	210	160	100	40

The road class ranges from 2 through 6, with decreasing speed characteristics as class increases. The trucks that haul the logs are classified based on their capacity. The truck class ranges from 1 through 5, with decreasing capacities as class increases.

The scenarios within the domain of FOREX were limited to outcomes that breakeven or result in profitable ventures. Outcomes that did not breakeven were not included in the database.

The choice of the variables influencing the harvesting of the forest stand has been made by USDA Forest Service experts at the Northeastern Forest Experiment Station based on those that would provide the most applicability in the forest management domain (LeDoux 1986, LeDoux et al. 1994).

RESULTS FROM MANAGE

For each site index, given type, and for different combinations of the variables, MANAGE was executed approximately 1,000 times. Figure 1 shows the various combinations for which the runs were performed. The objective was to execute MANAGE for a specific set of conditions and change the age at which the harvesting activities are done so that breakeven is achieved.

Figure 2 shows a sample MANAGE simulation run. The results from MANAGE show age of the stand when the activity was performed, the number of trees that were present, average dbh, the volume of wood harvested, the sale cost of the wood, the cost of logging, and the PNW.

In this manner, until financial breakeven was achieved in each run, MANAGE was executed for different site indices, thus bringing the total number of runs to approximately 4,500. The data generated from such numerous executions of MANAGE were analyzed, organized and captured in DBASE III[4], a database management system.

DATABASE DESIGN

A database file was created for each site index, in DBASE III. The fields used for each record were: the site index; yarding distance; the road class; the truck class; the percentage of thinning; the activity age; the PNW of the activity; the optimal rotation age; the PNW corresponding to the optimal rotation age; the total PNW; the volume of timber by grade 1, grade 2, grade 3, and cordwood; dbh; total volume of timber for all grades; volume of timber for each grade at the optimal rotation age; number of trees cut at the activity age; and average dbh of the trees harvested at the optimal rotation age.

EXECUTION OF FOREX

The flow chart shown in figure 3 illustrates the execution of FOREX. At the start, the user is presented with two options: to obtain information on the PNW, dbh, and the volume for a specific set of management activity parameters (option 1), or to know what type of management activity to perform to attain the desired minimum PNW, dbh, and/or the volume (option 2).

Option 1

When the user chooses option 1, FOREX presents the user with a choice of site indices to specify. After specifying the appropriate site index, the user is then required to specify the values for the constant parameters in the forest management domain, such as the yarding distance, the buck type, the road class, the truck class, and the percentage and the number of thinnings desired. An important and convenient feature of FOREX is that the user can specify the value of any parameter as "unknown" and FOREX will accept it. If all parameters are known, then there will be a unique solution or result. If one or more parameters are unknown, then FOREX will present as many alternate solutions or results as applicable.

The results can be sent to a printer, a text file, or the computer screen, as desired by the user. The output may consist of hundreds of options or a few options depending upon the values of the parameters. The user may be interested in curtailing the number of alternate solutions by specifying a lower bound on the PNW, dbh, and/or the volume of wood obtained from the management activities. Another important feature of FOREX is that it remembers the values of the constant forest stand parameters used in the previous run when the user begins a fresh consultation. This feature enables the establishment of a link between successive executions of FOREX.

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[4]DBASE III, Registered Trademark of Ashton Tate; 1986.

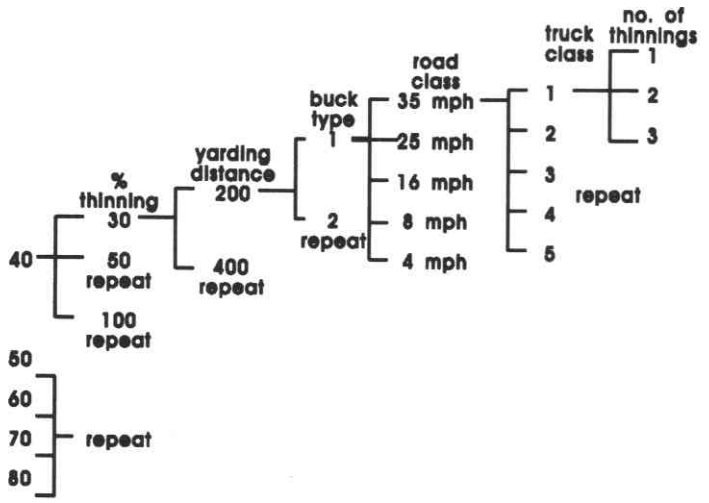

Figure 1. Various combinations of parameters.

```
                                    MANAGE pc
** COMPLETED **                <↑↓> to scroll summary          <ESC> main menu
** data is per acre  **                                  <space bar> acre/tract toggle
                              Summary of Current Run
Age  Live   AvgDBH     MerchVolume     MillPrice     LoggingCost       PNW
     Trees  (inches)      (ft3)          ($)           ($)            ($)
 90   226   10.17        3,317.35      2,824.91       1,617.53 v      1,207.38
 90   120   12.87        2,906.66      2,686.47       1,332.43 v      1,354.04
 90   106    5.73          410.70        138.44         512.40dv       -373.97
100   106    7.32          691.58        233.12         470.09 v       -160.09
110   103    8.82        1,030.64        347.41         560.57 v        -97.28
120    99   10.23        1,398.14        471.28         706.48 v        -72.51
130    98   11.61        1,815.57        606.90         870.60 v        -54.93
140    97   12.92        2,287.71      1,018.37       1,055.28 v         -5.19
150    95   14.11        2,725.18      1,975.58       1,222.04 v         71.63
160    94   15.31        3,249.17      3,029.88       1,422.36 v        103.23
170    89   16.34        3,598.45      4,201.04       1,548.25 v        115.09
180    86   17.27        3,972.84      5,540.42       1,685.14 v        112.99
190    80   18.41        4,313.53      7,014.73       1,802.11 v        103.21
200    76   19.37        4,647.73      8,390.10       1,919.06Dv         86.56
                          ***** end of projections *****

Manage Activities:        1              PNW of Activity(s):$ 1354.04
Optimal Rotation Age:   170            PNW of Optimal Rotation:$  115.09
Range of measures ages: 90 to 200 years                 Sum :$ 1469.13
```

Figure 2. Sample MANAGE run.

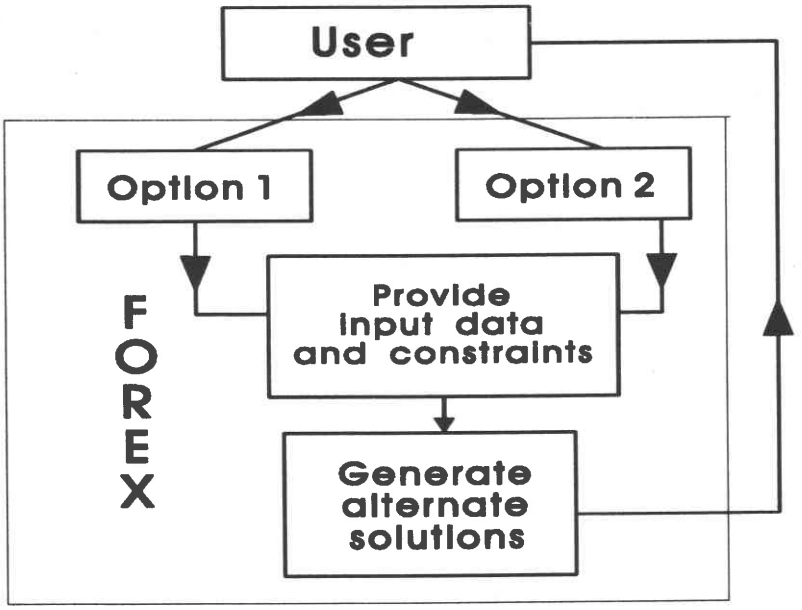

Figure 3. Execution of FOREX.

Option 2

In choosing option 2, the user elects to specify lower bounds on the PNW, dbh, and the volume at the very beginning, even before specifying any values for the forest stand parameters. This option is usually chosen by users who are not aware of the forest stand parameter values but do have a clear understanding of the objectives they want to achieve. They may be interested in obtaining a minimum PNW, and/or a minimum dbh, and/or a minimum volume of wood, and they do not care which management activity is used to attain them.

Thus, after choosing option 2, the user is asked to specify the lower bounds on the PNW, dbh, and/or the volume. The dbh and the volume constraints can be set for the last activity, whereas the PNW constraint can be set as a sum for all activities performed. This is followed by the specification of the forest stand parameters--again it is acceptable for many of these parameters to be "unknown." The second option presents a set of management activities that will meet or exceed the user-specified lower bounds.

If the user is presented with a large number of alternative solutions, the user may be interested in curtailing the number of solutions, as described for the first option. Although the program will cease to execute after providing the solutions for the second option, an important feature of FOREX is that it remembers the lower bound values used for the objectives in the previous execution of the FOREX second option. As for the first option, the results of the second option can be sent to a printer, a text file, or the computer screen. In both options, when the number of alternate solutions is less than five, the results will only print to the screen.

SYSTEM DESIGN FEATURES

As stated previously, the system is able to handle user-specified values of "unknown" to any variable. This situation is a characteristic of an expert system. If any variable is labelled as "unknown," the system reacts by finding the solution space that accommodates all possible values of the variable, if the user is pursuing the first option. On the other hand, if the user is pursuing the second option, wherein constraints are placed on the PNW, wood volume, and tree diameter, the system finds the solution space accommodating only those values of the "unknown" variable that lead to the values of PNW, wood volume, and dbh satifsying the constraints.

If the user would like to execute FOREX using the values of variables specified in the previous run, the system is designed to "remember" the values of the previous run. The system asks if the user wants to use those values. This aspect of system design facilitates sensitivity analysis.

OUTPUT OF FOREX

A typical output from FOREX is shown in figure 4. The user has chosen option 1 with a site index of 40, yarding distance of 200 feet, road class 4, (16-mph road), truck class 2, and buck type 1. The next question presented to the user is the level of thinning desired. The user states the answer as "unknown". The system then determines the number of alternate solutions as 7. This means that there are seven ways (options) to harvest the stand and realize benefits. Option 1 refers to full harvest. The optimal rotation age and the PNW are reported along with wood volume and the number of trees cut. Similar information is provided for option 2, which specifies 30 percent thinning once. Output includes average dbh of the trees, number of trees, the total volume of timber in cubic feet, along with the volumes of each grade (grades 1 through 3) in board feet and for grade 4 in cubic feet, PNW. The output can be sent to the computer screen, a text file, or to the printer. The useful aspect regarding the system is that the user can specify the name of the ASCII file and the system will not only create it, but also successfully append all results of all consultations with FOREX in that file, if the user so desires. Thus, the FOREX system design facilitates iterative use and hence generates a number of alternatives from which the user can select the best.

CONCLUSIONS

FOREX is an expert database integrated system that uses the data from simulation runs. It provides the user with the ability to perform sensitivity analysis and eliminates the need to sort through simulation runs. FOREX is an intelligent computer software that uses the principles of articifial intelligence and interfaces effectively with a database management system to bring a powerful tool into the hands of the forest manager.

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```
**********************************************************************
Option 1    Site:40       Yarding Distance:200     BuckType:1
                Road Class:4  Truck Class:2        Origin: 1932

                 The activity chosen is full harvest.

Optimal Rotation Age (ORA) is 160 years.
                                      PNW at ora is $88
The Average DBH is 14.12 inches.

Total Volume    = 4383 cu.ft
Grade 1 = 4971 board ft.
Grade 2 = 774 board ft.
Grade 3 = 159 board ft.
Grade 4 = 3104 cu.ft.
# of trees = 150

**********************************************************************
press any key to continue!!!
```

```
**********************************************************************
Option 2    Site:40       Yarding Distance:200     BuckType:1
                Road Class:4  Truck Class:2        Origin: 1932

Suggested activity is 30 % thinning 1 time.

Thinning        First      Full Harvest

Age (years)              130         160
# of Trees               62         120
DBH (inches)          12.02       14.50
Volume(cu.ft)          1262        3732
G1 (bd.ft)              486        4859
G2 (bd.ft)                0         175
G3 (bd.ft)              339        1058
G4 (cu.ft)             1046        2313
PNW ($)                   7          78

**********************************************************************
press any key to continue!!!
```

Figure 4. Sample output from FOREX.

EFFECT OF THE HARDWOOD RESOURCE ON THE SAWMILL INDUSTRY IN THE CENTRAL AND APPALACHIAN REGIONS

William Luppold [1]

Abstract: The Central and Appalachian hardwood regions contain a diverse and valuable timber resource. The regions are important to the hardwood industry because they contain 68 percent of the eastern hardwood sawtimber. Furthermore, more than 70 percent of the hardwood lumber produced in the United States is manufactured at mills located in 16 of the states in the regions. This paper examines the hardwood sawmill industry and its relationship to the hardwood resource in the Central and Appalachian Regions. The major conclusion is that there is considerable regional variation in the size and concentration of the sawmill industry. This variation is affected by the differences in the volume and density of the hardwood resource in different states in the regions.

INTRODUCTION

The hardwood forests within the 16 Central and Appalachian states (Figure 1) are a valuable natural resource. These forests contain 68 percent of the hardwood sawtimber volume in the United States (Powell and others 1993). A high proportion of this sawtimber is in the more desired species such as the select red and white oaks (Luppold and Dempsey 1994). Furthermore, virtually the entire United State's supply of black cherry, black walnut, and sugar maple exists in these states (Powell and others 1993). The large volume and high value of this timber base allow these regions to be a major source of hardwood lumber.

The large variety of climates and growing conditions in the Central and Appalachian states has caused the hardwood resource in these states to be extremely diverse. The four principal hardwood forest types existing in the regions are: aspen-birch, maple-beech-birch, oak-hickory, and oak-pine. There also is considerable local and regional variation in species mix between hardwood stands of similar forest types.

The hardwood industry adapts to local conditions such as volume, quality and type of resource, and transportation costs. Analysis by Luppold (in press) found regional differences in the average size and market concentration within the hardwood sawmill industry. This finding suggests that regional characteristics of the resource may be the cause. The objectives of this paper are to examine the hardwood industry in the Central and Appalachian Regions and to determine how differences in the resource base contribute to long term regional differences in the industry.

THE HARDWOOD LUMBER INDUSTRY

Information concerning the hardwood lumber industry has been incomplete or in error. For instance, U.S. Department of Commerce estimates of hardwood lumber production underestimates the size of the industry by more than 40 percent (Luppold and Dempsey 1994). However, detailed information about the industry is available from primary wood processing directories published by individual states. In this study, the data on the number of mills and average size of mills were primarily developed from these directories and USDA Forest Service records. The specific procedures used to develop the sawmill information presented in this study are outlined in Luppold (in press).

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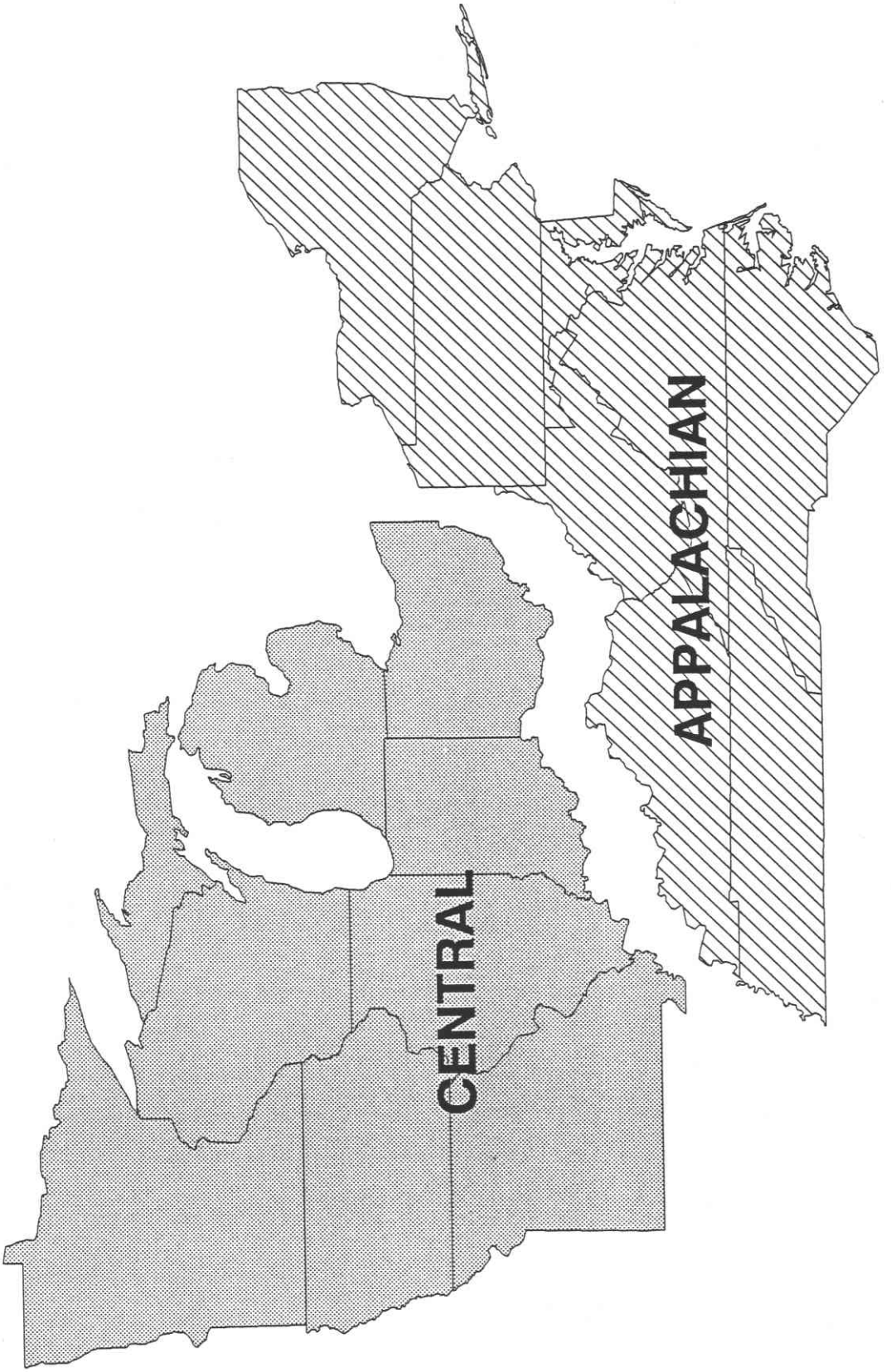

Figure 1. Central and Appalachian hardwood production regions.

Estimates of the hardwood lumber production capacity and average size of sawmills in the 16 Central and Appalachian states are summarized in Table 1.

Table 1. Estimated hardwood sawmill capacity, number of sawmills, and average capacity of sawmills in states in the Central and Appalachian Regions.

Region	State	Year	Estimated capacity of mills (mmbf)	Number	Average production (mmbf)
Central	Wisconsin	92	528	201	2.63
	Ohio	92	524	207	2.53
	Michigan	91	559	247	2.26
	Indiana	93	393	191	2.06
	Iowa	90	70	38	1.97
	Missouri	92	636	375	1.70
	Illinois	93	117	70	1.67
	Minnesota	90	244	240	1.02
	Total region		3,071	1,569	1.96
Appalachian	Maryland	94	179	46	3.90
	Virginia	92	717	239	2.98
	N Carolina	92	616	208	2.96
	W Virginia	92	581	226	2.57
	Tennessee	89	890	353	2.60
	Kentucky	90	936	395	2.37
	New York	91	511	221	2.31
	Pennsylvania	91	1,028	578	1.81
	Total region		5,458	2,266	2.41

In 1991, the sawmills in the states listed in Table 1 produced nearly 8.4 billion board feet of hardwood lumber (Luppold and Dempsey 1994). This volume represents 70 percent of the total United States production in 1991. The state with the largest sawmill capacity is Pennsylvania. Other states with considerable capacity include North Carolina, Virginia, Tennessee, and Kentucky. Relatively little lumber was produced in Iowa, Illinois, and Minnesota.

The 16 states listed in Table 1 contain over 3,800 hardwood sawmills that produce at least 100,000 board feet of hardwood lumber per year. In addition to these mills, there are at least 1,000 mills that produce less than 100,000 board feet per year in the states listed in Table 1. In Minnesota alone there are more than 425 small sawmills that had a combined total production of 11.4 million board feet in 1993. However, the total output of these smaller mills seems to be less than 5 percent of total production in the 16 state area.

The average size of hardwood sawmills varies considerably by state. However, there are some geographic similarities. Virginia and North Carolina have similar hardwood lumber industries when considering the level of production, the number of mills, and the average size of the mill. Average capacities of sawmills in Tennessee, West Virginia, Ohio, and Kentucky are similar. For the most part, sawmill size decreases the farther north and west a state is located in the 16 state area.

The small size of the average sawmills in Pennsylvania masked the fact that some of the largest mills in the 16 state area are in this state. Pennsylvania also has the largest number of hardwood sawmills producing more than 5 million board feet annually. One reason for the low statewide average sawmill size is the large number of Amish sawmills. Another reason is the large number of mills that specialize in railroad crosstie production. Both Amish-owned sawmills and sawmills that produce rail ties tend to produce between 100,000 and 1 million board feet of lumber annually. In 1991, 470 sawmills in Pennsylvania fell in this range of production.

LUMBER PRODUCTION RELATIVE TO THE HARDWOOD RESOURCE

Variation in the size, density, and quality of the hardwood resource between states may help explain regional differences in the hardwood sawmill industry. To understand the relationship between the hardwood sawmill industry and the resource, a comparison of the resource in the 16 states is presented in Table 2.

Table 2. Hardwood sawtimber inventories, timber density, and percent of resource in select species in states in the Central and Appalachian Regions.

Region	State	Sawtimber inventory (mmbf)	Timber density [a] (Million bdf/acre)
Central	Wisconsin	30,394	2.91
	Ohio	28,926	4.00
	Michigan	39,451	3.38
	Indiana	18,946	4.56
	Iowa	5,767	2.98
	Missouri	23,064	1.93
	Illinois	17,782	4.50
	Minnesota	22,693	2.36
Appalachian	Maryland	11,384	5.65
	Virginia	58,295	5.09
	N Carolina	62,541	5.70
	W Virginia	53,886	4.84
	Tennessee	42,963	4.04
	Kentucky	43,996	3.70
	New York	37,991	3.23
	Pennsylvania	59,421	4.20

a/ Since inventory statistics do not separate hardwood timberland from softwood timberland, the timber ratios are based on both hardwood and softwood inventories.

As would be expected, the volume of hardwood lumber produced is correlated with the size of the resource in a specific state. The states within the Appalachian Region have large volumes of hardwood sawtimber and the greatest sawmilling capacities. The states of Iowa, Illinois, and Minnesota have small inventories of hardwood timber and produce small quantities of hardwood lumber. Indiana and Missouri have relatively small inventories of hardwood sawtimber but produce high volumes of hardwood lumber. However, both these states were net importers of hardwood sawtimber during the early 1990's (Hackett and Mayer 1993; Hackett and others 1993). Another state that is a net importer of hardwood sawtimber is Ohio (Widmann and Long 1992).

To measure the relationship between standing timber volume and lumber production capacity, the following equation was estimated using ordinary least squares:

$$Ln(Capacity)_i = b_0 + b1\ Ln(Inventory)_i + b_2\ Importer \quad (1)$$

Where: $Ln(Capacity)_i$ = the natural log of sawmilling capacity for state i,
$Ln(Inventory)_i$ = the natural log of sawtimber inventory for state i,
Importer = a zero one variable that shifts the slope for the states of Missouri, Indiana, and Ohio (equals 1 if net importer)

The double log or multiplicative functional form was used for this equation because it provided a better statistical fit than a linear or other log linear forms. The ordinary least square results presented in Table 3 indicate that a 1 percent increase in sawtimber inventory will, on average, result in a 1.08 percent increase in sawmilling capacity. Because equation 1 was estimated using cross sectional data, the estimated results indicate a long run adjustment by the hardwood lumber industry in a particular state to the resource of that state. Short term increase in capacity due to incremental increases in inventory could be considerably lower than estimated.

Table 3. Ordinary least squares estimates of the relationship between hardwood sawmill capacity and the volume of the hardwood resource and average sawmill capacity and timber density for states in the Central and Appalachian Regions.

Equation[a]	Explanatory variable	Regression coefficient	Student's "t"[b]
(1) Capacity of sawmilling industry:			
Capacity	Intercept	-5.17	3.89
	Inventory	1.08	8.45
	Importer	0.51	2.45
R^2 = .848			
(2) Average capacity of sawmilling industry:			
Avercap	Intercept	0.07	.37
	Density	0.62	4.66
	INIL	-0.45	4.45
R^2 = .772			

a Equations described in text.
b Critical "t" value for P level of 0.05, 1.782.

The significance and sign of the importer slope shifter indicate that the estimated relationship between sawmill capacity and inventory is slightly higher for importing states. The relatively high R^2 and significance of all independent variables indicate that the relationship between sawmill capacity and sawtimber inventory is quite significant.

The size of a sawmill may be affected by the density of the forests on timberlands. This relationship exists because procurement costs increase with distance or as greater quantities of timber are demanded from a finite resource. In states where the resource is spread out over a large area, one would expect to find large numbers of small and intermediate sized mills. In regions where large volumes of timber exist in relatively small areas, one would expect to find large sawmills. As indicated in Tables 1 and 2, the average size of hardwood sawmills is correlated with timber density.

The major exceptions to the correlation between average mill size and timber density are Illinois, Indiana, and Pennsylvania. Both Illinois and Indiana have large agricultural industries that use much of the land base in these states. The timber resource in these states is often located in small patches of land that are unsuitable for farming. Although the densities of these stands are high, the spatial separation of these patches increases procurement costs. As previously mentioned, the small average size of sawmills in Pennsylvania is, in part, the result of Amish ownership and the rail tie industry.

To measure the relationship between average capacity of sawmills in a state and timber density, the following equation was estimated using ordinary least squares:

$$\text{Ln(Avercap)}_i = b_0 + b_1 \text{Ln(Density)}_i + b_2 \text{INIL} \quad (2)$$

Where: Ln(Avercap)_i = the natural log of average capacity of sawmills in state i,
Ln(Density)_i = the natural log of density of the sawtimber inventory for state i,
INIL = a zero one variable that shifts the slope for the states of Indiana and Illinois (equals 1 for IN and IL)

The double log or multiplicative functional form also was used in this equation. The ordinary least square results presented in Table 3 indicate that a 1 percent increase in sawtimber density will, on average, result in a 0.62 percent increase in average sawmilling capacity over the long run. The significance and sign of the INIL slope shifter indicate that average sawmilling capacity in Illinois and Indiana seems to be lower because of the scattered but highly dense hardwood resource in these states. The relatively high R^2 and significance of all independent variables indicate that the relationship between timber density and average sawmill capacity is significant.

SUMMARY AND CONCLUSIONS

The hardwood sawmill industry in the Central and Appalachian Regions is a diffuse group of manufacturers who are dependent on and affected by the diverse forest resource in these regions. Analysis presented in this paper found that the hardwood sawmill industry varies considerably from state to state. Most of the differences in the size of a sawmill industry between states are related to the volume of sawtimber that exists in the state. Furthermore, the size of the average sawmill is dependent on the density of the timber resource in the state.

The relationship between the hardwood resource and the hardwood sawmill industry aids in understanding how the constantly changing hardwood resource will affect future lumber production. In areas of an expanding resource, one can expect greater lumber production. If the resource is allowed to mature to a high density, one can expect larger and possibly more efficient sawmills. However, changes in demand for this timber resource by the pulp and engineered building products industries may change the relationship estimated in this paper. If these alternative users start consuming large volumes of hardwood sawtimber, there is a potential for a structural change within the hardwood industry.

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VARIATION IN PIN KNOT FREQUENCY IN BLACK WALNUT LUMBER CUT FROM A SMALL PROVENANCE/PROGENY TEST

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Abstract: This small study examined the frequency of knots (> 1 growth ring), pin knots (latent or suppressed buds), and pin knot clusters in 4/4 black walnut (*Juglans nigra* L.) lumber from 42 logs, 18 to 21 cm dbh, cut from a 14-year-old provenance/progeny test. Two boards from opposite sides of each log were analyzed for number of knots, pin knots, and pin knot clusters. The boards averaged 1.4 knots per linear meter across all sources with no differences among logs within a source or between the apical and basal halves. From 50 to 56% of the pin knots occurred as 2, 3, or 4 pin knots clustered within 3 cm of each other. Boards from Missouri "super" seedlings had fewer pin knot clusters than boards from nursery bedrun seedlings. On an average, the boards from all sources had 8.1 pin knots or pin knot clusters per linear meter with no differences between the apical and basal halves. In most cases, highly significant differences in pin knot numbers existed among the logs originating from the same seed or seedling sources. Selecting parent trees for desirable nut or timber characteristics did not affect the number of knots, pin knots, or pin knot clusters in their progeny. These results suggest that retention of suppressed buds is controlled by environmental and not genetic factors.

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AUTUMN PREDATION OF NORTHERN RED OAK SEED CROPS

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Abstract: Production and autumn predation of northern red oak acorns was measured over four years in five Pennsylvania stands dominated by this species. Mean annual production was 41,779/acre, of which an average of 7.9% was destroyed by insects or decay following insect attack, and an average of 38.6% was destroyed or removed by vertebrates. White-tailed deer appeared to be the dominant vertebrate consumer of acorns.

INTRODUCTION

Numerous factors contribute to the well-known difficulty of regenerating northern red oak (*Quercus rubra* L.) (Crow 1988), but the frequent lack of seedlings even after good seed crops (Steiner and others 1993) suggests that acorn predation may be a particularly important limitation. Unfortunately, northern red oak's heavily provisioned seed, which plays such a large role in the post-germination growth patterns and ecological response of the species (Kolb and Steiner 1990, Kolb and others 1990), is also an extremely attractive food source for granivorous vertebrates and insects. Losses to seed predators are usually large (Beck 1993).

Despite a fairly large body of literature on the production and loss of northern red oak (NRO) acorns, there is still uncertainty about the relative importance of various acorn consumers because of conflicting results and inconsistent methodology. Many published studies have been based upon inadequate samples of trees, stands, or years (*e.g.*, Korstian 1927), all of which are now known to be large sources of variation not only in acorn yield but also in the relative importance of various sources of acorn loss (Olson and Boyce 1971). In other studies, there has been no accounting of acorns lost entirely from the site, creating an underestimate for some agents of loss and an overestimate of others. This appears to have been the case in the classic and otherwise excellent study by Downs and McQuilkin (1944). In others, the use of acorn traps without parallel counts of loss from ground plots has undoubtedly resulted in underestimates of loss to ground-feeding vertebrates (Christisen and Korschgen 1955). Also, in many studies, rates of predation by insects have been reported on the basis of acorns collected (presumably after some predation by vertebrates) rather than on the basis of total production, and no distinction has been made between insect infestation and failure to germinate because of insect attack. Finally, it is not always clear in published reports whether several-year average losses to predation are means of annual percentages, which give undue weight to the very high proportion of predation in years when acorn production is very low, or represent cumulative loss to various agents as expressed on an annual basis. This study of production and autumn predation of seed crops in NRO in five stands over a four-year period was undertaken to clarify our understanding of this portion of the regeneration process, specifically the fate of fully developed acorns during the months of autumn.

METHODS

The study was performed in five forest stands located over a broad portion of central Pennsylvania, two in the Ridge and Valley region and three on the Allegheny Plateau. Northern red oak was the dominant species in the overstory of each stand, with a mean of 49.3% (range 39.8 to 67.8) of stand basal area. Other common species were *Acer rubrum*

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(mean = 24.8%) and other oaks (15.2%), including *Q. alba*, *Q. prinus*, *Q. velutina*, and *Q. coccinea*. At the beginning of the study, stand mean ages of dominant and codominant NRO ranged from 70 to 123 years. More information on the characteristics of these stands may be found in Steiner and others (1993).

At each research site, a study area of 3.5 to 4.3 acres in size was delineated in October 1990 by marking corner trees. In November of each year from 1990 through 1993, the size of the NRO seed crop was determined by the cap-count method (Shaw 1974) performed in 4.31-ft^2 (0.40 m^2) ground plots. At each study site, approximately 75 quadrats were located at fixed intervals on equally spaced transects that covered the entire study area. Although most caps had fallen by the time counts were made, usually in the second or third week of November, a small number of unfallen caps sometimes remained in the tree crowns. The density of these was estimated by surveying an estimated 10-foot square portion of the canopy directly above each quadrat with binoculars, and ground counts were revised upward as needed. Depending upon year and site, unfallen caps amounted to 0 - 11% of total production. In addition to detached acorn caps, the number of fresh acorns was counted, omitting those that remained firmly attached to their caps, which were typically undersized and considered to have been aborted before full maturity. Fresh acorns were tallied by two categories: 1) intact or 2) partially consumed (by small rodents) at the proximal end of the nut. Acorns with <30% of cotyledons removed were assumed to be viable, subject to germination tests; those with >30% consumed were tallied as having been lost to vertebrate (rodent) predation. Coefficients of variation for acorn and cap counts averaged 12.8% of stand/year means.

Each fresh, detached acorn cap was assumed to represent a viable acorn, excluding losses to vertebrate and insect predation. Differences between counts of caps and counts of fresh acorns were attributed to predation by vertebrates feeding on the ground or in tree tops prior to the November measurements. At the time of production counts in November, samples of fresh acorns were collected from each site for viability analysis. Except where no fresh acorns could be found, sample sizes ranged from as few as 18 to as many as 897, with an average of 170. Sample sizes of fewer than 100 acorns represented *all* the acorns that could be found on the site. Acorns were collected without regard to any external evidence of injury, including insect infestation. Acorn samples were fully hydrated and stored at -1.0°C until February, when they were planted in flats of moist peat moss in the greenhouse for determination of viability. In April or May, after germination had ceased, nongerminants were cut open and classed by apparent cause of death. Germination failure averaged 44.0% in acorns that had been partially consumed by rodents, and this loss was attributed to vertebrate predation. All other germination failures were attributable directly or indirectly to insects (decaying acorns were assumed to have become infected through insect attack [Oak 1993]), and this was the basis for estimated acorn losses to insect predation. Insect-infested acorns that successfully produced a seedling were not counted as lost to predators.

Signs of foraging by white-tailed deer (*Odocoileus virginianus*), especially in years of abundant acorn production, suggested that this species caused much of the loss to vertebrates. To estimate loss by deer *vs.* other vertebrates, a five-strand electrified fence was installed around two noncontiguous plots at one of the research sites on 17-18 August 1993, prior to acorn dissemination. Each plot enclosed about one-fourth of the total study area at the site, and two similar-sized plots were left unfenced. The four plots enclosed almost identical basal areas of NRO. In addition to the normal assessment of acorn production in November, the plots were remeasured five weeks later on 17 December to determine the progressive loss of acorns from fenced *vs.* unfenced plots. Where possible, the statistical significance of differences between means was determined by t-test, analysis of variance, or paired comparisons t-test.

RESULTS AND DISCUSSION

The NRO seed crop, based upon counts of fresh caps, varied from 540/acre to 198,510/acre among the four years of the study and all five stands. Mean annual NRO acorn production for all stands and years was 41,779/acre (Table 1), or 725 per ft^2 basal area of NRO. Every stand exceeded this level of production in at least one year of the four. Table 1 shows mean annual acorn production averaged across stands. The years 1991 and 1992 had relatively low production, and the years 1990 and 1993 had relatively high production. Although stands varied considerably in

mean annual production, as shown in the following tabulation, only the main effect of years was statistically significant ($P < 0.05$) in analysis of variance for total production:

Stand number	Mean annual acorn production, 1990-93 (acorns/acre)
I	32,652
II	23,515
III	78,023
IV	55,200
V	19,505

The contribution of stand to variation in seed crop size was only 18% as large as the contribution of years, even though study sites were separated by as much as 66 miles. In simple terms, there was a tendency for good or poor seed crops to occur simultaneously at all sites. Downs and McQuilkin (1944) found the same result over a seven-year period of study in two stands separated by 100 miles.

Table 1. -- Production and autumn predation of northern red oak acorns from 1990 through 1993, averaged over five Pennsylvania stands. Means within a row that share the same letter are not significantly different at $P < 0.05$.

Year	Pre-November Vertebrate loss	Acorns present in November		Total Production
		Nonviable	Viable	
1990	47,740 a	3,610 b	31,426 a	82,776
1991	9,284 a	1,488 b	1,454 b	12,226
1992	3,790 a	370 b	454 b	4,614
1993	3,626 c	7,780 b	56,094 a	67,500
Mean	16,110 a	3,312 b	22,357 a	41,779

Over all years and stands, the number of viable acorns remaining on the ground in November averaged 22,357/acre (Table 1), or 53.5% of mean annual production. Acorns that were not viable in November as a direct or indirect result of insect infestation averaged 7.9% of production. Of a total of 469 acorns that failed to germinate in viability trials, 66.3% had been attacked by *Curculio* spp. and 4.3% by *Callirhytis*. Most of the remainder were rotten, though a small percentage was infested with acorn moths. Rotten acorns were assumed to be decayed as a result of attack by insects, some of which could also have been *Curculio*. Thus, the percentage loss attributable to *Curculio* could be underestimated.

This study's mean loss of 7.9% of potential production to insects is considerably lower than the 24 - 35% insect infestation rates for NRO acorns reported in many long-term studies (Beck 1977, Downs and McQuilkin 1944, Tryon and Carvell 1962). This is due in part to the methodological discrepancies mentioned in the introduction, especially 1) the expression by others of insect infestation as a percentage of recovered acorns rather than total production and 2) the difference between occurrence of infestation and failure to germinate as a result of infestation. Walters and Auchmoody (1993) found that 53% of acorns attacked by *Curculio* spp. were able to germinate and produce seedlings. In addition, some acorns that had been infested with insects in the present study may have been consumed or cached by vertebrates and thus counted against that source of acorn loss. Linda Gribko (personal communication) found that *Peromyscus* spp. consumed weeviled acorns as readily as sound acorns in the study reported by Gribko and Hix (1993). In any event, it appears that the loss of potential NRO reproduction solely attributable to autumn-feeding insects may be considerably less than commonly supposed. Of course, insects may also contribute to losses at other stages of the reproduction process.

Autumn predation of the seed crop by vertebrates averaged 38.6% of production. Vertebrates were a significantly ($P < 0.05$) more important agent of acorn loss than insects for the four-year study period. Of course, some of the acorns that appeared to have been consumed by vertebrates before November may have been merely cached in a location suitable for germination. However, this fraction still represents a potential loss (most cached acorns are consumed), and in fact total autumn and winter losses to vertebrates (net of spring germination) have consistently exceeded 38.6% (Steiner, unpublished data). Downs and McQuilkin (1944) found an average of 24% of acorns damaged by rodents or birds *at the time of dissemination*. Subsequent predation by vertebrates from ground plots is not clearly described in their report, but in one bumper crop year only 18% of production (five oak species) remained on the ground by mid-winter. Tryon and Carvell (1962) found an average loss to arboreal and ground-feeding vertebrates of 50.2% over a four-year period of study.

Peromyscus leucopus (white-footed mouse) is a common small rodent at all five study sites (Yahner and Steiner, unpublished data). Some evidence suggests that this and a related species are important predators of NRO acorns (Gribko and Hix 1993). According to Verme (1956), *P. leucopus* typically consumes a portion of the cotyledons by feeding through a hole chewed in the cap end of the acorn. In this study, such partial feeding typically amounted to perhaps a little more than one-third of the cotyledons, leaving the embryo and most of the food reserves intact. In fact, Verme found that white-footed mice *always* consumed only a portion of NRO acorns, suggesting that partially consumed acorns may be an accurate measure of predation by this species (other than caching). Such acorns amounted to an average of only 3.9% of mean annual production in this study, and many of those were still capable of germination. This suggests relatively light predation by *Peromyscus*, although additional acorns may have been removed and cached or consumed in burrows or tree cavities. Other information also indicates that this species may have relatively minor importance as a seed predator. Verme (1956) reported an average consumption of 1.8 acorns/day when *Peromyscus leucopus* was fed a diet of only acorns. Using his assumed density of 7 mice/acre (which appears liberal in light of the census data reported by Jensen [1982] and Thorn and Tzilkowski [1991]), this species would have consumed less than 3% of the average annual acorn production in this study by the time production was measured in mid-November.

Other potentially important vertebrate predators that are known to be present at or near the study sites include gray squirrels (*Sciurus carolinensis*), eastern chipmunks (*Tamias striatus*), white-tailed deer, wild turkeys (*Meleagris gallopavo*), and black bears (*Ursus americanus*). The acorns of NRO are generally avoided by blue jays, which can be heavy predators on oaks with smaller acorns (Bossema 1979, Darley-Hill and Johnson 1981, Scarlett and Smith 1991). In 201 hours on site for the measurements reported here, there were 7 gray squirrels sighted, 8 eastern chipmunks, 3 white-tailed deer, and no wild turkeys or black bears. Despite the documented potential for heavy acorn removals by gray squirrels (Thorn and Tzilkowski 1991), population densities of this animal can be quite low (Healy 1988). From my own experience as a squirrel hunter, the infrequency with which squirrels and chipmunks were sighted in this study, despite a sometimes abundant source of food, is rather typical of deep-woods Pennsylvania stands such as these.

In contrast to small mammals, signs of deer foraging were abundant on the study plots, especially when acorn production was high. The study area that was partially fenced to exclude deer in August 1993 provided an estimate of the relative impact of deer foraging on the NRO seed crop. By mid-December, a significantly ($P < 0.05$) higher percentage of the total NRO seed crop of 45,920/acre remained on fenced plots (65.4%) compared to unfenced plots (32.3%).

Furthermore, some undetermined proportion of the acorns lost within fenced areas was also a result of deer feeding, based upon the presence within fenced areas of disturbed leaf litter, masticated pericarps, and droppings. In other words, some of the losses attributed to small vertebrates were actually caused by deer. These observations indicate that deer accounted for at least 48.9%, and probably more, of total removals by vertebrates by mid-December.

Downs and McQuilkin (1944) obtained a similar result in another study area with high deer densities (18 - 32 per square mile). They concluded that deer were far more important than rodents as predators of acorns. Pennsylvania deer densities are higher than this prior to hunting season in late November (37/mi^2 in 1992 according to a personal communication from William Palmer, Pennsylvania Game Commission), but it is impossible to draw direct a direct comparison without knowing the basis for estimating density. Indirect evidence also suggests that deer are more important consumers of acorns than other vertebrates known to feed heavily on acorns, at least in Pennsylvania. Pennsylvania Game Commission estimates of population densities and animal weights indicate that mean deer biomass per forested acre is about 10 times that of gray squirrels and 60 times that of bear and turkeys (William Palmer and William Drake, personal communications). In Missouri, acorns made up 46% of the food consumed by deer during September, October, and November (Christisen and Korschgen 1955). Of course, feeding preferences vary among animals. However, the known tendency of deer to feed heavily on acorns, coupled with their overwhelming biomass abundance (in Pennsylvania) compared to other major consumers, suggests that they must necessarily have a disproportionate influence on acorn predation.

CONCLUSIONS

In five stands with a mean NRO stocking of 49% of total basal area, the annual production of mature acorns averaged 41,779/acre over a five-year period. Yearly variation in seed crop size was about five times as important as stand-to-stand differences. The number of viable acorns remaining on the ground in November averaged 53.5% of mean annual production. Of the 46.5% loss, only 7.9% was caused by insect attack or by fungal or bacterial infection following insect attack. This figure is less than other published figures for insect predation of NRO, which should be regarded as overestimates for reasons explained. The remaining 38.6% of acorn production was removed by vertebrates. In a stand that was partially fenced to exclude deer in 1993, deer predation accounted for a minimum of 48.9% of total acorn removals by vertebrates by mid-December. Other evidence supports a conclusion that deer were the dominant consumers of NRO acorns on these sites.

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PLANTING DEPTH EFFECTS AND WATER POTENTIAL EFFECTS ON OAK SEEDLING EMERGENCE AND ACORN GERMINATION

Wayne A. Smiles and Jeffrey O. Dawson[1]

Abstract: The effects of four planting depths (0, 3, 7, 11 cm) and acorn size on the percentage seedling emergence of red, pin, and black oak were determined. In a complimentary study, the effects of five water potential treatments (0, -.2, -.4, -.6, -1.0 MPa) on the percentage germination of red, pin, and black oak acorns were measured.

Percentage seedling emergence was greatest at 3 and 7 cm planting depths for all species. It was least at both the 0 cm planting depth for all species and the 11 cm planting depth for red and black oak. Percentage emergence was greater for larger acorns within a species, but pin oak, which had the smallest acorns, had the greatest percentage emergence among the three oak species studied. The number of days after planting to time of emergence was negatively correlated with acorn size and positively correlated with planting depth for a species.

Total percentage germination after 60 days in polyethylene glycol osmoticum treatments was greatest for pin oak and least for black oak at all levels. Total percentage germination decreased with decreasing water potentials for all species, with levels of -0.6 and -1.0 MPa virtually arresting germination.

These results indicate that a 3-7 cm planting depth was optimal for these oak species under our experimental conditions. Larger acorns of a given species had a higher percentage emergence than smaller acorns. The germination rate of acorns after exposure to osmotic stress did not coincide with the relative drought hardiness of the parent oak species.

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USE OF PLASTIC FILMS FOR WEED CONTROL

DURING FIELD ESTABLISHMENT OF MICROPROPAGATED HARDWOODS

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Abstract: This study compares the use of plastic films to conventional methods for establishing hardwoods on a recently cultivated old field site using 1-year-old micropropagated plantlets of white ash (*Fraxinus americana* L.) and silver maple (*Acer saccharinum* L.). After one growing season in the field, height of plantlets with all weed control treatments exceeded the height of the plantlets in the non-weeded control. After two growing seasons, plantlets of both species established with plastic films were taller than plantlets with conventional methods of weed control or in the non-weeded plots. No differences were found in plantlet height growth when established with opaque white film, solid black plastic, or porous black plastic. Silver maple plantlets were taller than the white ash plantlets after the first (47 vs 35 cm), second year (114 vs 72 cm), and third years (179 vs 101 cm). Soils under the plastic films had more available moisture in the spring and less available moisture during the summer than the other treatments. During the late fall, soil moisture was again near field capacity in all treatments. After three growing seasons, the solid plastic films had begun to disintegrate. Our results suggest that plastic films can be an effective alternative to chemical or mechanical weed control when establishing hardwoods.

INTRODUCTION

Successful establishment of hardwood seedlings usually requires good site preparation, weed control, and other cultural practices to maintain adequate soil moisture and fertility during the growing season. Lack of weed control during seedling establishment usually results in reduced survival and slowed tree growth (Van Sambeek and Rietveld 1983b). Currently, the most common methods used to achieve adequate weed control have included the use of mechanical cultivation or registered herbicides to control competing vegetation. Unfortunately, hardwoods such as white ash are extremely sensitive to most herbicides (Bey and Williams 1977). In addition, effective herbicides have not been registered for trees such as silver maple.

Recent evidence suggests that micropropagated plantlets are more sensitive to herbicides than conventionally propagated plants (Mudge et al. 1987; Neal et al. 1990). Micropropagated plantlets offer several advantages over the use of seed-produced seedlings for research and field plantings. Use of clonal planting stock can reduce genetic variability and experimental error thus allowing for greater precision in research studies (Libby 1974).

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In addition, micropropagation has the potential to produce a high number of genetically improved plantlets inexpensively in a short period for field plantings.

With declining public acceptance of herbicide use, mulching with plastic or polyethylene films is being reevaluated as a promising technique for enhancing transplant survival and early growth (Windell 1993). Mulches can provide more or less permanent control of weeds during hardwood establishment, as well as more favorable soil conditions through reduced evaporative loss of soil moisture, reduced leaching of nutrients, and reduced soil erosion (Truax and Gagnon 1993). Plastic films are becoming more popular because equipment is available to mechanically lay these materials. Ashby et al. (1992) and Huetteman et al. (1992) both reported successful establishment of silver maple plantlets under droughty conditions using mechanically-laid opaque white-on-black plastic mulch.

Plastic films can have a wide range of optical properties that can markedly modify soil and air temperatures (Ham et al. 1993). Air temperatures immediately above plastic films can be 3 to 5° C higher than 1.5 m above the film. Recently, Trinka and Pritts (1992) recorded higher air temperatures immediately above white film than above black film. Depending on the type of film, midday soil temperatures under plastic films can be as much as 10° C higher than in adjacent uncovered soils (Heiskanen and Raitio 1992). Fritschen and Shaw (1960) reported that soil surfaces under clear films had higher mid-day temperatures than either black plastic or black plastic painted white. They also found temperature differences between bare soil and soil covered with plastic films tended to decrease as soils become drier.

Because plastic films can significantly alter the microenvironment, it is not clear what effect they would have on hardwood establishment, especially the establishment of containerized, micropropagated hardwoods or how the use of plastic films would compare to the conventional methods of weed control. The purpose of our study was to evaluate plantlet growth of white ash and silver maple and to compare changes in soil moisture potential and soil temperature when using three different plastic films and the more traditional methods of weed control for establishing hardwoods.

METHODS

The research planting was established on an old field site at the Southern Illinois University's Horticulture Research Center near Carbondale, IL. The study area measured approximately 25 x 50 m and was located along a mid-slope position with an eroded Hosmer silt loam (fine-silty, mixed, mesic Typic Fragiudalfs, pH 5.5). The tall fescue hay field, which previously was mowed annually, was plowed in April 1992 and cultivated twice before laying out the study in mid-May 1992. The research planting consisted of five rows, 4 m apart, oriented from west to east along the south-facing slope. Each row was treated as a block and marked off into six 1.0 m-wide by 7.5 m-long plots.

Six treatments were randomly assigned within each block: (a) non-weeded check, (b) clean cultivation, (c) herbicide treatment, (d) porous black plastic film, (e) solid black plastic film, and (f) solid white plastic film. The porous black film (VisPore tree mats) consists of approximately 60 microfunnels per cm^2 designed to allow rainwater and air to penetrate the 2.5 mil-thick plastic film. The solid plastic film was a highly reflective, opaque white-on-black 1.25 mil-thick polyethylene film that could be installed either white side up or black side up. The solid white film was laid with a tractor-pulled mulch laying machine in mid-May 1992. The two black films were manually laid in trenches formed with the mulch laying machine. When laid, the outside 15 cm of the 1.3 m wide plastic films was buried resulting in 1.0 m wide experimental plots.

The clean cultivated plots were hand-hoed twice in 1992 and rototilled twice in 1993 and 1994. The herbicide plots were sprayed with 55 ml/ha sulfometuron (Oust) and 6.4 L/ha glyphosate (Roundup) in June 1992. Glyphosate only at 6.4 L/ha was used in August 1993 and June 1994. Hardwoods were protected from direct application of herbicides using a stove pipe. The area between each row of plots was mowed twice in 1992, 1993, and 1994. The plantlets were protected from deer browse with a portable 3-wire electric fence. Some rabbit damage occurred on the smaller plantlets during the 1992-1993 winter.

Two white ash and three silver maple plantlets were randomly assigned to each plot and planted 1.5 m apart. Silver maple clones were planted as dormant containerized stock overwintered in a cooler and as plantlets in leaf from a shadehouse. White ash clones were planted as containerized plantlets in leaf from a shadehouse. Height of the white ash and silver maple plantlets averaged 24 and 28 cm, respectively, at the time of planting. Plantlets were established using KBC planting bars on 27 May 1992. A 9 gram starter (Agriform) fertilizer tablet (22 N - 8 P_2O_5 - 2 K_2O with 3% Ca, 1% S, 0.5% Fe, and 0.1% Zn) was placed in the first closure hole of each plantlet. Because the soils were relatively dry at the time of planting, approximately 2.5 cm of irrigation water was applied to the entire area on 29 May 1992. The plantlets received no subsequent fertilization or irrigation.

A soil moisture tensiometer was installed in each plot from June to October, 1992 and 1993. The 50-cm-long tensiometers were installed at a 45° angle with the ceramic cup approximately 30 cm deep located midway between a silver maple and a white ash plantlet. Tensiometers were nominally read twice a week during the growing season until soil moisture returned to field capacity in the fall. Because fine silica was used to reinforce the soil-ceramic cup interface, soil water potential readings between -80 and -90 kPa could be reliably obtained in most cases. To compare effects of vegetation management treatments on soil moisture potential during the growing season, we quantified the number of days each month soil moisture potential was less than -33 kPa (little or no moisture stress), -34 to -67 kPA (moderate moisture stress), and less than -67kPA (severe moisture stress) for each plot. These values are based on previous research growing hardwood seedlings under different moisture regimes (Rink and Van Sambeek 1985, 1987a).

One soil thermocouple was also installed 15 cm deep midway between two plantlets near the middle of each plot. Soil temperatures at 0400 and 1600 hours were recorded daily for a two week period from 26 August to 12 September 1992. Daily minimum and maximum air temperature and rainfall were recorded at the Horticulture Research Center weather station located 0.5 km southeast of the planting site.

Plantlet height was measured at planting and two or three times during the first three growing seasons. The amount of winter dieback and length of the spring flush were measured during May 1993. The number and length of all lateral shoots were determined after the first growing season. During July and/or August of the first and second growing seasons, the terminal shoots of each plantlet were examined to determine if they were still actively producing new growth.

Treatment means by species were computed from individual plantlet data and subjected to analysis of variance (ANOVA) to test for effects of weed control treatments, species of plantlets, and their interaction. We used a randomized complete block with split plots design with weed control treatments as main plots and species of plantlets as the subplots. Comparison of treatment or species means were based on Fisher's protected LSD (1% and 5% t-test values). The five planned allowable orthogonal contrasts were comparison of (1) non-weeded check against the five other treatments, (2) conventional weed control (herbicide and cultivation) against the plastic films, (3) herbicide against cultivation, (4) solid plastic films against porous plastic film, and (5) solid white-on-black against solid black-on-white film.

RESULTS

Plantlet Response to Vegetation Management Practices

Plantlet survival averaged 90 percent after the first growing season with no statistical differences between the non-weeded control or plots where vegetation was controlled chemically, mechanically, or with plastic films (Table 1). All the white ash plantlets survived while mortality exceeded 20% for silver maple plantlets after the first growing season. Most of the dead silver maple plantlets were of a single clone planted as containerized stock in leaf. The spring application of sulfometuron and glyphosate killed a few additional silver maple plantlets. Little additional mortality has occurred for either species during the second and third growing seasons.

Table 1.--First year mortality, lateral shoot growth, winter dieback, and spring flush of white ash and silver maple plantlets under six different weed control practices.

Treatment and species	Mortality	Lateral shoot Number	Lateral shoot Length	Plantlets with dieback	Spring flush 1993
	- % -		- cm -	- % -	- cm -
SPECIES MEANS:					
White ash	0	1.6	11.7	35	19
Silver maple	21	1.9	27.0	77	34
Significance[a]:	**	ns	**	**	**
TREATMENT MEANS:					
Check	20	0.7	1.5	67	24
Cultivate	17	1.4	15.9	65	26
Herbicide	10	1.3	16.2	72	34
Black Film	7	3.1	34.6	45	27
White Film	6	2.1	23.1	38	26
Porous black	5	1.9	24.7	52	29
Significance[a]:	ns	*	*	*	ns
5% t-test value:		2.4	17.5	28	
1% t-test value:		3.2	23.9	40	
Tested Contrasts:					
Check vs Others:		*	**	ns	
Conv. vs Plastic:		*	*	*	
Cultv. vs Herb.:		ns	ns	ns	
Solid vs Porous:		ns	ns	ns	
Black vs White:		ns	ns	ns	

[a] Nonsignificant (ns) or significant effects at the 5% (*) or 1% (**) according to F-test for 1 and 24 df (species) and for 5 and 20 df (treatments).

Plantlets established with plastic film for vegetation control had significantly more lateral shoots than plantlets without plastic films after the first growing season (Table 1). In addition, these plantlets also had the greatest cumulative length of lateral shoots. Silver maple plantlets produced over twice the length of lateral shoots as the white ash plantlets. All the plantlets receiving some type of weed control had significantly more and longer cumulative lenght of lateral shoots than plantlets in the non-treated check plots.

Terminal shoot dieback or winter-kill occurred on many of the silver maple and white ash plantlets in all treatments (Table 1). The lowest incidence of dieback occurred in plots with plastic mulch (38 to 52%) and the highest under the conventional cultural practices (65 to 72%). Winter dieback on the silver maple plantlets averaged 3.6 cm compared to only 1.7 cm on the white ash plantlets. No differences in the amount of winter dieback were found with respect to weed control practice.

The silver maple plantlets outgrew the white ash plantlets during the first three growing seasons (Table 2). After the first growing season, height of plantlets in all weed control treatments exceeded that of the plantlets in the non-weeded check treatment. This also remained true throughout most of the second growing season. By the end of the second growing season, plantlets established with plastic films were taller than those established with conventional methods of weed control.

Table 2.--Height of white ash and silver maple plantlets through three growing seasons under six different weed control practrices.

Treatment and Species	Oct. 1992	May 1993	July 1993	Oct. 1993	July 1994	Oct. 1994
	cm					
SPECIES MEANS:						
White ash	34	54	64	73	102	101
Silver maple	46	79	92	114	170	179
Significance[a]:	**	**	**	**	**	**
TREATMENT MEANS:						
Check	23	43	48	49	91	89
Cultivate	46	67	79	96	140	140
Herbicide	37	74	80	84	126	144
Black Film	47	73	89	111	148	150
White Film	47	71	83	105	143	144
Porous black	51	79	95	118	55	158
Significance[a]:	**	**	**	**	**	**
5% t-test value:	11.5	12.6	16.2	16.4	25.9	27.8
1% t-test value:	15.7	17.21	22.1	22.4	35.3	37.8
Tested Contrasts:						
Check vs Others:	**	**	**	**	**	**
Conv. vs Plastic:	ns	ns	ns	**	**	ns
Cultv. vs Herb.:	ns	ns	ns	ns	ns	ns
Solid vs Porous:	ns	ns	ns	ns	ns	ns
Black vs White:	ns	ns	ns	ns	ns	ns

[a] Nonsignificant (ns) or significant effects at the 5% (*) or 1% (**) according to F-test for 1 and 24 df (species) and for 5 and 20 df (treatments).

During July of the first growing season, over 90 percent of the plantlets in the plots with plastic film had actively elongating shoots while only 65 percent of the plantlets in the non-weeded control plots had actively elongating terminal shoots. In addition, 88 percent of the silver maple plantlets had elongating shoots compared to 70 percent of white ash plantlets. Similar observations were made in July of the second growing season. The percentage of plantlets with some type of weed control having elongating shoots decreased to 50 percent or less during August of the second growing season; however, only 12 percent of the plantlets in the non-weeded plots still had elongating terminal shoots. In August, nearly four times more silver maple plantlets (57 percent) still had elongating shoots compared to the white ash plantlets (15 percent).

Effectiveness of Vegetation Management

During the first growing season, the non-weeded check plots quickly developed a heavy cover of broadleaved annuals in the spring which was followed by a dense stand of foxtail (*Setaria* spp.). During the second and third growing seasons, the non-weeded check plots had a dense cover of broadleaved annuals, biennials, and perennials with fewer grasses. Mechanical cultivation generally provided short term control of competing vegetation and needed to be repeated several times during each growing season. Sulfometuron and glyphosate effectively controlled vegetation throughout the first growing season and into the second growing season before a second application of glyphosate was applied to control emergent vegetation.

Essentially no vegetation grew within the 1 x 7.5 m plots when covered with porous black, solid black, or solid white plastic film for the first two growing seasons. What vegetation there was grew mostly through the planting hole and not the plastic film. The solid black and solid white films began to disintegrate after the second growing season. The thicker porous black film remained largely intact throughout the third growing season.

The method of weed control altered soil moisture depletion and recovery during both the first and second growing seasons (Fig. 1). In general, soil moisture potential decreased most rapidly in the check plots covered with dense vegetation. Soil moisture potentials declined more slowly under the plastic film than in the other treatments; however, they were also the last to return to field capacity at the end of the growing season. All three plastic films were laid with the edges slightly higher than the center of the plots where the trees were planted. During thunderstorms, precipitation tended to initially sheet across all three films and enter through the planting holes at the base of each plantlet. Water also readily penetrated the porous black film once the micropores were wetted.

During the spring of the first and second growing seasons, plantlets in plots with weed control experienced more days with little or no soil moisture stress (soil water potentials from 0 to -33 kPa) than the non-weeded control (Table 3). Near the beginning of the first growing season, plots with the plastic mulch had more days with adequate soil moisture than the plots with conventional weed control; however, no differences were found during the fall of the first or second growing season.

Table 3.--Number of days plantlets experienced little or no moisture stress[a] during the first and second growing season under six different weed control practices[b].

Weed control	First growing season			Second growing season			
Treatment	July	Aug.	Sept.	June	July	Aug.	Sept.
	------ days --------			--------- days ---------			
Check	0	1	23	9	0	0	9
Cultivate	2	0	18	12	14	5	16
Herbicide	2	1	27	12	6	1	12
Black film	6	1	17	19	17	7	17
White film	5	1	20	19	10	1	12
Porous black	3	0	26	24	12	0	8
Significance[c]:	*	ns	ns	*	*	ns	ns
5% t-test value:	4.2			10.6	8.9		
1% t-test value:	5.8			14.6	12.2		
Tested Contrasts:							
Check vs Others	*			*	**		
Conv. vs Plastic:	*			*	ns		
Cultv. vs Herb.:	ns			ns	ns		
Solid vs Porous:	*			ns	ns		
Black vs White:	ns			ns	ns		

[a] Soil moisture potentials between 0 and -33 kPa.

[b] Monthly precipitation from May through September was 3.30, 2.85, 6.71, 6.58, and 17.63 cm, respectively, in 1992 and was 12.52, 12.24, 8.97, 6.25, and 22.83 cm, respectively, in 1993.

[c] Nonsignificant (ns) or significant effects at the 5% (*) or 1% (**) according to F-test for 1 and 20 df (species) and for 5 and 20 df (treatments).

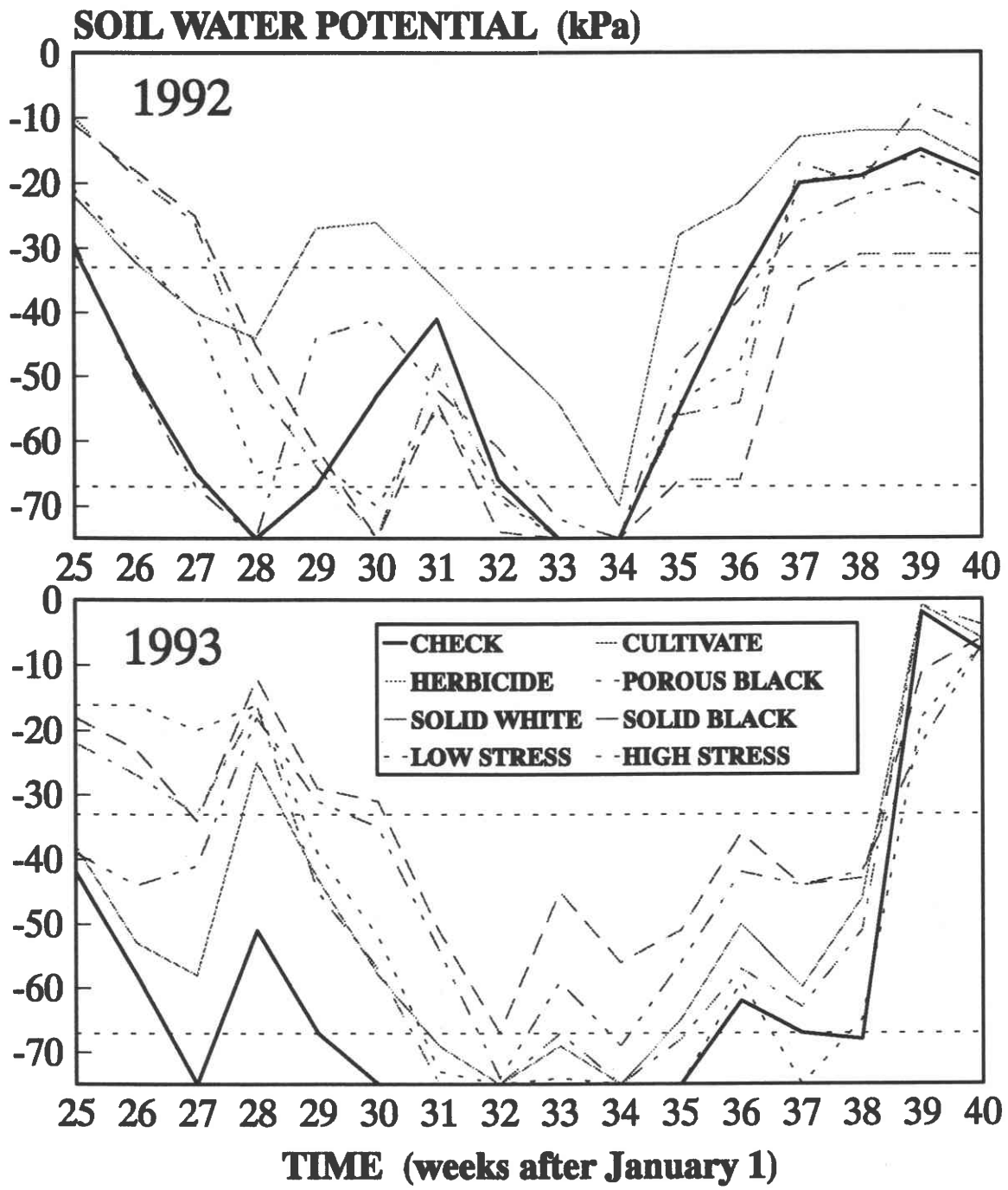

Figure 1.--Soil moisture potential under six weed control treatments for the first and second growing seasons after plantlet establishment.

During both the first and second growing seasons, plantlets in all treatments experienced 18 or more days with severe moisture stress (soil moisture potential less than -67 kPa) (Table 4). The number of days with severe moisture stress was highest in the non-weeded check plots at the beginning of each growing season. In general, the greatest number of days with severe moisture stress was least in the herbicide-treated and cultivated plots, intermediate in plots covered with plastic mulches, and highest in the check plots.

As found in other studies, the type of weed control can significantly alter the daily changes in soil temperature under both dry and wet soil conditions (Table 5). During a week of relatively clear days in late August when soil moisture potentials were under -67 kPa, the greatest daily change in soil temperatures occurred under the solid black plastic film. The same pattern was seen during a week of relatively cloudy days in early September; however, the magnitude of the daily changes under solid black plastic was not as great. The micropores in the porous black plastic may have allowed greater heat exchange, so that daily soil temperature changes were more nearly equal those for daily changes in other plots without dense vegetation. The daily change in soil temperature was least for the solid white plastic film under both clear and cloudy weather conditions. The lowest soil temperatures were found in the non-weeded control where the dense vegetation significantly reduced the amount of solar radiation reaching the soil surface.

DISCUSSION

Growth rates of the micropropagated silver maple trees were similar to that reported in other studies where height tended to double each year during the first three years (Ashby et al. 1992). This is the first report on growth rates of field-planted white ash plantlets; however, their height growth was similar to what would be expected from small bare-root seedlings of white ash.

The reduced incidence of terminal shoot dieback on plantlets in the plots using plastic films for weed control may have long-term consequences on tree form and development. Both silver maple and white ash have opposite phyllotaxy and frequently develop multiple stems following damage to the terminal bud. Because multiple stems are less desirable than a single stem for most forestry applications, the use of plastic films should have beneficial growth effects well beyond plantation establishment.

Past studies on black walnut have shown a strong correlation between the cumulative length of lateral shoots after the first growing season and rapid terminal shoot growth the second growing season (Rink and Van Sambeek 1987b). This pattern also appears to be true for white ash and silver maple, because the plantlets with the greatest cumulative lateral shoot growth after the first year tended to be the tallest after the second year.

The amount of soil warming under the solid black plastic was significantly higher than under the porous black plastic. Rapid heat exchange through the micropores appears to another important property of the porous black plastic in addition to allowing moisture to penetrate the plastic film. The changes in soil temperature we found under the solid black plastic were not as great as reported by other researchers. The rough soil surface under the black plastic or relatively dry soil conditions may have reduced the heat transfer (Ham et al. 1993, Heiskanen and Raitio 1992).

Although no root excavations were done, results from an unpublished study suggest that lateral root elongation on field-planted silver maple plantlets occurs very rapidly in moist soil and should extend beyond the edges of the plastic film late during the first growing season. With good weed control and adequate soil moisture, lateral root extension on black walnut seedlings has been shown to exceed 1.0 m after one growing season (Van Sambeek and Rietveld 1983a). The percentage of plantlets with elongating shoots in late summer and height growth of both silver maple and white ash were higher for plantlets established with plastic films than without plastic films. Increased plantlet growth early in the growing season may have resulted from a longer period without significant soil moisture stress. The solid plastic films apparently conserved the soil moisture by reducing the amount of evaporative losses when compared to changes found in soil water potentials found in the herbicide or cultivation treatments.

Table 4.--Number of days plantlets experienced severe moisture stress[a] during the first and second growing season under six different weed control practices[b].

Weed control	First growing season			Second growing season			
Treatment	July	Aug.	Sept.	June	July	Aug.	Sept.
	------- days -------			-------- days ---------			
Check	17	20	1	3	20	29	9
Cultivate	13	20	0	4	2	20	5
Herbicide	10	16	0	4	6	24	5
Black film	10	25	4	0	0	16	2
White film	15	22	2	1	8	27	8
Porous black	11	22	2	0	5	27	12
Significance[c]:	*	ns	ns	ns	**	ns	**
5% t-test value:	9.8				8.2		5.8
1% t-test value:	13.4				11.3		8.1
Tested Contrasts:							
Check vs Others:	*				**		ns
Conv. vs Plastic:	*				ns		ns
Cultv. vs Herb.:	ns				ns		ns
Solid vs Porous:	ns				ns		**
Black vs White:	ns				*		*

[a] Soil moisture potentials less than -67 kPa.

[b] Monthly precipitation from May through September was 3.30, 2.85, 6.71, 6.58, and 17.63 cm, respectively, in 1992 and was 12.52, 12.24, 8.97, 6.25, and 22.83 cm, respectively, in 1993.

[c] Nonsignificant (ns) or significant effects at the 5% (*) or 1% (**) according to F-test for 1 and 20 df (species) and for 5 and 20 df (treatments).

Table 5.--Soil temperature 15 cm below soil surface at 0400 and 1600 hours under six different weed control practices from 28 August to 12 September 1992.

Weed control	28 August to 4 September[a]			5 September to 12 September[b]		
Treatment	0400	1600	Diff.	0400	1600	Diff.
	------°C -----			----- °C -----		
Check	16.2	17.6	1.33	14.9	15.6	0.71
Cultivate	17.3	19.0	1.68	15.7	16.8	1.12
Herbicide	17.4	19.0	1.56	15.8	16.9	1.14
Black film	16.4	20.1	3.71	15.5	18.4	2.83
White film	16.6	17.1	0.52	15.9	16.2	0.31
Porous black	16.2	18.3	2.06	15.2	16.8	1.64
Significance[c]:	**	**	**	**	**	**
5% t-test value:	0.26	0.24	0.34	0.33	0.53	0.57
1% t-test value:	0.35	0.33	0.46	0.44	0.72	0.77
Tested Contrasts:						
Check vs Others	**	**	**	**	**	**
Conv. vs Plastic:	**	**	**	*	ns	*
Cultv. vs Herb.:	ns	ns	ns	ns	ns	ns
Solid vs Porous:	*	**	ns	**	*	ns
Black vs White:	*	**	**	*	**	**

[a] Clear days with an average air temperature of 21.9 °C.

[b] Cloudy, rainy days with an average air temperature of 23.4 °C.

[c] Nonsignificant (ns) or significant effects at the 5% (*) or 1% (**) according to F-test for 5 and 30 df.

Preliminary estimates suggest costs to establish hardwood seedlings using repeated mechanical cultivation for only one year will exceed that for laying plastic films. Costs for applying herbicides for two years are comparable to laying plastic films. Although thicker plastic films will cost slightly more, the cost should be recovered quickly through improved tree growth if the plastic film provides three or more years of weed control. Technologies already exist for manufacturing plastic films capable of giving three or more years of weed control under continual exposure to sunlight. Conversely, soil can be used to partially cover the plastic film or higher density plantings can be established that will result in more rapid canopy closure and prolong life of the plastic mulch. Future research is needed on developing the most appropriate methods for laying plastic films for forestry applications and their effects on tree growth.

SUMMARY

We found the use of plastic films to control competing vegetation to be as effective as the conventional methods currently used when establishing hardwood plantings. Plastic films slowed changes in soil moisture potential during the spring, thus prolonging the period before plantlets were exposed to moisture stress. Soil moisture potential returned to field capacity relatively quickly under all treatments late in the fall. Overall the best tree growth occurred in plots where competing vegetation was controlled with plastic films. Because the plastic mulches remained intact for 2 or 3 years, they also reduced maintenance costs following plantation establishment. We expect with closer spacings and more rapid accumulation of leaf litter the durability of plastic mulches can be further increased.

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PROTECTION OF TREE SEEDLINGS FROM DEER BROWSING

Jeffrey S. Ward and George R. Stephens[1]

Abstract: Browsing by large deer herds has seriously impaired successful regeneration on some Connecticut forests. Six plots were established in 1990 to examine the effectiveness of 5 deer browsing protection devices for 5 tree species. Protective devices included plastic mesh sleeves (60-cm), Reemay (spunbonded polypropylene) sleeves (60-cm), Tubex tree shelters (120 and 180-cm), and Corrulite tree shelters (120-cm) to be compared to unprotected controls. Species included eastern white pine, eastern hemlock, Norway spruce, northern red oak, and black walnut. After 3 growing seasons, seedlings within tree shelters were significantly taller than seedlings protected by plastic mesh and Reemay sleeves, except for eastern hemlock. Tree shelters stimulated growth of hardwood seedlings more than conifer seedlings. Mortality was lower for seedlings protected by tree shelters than for other treatments. Unprotected black walnut seedlings were actually shorter after 3 yr than when planted because of browsing. The Reemay sleeves and plastic mesh were not durable; most were damaged by weather or animals. The mesh caps placed over tree shelters to prevent bird mortality distorted many trees emerging from tree shelters. Browse damage was observed on trees growing out of 120-cm tree shelters.

INTRODUCTION

Heavy browsing by large deer herds has seriously impaired natural regeneration on some forests in Connecticut. The deer herd in Connecticut has increased from an estimated 19,000 in the mid-1970s (Anderson 1984) to over 50,000 in 1993 (Kilpatrick 1993). Deer browsing has become a problem throughout much of the central hardwood region (Marquis 1977, Kelty and Nyland 1983, Kittredge and others 1992). Deer browsing is especially acute in parks and natural areas where hunting is prohibited and herd sizes are artificially high because of the lack of natural predators (Miller and others 1992, Girard and others 1993).

Early work showed that hardwood growth was increased by protection from deer browsing using mesh tubes (Marquis 1977). Tree shelters (solid plastic tubes) increase hardwood tree growth by providing protection from browsing and providing favorable growing conditions. Tuley (1985, cited from Potter 1988) discovered that early oak height growth in England was dramatically increased by using plastic tubes. Tree shelters have increased height growth of northern red oak (Lantagne and others 1990, Lantagne 1991, Minter and others 1992, Kittredge and others 1992, Smith 1993, Walters 1993), and black walnut (Ponder 1991).

Alternatives to tree shelters include fencing the planting area and repellents. Deer fences are effective in reducing browse damage (McCormick and others 1993, George and others 1991), but may require frequent maintenance inspections (George and others 1991) and also increase growth of less desired species such as red maple (McCormick and others 1993). Although repellents have reduced deer browsing of orchards (Conover and Kania 1988, Swihart and Conover 1988, Byers and Scanlon 1987), nurseries (Conover 1984), and Christmas tree plantations (Raymond[2]); control is inconsistent (Conover 1984, Swihart and Conover 1988) and needs to be applied several times a year (Conover 1987, Raymond[2]). Clearly, repellents are inadequate for forest plantations. This report will examine how various browsing protection methods affect growth and survival of seedlings.

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METHODS

In spring 1990 six study sites were established: 3 at Mohawk State Forest and 3 at Lake Gaillard. Both forests are actively managed for timber and hunting is prohibited. Consequently, large deer herds have developed which prevent natural regeneration and destroy artificial forest plantations. Study sites on Mohawk State Forest were recent red pine clearcuts. Two Lake Gaillard study sites were former pasture with scattered woody regeneration, the other was a newly abandoned corn field. All plots were cleaned with chainsaw and machete prior to planting and 2 years later.

Northern red oak (*Quercus rubra*), eastern white pine (*Pinus strobus*), and Norway spruce (*Picea abies*) were planted at both forests. Additionally, eastern hemlock (*Tsuga canadensis*) was planted at Mohawk State Forest and black walnut (*Juglans nigra*) was planted at Lake Gaillard. Seedling height (cm) and root collar diameter (mm) were measured prior to planting. Mean root collar diameters were: black walnut-7.1 mm, northern red oak-5.8 mm, eastern white pine-5.7 mm, Norway spruce-3.6 mm, and eastern hemlock 2.6 mm. Seedlings were stratified by root collar diameter before assignment to treatments. At each of the six sites 8-20 seedlings received each treatment.

The 6 treatments included: 60 cm high plastic mesh sleeve supported by a bamboo stake, 60 cm high spunbonded polypropylene (Reemay[3]) sleeve supported by a bamboo stake, 120 cm Tubex[3] tree shelter, 180 cm Tubex[3] tree shelter, 120 cm Corrulite[3] tree shelter, and an unprotected control. Both bamboo and wood stakes were untreated. Mesh caps provided were placed over all tree shelters to prevent songbird entry into the tubes.

Tree heights (nearest cm), browse damage, and any distortions of the terminal leader were measured at the end of each growing season (~15 September). Browse damage was noted at the beginning (~1 December), middle (~15 February), and end of winter (~1 April). Damage to protective devices was noted during each field check. Failure of protective device was defined as a fallen tree shelter or sleeve, removal of mesh or Reemay sleeve by animals, or disintegration of sleeve to degree that terminal leader was not covered.

Deer density was estimated by the pellet-group count technique (Neff 1964) in 1991 at both forests and 1992 at Lake Gaillard. Five of the study sites were separated by at least 5 km to increase probability that each study area was browsed by distinct deer herds. There was a single pellet-group survey for the 2 study areas at Mohawk State Forest which were only 1 km apart. At each study site 37 to 50 1/247 ha circular plots were randomly located within 1 km of each plot center. Rabbit pellet groups were also counted.

Tukey's HSD test (SYSTAT 1992) was used to test differences in height growth among treatments by size class and year. Chi-square statistics were used to determine whether cumulative browse differed among treatments and whether cumulative browse of unprotected seedlings differed among species. Chi-square statistics were used to determine whether mortality and cumulative failure differed among treatments by species. Preliminary analysis found no difference in growth, mortality, browse, or failure among tree shelter types. Therefore, data for the 3 tree shelter types were combined. Differences were considered significant at $P \leq 0.05$.

RESULTS

Deer density was estimated to be 18/km² at both Mohawk State Forest study areas in 1991. Estimates of deer density at Lake Gaillard ranged from 17-31/km² in 1991 (mean 24/km²) and from 13-24/km² in 1992 (mean 18/km²). At least one pellet-group was found on 62% of sample plots in 1991 at Lake Gaillard; 17% of sample plots had at least one rabbit pellet-group. At Mohawk State Forest, 49% and 26% of sample plots had at least one deer and rabbit pellet-group, respectively. The 1992 survey found deer pellet-groups on 47% of plots and rabbit pellet-groups on 12% of plots.

[3]Use of tradenames does imply endorsement by the Connecticut Agricultural Experiment Station.

Heights of all species, except eastern hemlock, were significantly greater after 3 growing seasons when protected by tree shelters than when unprotected or protected by sleeves (Table 1). The increased height of seedlings protected by tree shelters was significant after 1 growing season. Unprotected black walnuts were actually smaller after 3 growing seasons than when planted. Seedlings protected by sleeves were not significantly taller than unprotected seedlings after 3 growing seasons, except for black walnut protected by mesh sleeves. Hardwoods clearly responded better to tree shelters than conifers. Relative to unprotected controls, trees in tree shelters were taller; black walnut, 215%; northern red oak, 145%; white pine, 70%; Norway spruce, 30%; and hemlock, 20%.

Table 1. Seedling height (cm) at end of growing season by species, treatment, and years since planting.

		Years since planting		
	Initial	1st	2nd	3rd
Northern red oak				
Control	31.1 a*	28.9 a	32.6 a	45.1 a
Mesh	30.3 a	32.0 a	41.2 a	50.5 a
Reemay	31.8 a	31.2 a	38.1 a	50.2 a
Tubes	30.2 a	37.4 b	71.2 b	110.6 b
Black walnut				
Control	34.0 a	38.3 a	38.8 a	28.2 a
Mesh	35.4 a	41.9 a	45.6 b	48.3 b
Reemay	34.1 a	39.7 a	42.0 ab	40.1 ab
Tubes	36.1 a	46.8 b	65.5 c	89.0 c
Eastern white pine				
Control	21.5 a	23.0 a	31.0 a	51.8 a
Mesh	22.1 a	23.9 ab	38.2 b	57.8 a
Reemay	21.8 a	23.6 a	37.4 b	57.4 a
Tubes	21.5 a	25.6 b	51.0 c	88.3 b
Eastern hemlock				
Control	19.6 a	18.3 ab	21.8 ab	40.4 ab
Mesh	19.6 a	17.2 a	28.0 b	35.9 a
Reemay	18.9 a	17.5 a	18.0 a	28.1 a
Tubes	19.1 a	20.2 b	34.7 c	48.5 b
Norway spruce				
Control	21.6 a	25.2 ab	29.9 a	47.2 a
Mesh	22.3 a	24.1 a	31.3 a	46.7 a
Reemay	22.6 a	24.9 ab	31.0 a	45.8 a
Tubes	22.4 a	26.3 b	40.6 b	61.4 b

*Column values for each species followed by the same letter do not differ significantly at $P \leq 0.05$.

Table 2. Cumulative mortality[a] (%) by species, treatment, and years since planting and original sample size in 1989.

	Years since planting			Sample
	1st	2nd	3rd	size
Northern red oak				
Control	6	12	8[a]	36
Mesh	7	8	14	72
Reemay	3	3	8	72
Tubes	0	8	2	108
Black walnut				
Control	0	0	8	24
Mesh	0	0	0	48
Reemay	2	4	6	48
Tubes	0	12	0	72
Eastern white pine				
Control	28	32	37	60
Mesh	29	36	38	120
Reemay	23	32	35	120
Tubes	24	26	28	180
Eastern hemlock				
Control	30	50	57	30
Mesh	27	47	44	60
Reemay	28	27	40	60
Tubes	4	19	21	90
Norway spruce				
Control	8	30	34	50
Mesh	17	28	29	100
Reemay	11	32	36	100
Tubes	10	25	27	150

[a]Cumulative mortality may decrease because some trees resprouted following aboveground dieback.

Total mortality after 3 growing seasons was not independent of treatment for all species (Table 2). Mortality of seedlings protected by tree shelters was much lower than other treatments. Again, there was a difference between hardwoods and conifers. Mortality of conifers protected tree shelters was reduced by about one-third compared with other treatments. In contrast, northern red oak mortality was reduced by at least 75% and there was no mortality of black walnut protected by tree shelters.

Browse of terminal buds was not independent of treatment ($\chi^2 = 302.3$, d.f.= 3, $P \leq 0.001$). Cumulative browse of seedlings protected by tree shelters was 75-89% lower than for other treatments (Table 3). Sleeves reduced browsing by 34-56% relative to unprotected seedlings. Browse of seedlings within tree shelters was caused by either mice

gnawing through tubes and then on seedlings or deer browsing on seedlings which had grown out of 120-cm tree shelters.

There was a significant difference among species in the percentage of unprotected seedlings which were browsed ($\chi^2 = 27.7$, d.f.= 4, $P \leq 0.001$). Terminal buds of hardwoods were browsed much more frequently than conifers. Ninety-two percent of both unprotected northern red oak and black walnut were browsed compared with 57% of white pine, 54% of Norway spruce, and 47% of eastern hemlock.

The principal reason for higher browse levels of seedlings protected by mesh and Reemay sleeves was a higher failure level compared with tree shelters (Table 4). There was a significant difference of failure rate among protection devices ($\chi^2 = 213.4$, d.f.= 2, $P \leq 0.001$). By the end of the winter after the 3rd growing season fewer than half of seedlings protected by sleeves were still protected, compared with 75% of seedlings protected with tree shelters.

Distorted terminal growth of unprotected seedlings was rare, only 1.5% of total (Table 5). The proportion of seedlings (combined species) with distorted tops was not independent of treatment ($\chi^2 = 68.1$, d.f.= 2, $P \leq 0.001$). The proportion of seedlings with distorted tops was not independent of treatment for all species except black walnut. Distortion rates were approximately 3 times higher for seedling protected by sleeves than by tree shelters. Reemay sleeves caused more distortions than mesh sleeves for all species except Norway spruce.

Table 3. Annual browse of terminal bud (%) and cumulative browse (1989-1992) by treatment and years since planting.

	Years since planting			Cumulative	Sample
Treatment	1st	2nd	3rd	browse (%)	size
Control	47.5	31.0	25.5	65.0	200
Mesh sleeves	5.0	12.8	18.8	28.5	400
Reemay sleeves	6.5	24.8	25.8	43.0	400
Tubes	0.0	5.8	2.5	7.3	600
Combined	8.8	15.4	15.3	28.8	1600

Table 4. Failure of browsing protection devices (%) by treatment and years since planting. See text for description of types of failure. Percentages may not add because of rounding error.

	Years since planting			Cumulative	Sample
Treatment	1st	2nd	3rd	failure (%)	size
Reemay	0.0	44.5	26.0	70.5	400
Mesh	0.3	27.8	27.5	55.5	400
Tubes	0.0	8.0	17.3	25.3	600

Table 5. Percent of seedlings with distorted top growth by the end of the 3rd growing season by species and protection device.

Treatment	Northern red oak	Black walnut	Eastern white pine	Norway spruce	Eastern hemlock	Combined species
Control	5.6	0.0	1.7	0.0	0.0	1.5
Mesh	22.2	14.6	10.0	47.0	15.0	22.8
Reemay	43.1	22.9	24.2	17.0	28.3	26.3
Tubes	22.2	8.3	4.4	4.7	2.2	7.8

DISCUSSION

Study sites for this research were selected because natural regeneration was inadequate and browsing damage was heavy in recent plantations. Regeneration was dominated by browse resistant species such as *Carpinus caroliniana*, *Ostrya virginiana*, *Acer pensylvanicum*, *Berberis* spp., and ferns. Earlier studies have established that natural regeneration may be inadequate when deer densities exceed 7-8 deer/km² (Behrend 1970, Anderson 1984, Tilghmann 1989). The lowest deer density observed in this study, 13 deer/km², was clearly too high to obtain sufficient natural regeneration. A browse line has developed in the areas with more than 20 deer/km². It should be noted that pellet-group counting has been criticized as unreliable (Fuller 1991), although studies have reported good correlations with track counts (Mooty and Karns 1984) and drive-line census (deCalesta and Witmer 1990). Therefore, deer densities reported here should only be considered as approximations.

Compared with unprotected seedlings and seedlings protected with sleeves, hardwood seedlings protected by tree shelters have grown taller, lower mortality rates, lower percentage of distorted terminal leaders, and lower browse pressure. This research concurs with earlier studies that tree shelters increase growth of individual hardwood seedlings (Lantagne and others 1990, Lantagne 1991, Ponder 1991, Minter and others 1992, Kittredge and others 1992, Walters 1993, Smith 1993). Tree shelters have the added benefit of reducing browse damage (Table 3).

Tree shelters are not a plant and walk away alternative (Smith 1993). Tree shelters failed not only because of rotting stakes, which we expected, but also because of buck rubbing and wind rocking in wet soils. It is difficult to properly set stakes in rocky and shallow soils. There were 3 causes of distorted terminals for trees within tubes. Wasps nests in tube interiors formed a physical barrier for terminal expansion, especially second flush growth. Emerging terminal leaders often snagged on the mesh placed over tubes to bird entry. Snagged terminals spiraled until penetrated the mesh. Lastly some trees gradually spiraled against tube interiors until emerging.

It is still too early to determine if tree shelters are suitable for conifer species. Although height of eastern white pine and Norway spruce protected by tree shelters was higher than for other treatments, the increased height was only equivalent to one year's growth (Table 1).

Both types of sleeves were unsuitable for newly planted seedlings due to high failure and terminal distortion rates. Consequently, seedlings protected by sleeves were not taller than unprotected seedlings after 3 growing seasons . High failure rates resulted in unacceptable levels of terminal buds browsing, especially of hardwoods. The most common cause of sleeve failure was disintegration of sleeve to degree that the terminal leader was not covered. Other causes were removal by deer pulling sleeves off seedlings and animal gnawing of sleeves. Reemay sleeves gradually fray and the sleeve interior can become a dense fiber web which snags and distorts expanding terminal growth. Most distortions associated with mesh sleeves occurred when sleeve alignment deviated from vertical and expanding terminals became snagged. Sleeves were pulled from vertical alignment by animals, competing vegetation, wind, and rain, ice, and snow loads.

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EFFECTS OF TREE SHELTERS ON PLANTED RED OAKS AFTER SIX GROWING SEASONS

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Abstract: A 22 year-old shelterwood treatment that failed to regenerate oaks on an upland site in Michigan was clearcut in the fall of 1986 and planted with 2-0 northern red oak (*Quercus rubra* L.) seedlings the following spring. The four planting treatments included: control (clearcut harvest only), brush control only, 48 inch tree shelters only, and brush control plus tree shelters. After three years, 65 percent of sheltered northern red oak seedlings were at least 47 inches tall and on average 21 inches taller than unsheltered seedlings. After six years, a total of 92 percent of sheltered trees had reached a minimum height of 47 inches. By 1992, the height growth of sheltered trees made DBH measurements possible on 80 percent of sheltered trees (height $\geq$ 4.6 ft) compared to only 52 percent of unsheltered trees. Sheltered trees had significantly greater average diameters. Tree shelters have improved survival, diameter and total height growth over the first six growing seasons. The data showed however, that average tree height growth was comparable between sheltered and unsheltered trees for the past three growing seasons. The 19 inch total height advantage appears to be the result of accelerated growth during the first three growing seasons when trees were within the 48 inch tall shelters.

INTRODUCTION

The dramatic height growth results reported for sessile oak (*Quercus petraea* (Matt.)) in tree shelters in Great Britain (Tuley 1983), combined with the difficulty of regenerating northern red oak (*Quercus rubra* L.) in the United States (Lorimer 1989) contributed to the initial testing of tree shelters in the United States. The promise of improved survival and height growth for a wide variety of other species however, contributed to their widespread application by many public agencies, private industries and individuals before long-term research results were available. Early research results have indicated that tree shelters have some potential benefits under the right circumstances (Applegate and Bragg 1989, Burger and others 1992, Smith 1993).

Published research accounts on the impact of tree shelters on tree growth are limited for North American plant species. Potter (1991) wrote an excellent handbook summarizing the results of research for tree shelter use in Great Britain. Windell (1991) published an excellent compilation of information on tree shelters for use by researchers in the United States. He summarized current research and information on general use, economic feasibility, and materials and construction techniques. Early indications are that tree shelters protect planted seedlings from animal damage and increase early seedling survival and height growth (Lantagne and others 1990), however little information is available on the longevity of the benefits accrued by tree shelter use. The objective of this study is to evaluate the long-term effectiveness of tree shelters in establishing northern red oak. This paper presents 6-year results of a tree shelter study established in a Michigan clearcut.

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METHODS

The study is located in an oak-hickory forest at the W.K. Kellogg Experimental Forest in southern Michigan. Soils are predominantly sandy loam (Alfisols) that developed from stony glacial drift on a rolling, well drained site. An average northern red oak site index of 66 (base age 50) was measured for the site before harvest. The study area underwent a shelterwood seed cut in 1954 and a removal cut in 1964 as part of a previous oak regeneration study (Rudolph and Lemmien 1976). Before the study was established, all stems greater than 2 inches diameter at breast height (DBH) were removed in a whole-tree clearcut harvest in the fall of 1986. All remaining stems greater than 1 inch DBH were removed by hand.

A 2 X 2 factorial combination of treatments was replicated on 4 one acre blocks in a split-plot design. A 22 foot buffer zone was maintained between treatment blocks. Each acre block was split to accommodate brush control treatments (main plots). Each main plot was then split to accommodate tree shelter treatments (split plots). The four treatments were: control (no brush control, no shelters); brush control only; 48 inch shelters only, and brush control and shelters. All treatments were randomly applied within replications.

Brush control occurred as a prescribed burn and basal application of triclopyr and oil to non-oak species before harvest. Basal applications of triclopyr and oil with a backpack sprayer were also applied to non-oak species in the three years following harvest at an average cost of $40 to $45 per acre per year. The highest estimated average cost to treat each tree over a three year period was $0.38 per tree at the assigned planting density of 360 trees per acre.

Plastic shelters were constructed from 20 X 48 X 0.25 inch sheets of white corrugated polyethylene plastic. Each plastic sheet cost about $1.50 and stakes about $0.50 each. Individual sheets were formed into roughly 5- X 5-inch square shelters and stapled to 1- X 1-inch wooden stakes driven into the ground next to planted oak seedlings. The 48 inch tall shelters were placed close to the soil surface but were not sealed at the soil surface. Stakes were also driven into the ground next to all unsheltered seedlings. The stakes and shelters were checked and replaced as necessary each spring and fall in each year of the study. Maintenance costs in terms of labor used, averaged over 20 hours per acre per year for these shelters.

Northern red oak acorns collected from the study site in the fall of 1984 were float tested for viability. "Sinkers" were collected and 8 per square foot were sown in a prepared nursery bed. During the first lag phase of development during the second nursery season, seedlings were undercut in the nursery bed at 6 to 8 inches to encourage development of a fibrous root system (Hanson et al. 1986, Johnson et al. 1986). The 2-0 northern red oak seedlings were lifted in mid-April 1987, sorted to a minimum 3/8-inch root collar diameter (Johnson 1986), root pruned to 8 inches, wrapped and stored at 1° C until planting. Seedlings were planted over a two day period in late April 1987 with dibble bars at a 11- X 11-foot spacing for a total of 90 oak seedlings per treatment or 360 seedlings per block. After planting, seedlings were clipped 7 inches above the ground-line (Johnson et al. 1986). The 48 inch tree shelters were then installed. All work was completed by May 1, 1987.

Heights of all planted northern red oak seedlings were measured in the fall of 1992. The incidence of browse was not formally evaluated as surveys in previous years had found < 10 percent of unprotected trees affected. DBH measurements were collected for the first time on all trees ≥ 4.6 feet in height after the 1992 growing season. Analysis of variance was used to test for significant differences among treatment means. The standard assumptions of analysis of variance were verified before data analysis. Percentages were transformed with the arcsine procedure (Little and Hills 1978). Chi-square analysis was used to test for differences in height class distributions and diameter class distributions for sheltered and unsheltered seedlings.

RESULTS

Over the first six years of the study, control of non-oak woody species has had no significant effect on the total height of planted red oak ($P = 0.24$), but survival and total height of sheltered and unsheltered trees have differed significantly (Table 1). There were no significant brush control X shelter interactions for any measured variable. Seedling survival and height for sheltered seedlings were significantly greater than for unsheltered seedlings (Table 2). Although sheltered trees have maintained a 19 inch total height advantage, the height growth of sheltered and unsheltered trees over the past three growing seasons has not differed (Table 2). Early growth gains resulted in 77 percent of the trees originally planted in shelters to average over 47 inches high after six growing seasons compared to only 50 percent of the unsheltered trees originally planted in the study ($P < 0.001$). Sixty-seven percent of the total number of sheltered trees originally planted were tall enough (≥ 4.6 ft) in 1992 to collect DBH measurements compared to a total of 38 percent of unsheltered trees ($P < 0.005$). Sheltered trees were also found to have significantly larger diameters than the measured unsheltered trees ($P < 0.0029$) (Table 2).

Table 1. Analysis of variance for total tree height for planted trees.

Factors	DF	Mean Square	F Ratio
Main Plots			
Replication (Rep)	3	877.05	
Brush Control (BC)	1	527.49	2.03
Rep X BC (Error a)	3	260.02	
Split Plots			
Shelters (S)	1	10257.48	34.03**
BC X S	1	533.80	1.77
Error	6	301.39	

** -- Significant ($p < 0.0011$)

Figure 1 illustrates the effect of shelters on seedling height class distribution after six years. Sheltered seedlings dominate every category above five feet. A Chi-square analysis indicated that the distributions were significantly different ($P < 0.0001$). A similar analysis of diameter distributions between sheltered and unsheltered trees also indicated a significant shift towards larger diameters for sheltered seedlings ($P < 0.0001$).

DISCUSSION

Tree shelters can improve the early survival and height growth of oak seedlings (Minter et al. 1992, Lantagne 1991, Tuley 1983). Survival differences between brush control and tree shelter treatments were not a major factor in this experiment. Over the first three growing seasons, survival averaged above 90 percent for all treatments. Results similar to these have been reported by Minter et al. (1992). After six growing seasons, survival was 10 percent greater in treatments receiving brush control or tree shelters as compared to the controls (Table 2). It appears that either brush control or tree shelter treatments are sufficient to maintain survival above 80 percent for the first six growing seasons after establishment. Therefore, tree shelters are not necessarily required to maintain acceptable survival rates for planted northern red oak on this site.

Table 2. Effect of shelters and brush control on survival and height of planted northern red oak seedlings after the sixth growing season.[1]

Treatments	Survival	Total Height	Height Increase[2]	DBH[3]
	-- (%) --	-- (in) --	-- (in) --	-- (in) --
Main Plots				
Brush Control	83a	75a	11a	0.5a
No Brush Control	73a	80a	12a	0.5a
Split Plots				
Shelter	84a[4]	87a[5]	11a	0.6a[6]
No Shelter	73b	68b	12a	0.5b

[1] Means within columns for main and split plots followed by the same letter are not significantly different at P<0.05 as found in the Analysis of Variance.
[2] Average yearly height increase was determined by subtracting 1989 total height from 1992 total height for each individually measured seedlings and dividing by three.
[3] DBH measurements for all trees ≥ 4.6 feet in height after six growing seasons.
[4] Survival percentages were significantly different at P<0.0042.
[5] Total height was significantly different at P<0.0011.
[6] DBH was significantly different at P<0.0029.

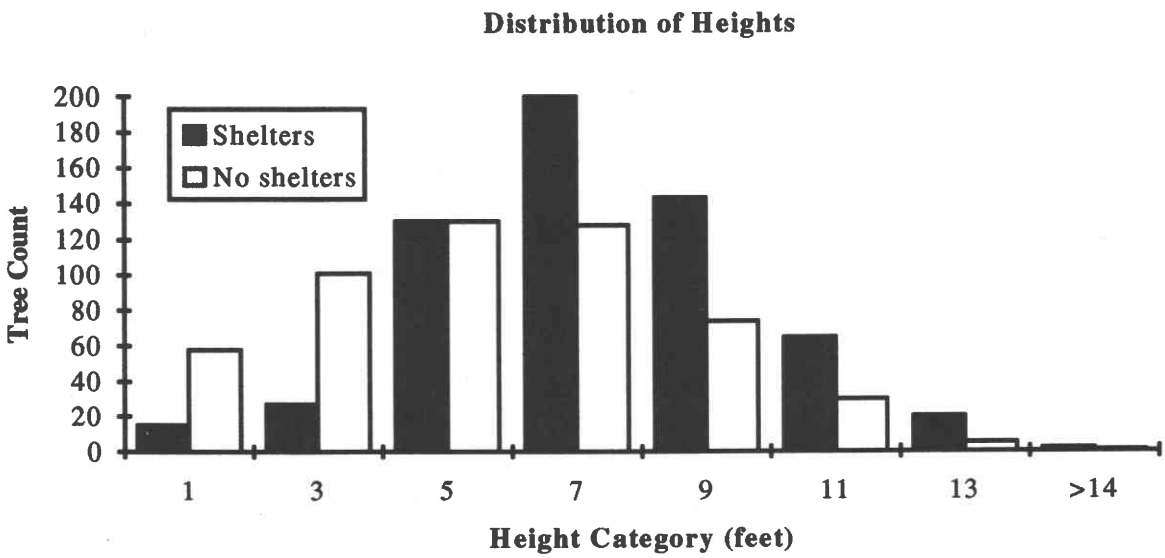

Figure 1: Height distributions of sheltered and unsheltered northern red oak seedlings after the sixth growing season. (The mid-range of each category is indicated on the x-axis. Distributions of the two classes of seedlings differ significantly based on Chi-square ($X^2 = 135$, $p < 0.0001$).

Although the analysis of the third year data affirmed the potential benefit of tree shelters on early growth and total tree height, sixth year results show no growth differences among treatments (Figure 2). Sheltered northern red oak seedlings have maintained an average height advantage of 19 inches, but growth rates between the treatments have stabilized. In the first three growing seasons, sheltered trees averaged over 11 in/year height growth, compared to less than 8 in/year height growth for unsheltered trees. In the last three years of the study, unsheltered trees grew over 11 in/year compared to less than 11 in/year for sheltered trees. In an earlier report on this study it was shown that increased linear stem elongation of the initial terminal flush was more important to total height growth than an increased number of flushes (Lantagne 1991). As would be expected, the latest results indicate that once these trees grew out of the shelter, overall terminal shoot growth slowed. This reduction in growth has been documented for northern red oak seedling sprouts that were clipped at ground level and placed in tree shelters (Kittredge et al. 1992). Tuley (1983) reported sessile oak height growth also slowed after year three in a tree shelter experiment in Great Britain. Potter (1991) did not report third year height growth decline for other tree shelter studies conducted in Great Britain. However, the maximum number of growing seasons for tree shelter research reported in the literature is five (Potter 1991). In this study, it appears that the apparent growth stimulus of tree shelters was lost over the last three growing seasons, similar to that found by Tuley (1983).

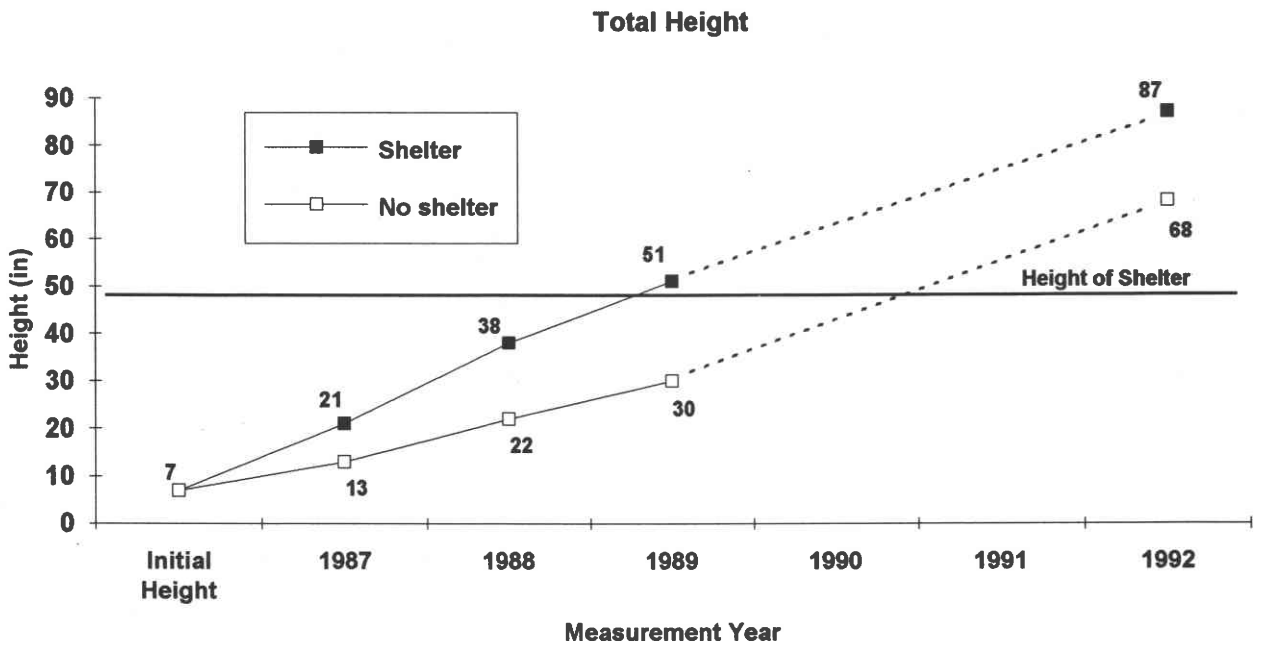

Figure 2: Mean Heights of surviving planted northern red oak seedlings. (All seedlings were top pruned at 7 inches above the groundline immediately following planting. Sheltered seedlings were significantly taller at the end of each growing season.)

The lack of a sheltered environment for the first flush may be the reason for reductions in average yearly height growth during the last three growing seasons. Tree shelters appear to increase the number of leaves available for photosynthesis (McNeel et al. 1993, Kittredge et al. 1992), and the length of time these leaves may remain active in producing photosynthate (Minter et al. 1992).

The interior of shelters have also been found to have elevated air temperatures, higher levels of CO^2 and a higher relative humidity which may contribute to the improved growth of sheltered seedlings (Minter et al. 1992, Potter 1991). In addition, trees protected from movement, as are seedlings in tree shelters, tend to grow taller and have smaller diameters (Rosenberg et al. 1983).

Although terminal shoot growth has slowed, the initial DBH measurements for sheltered trees were significantly greater than those measured for unsheltered trees. The early accelerated terminal shoot growth of sheltered seedlings did result in larger diameters after the trees exited the shelters. The reasons for this relationship are unclear but may be a result of being at DBH for a longer period combined with the stimulation of diameter growth in response to wind movement (Rosenberg et al. 1983).

SUMMARY

Many sheltered and unsheltered trees continue to maintain codominant crown positions with competing vegetation on the study site. The strong growth and development of unsheltered trees after six growing seasons may show the importance of using large, undercut and top clipped planting stock on the success of oak regeneration (Johnson et al. 1986). The cost of using and maintaining tree shelters, and the problem of plastic litter remaining in the forest, have to be evaluated considering the level of survival and growth response of unprotected trees in this study. The minimum cost for the tree shelter materials was over $2.00 per tree in this study. There were also additional costs associated with installation and maintenance that pushed the overall cost per tree higher. The cost of labor and material for controlling brush with herbicides for three years averaged less than $0.40 per tree. Even adding two additional years of brush control would only add $0.26 per tree for a total cost of less than $0.70 per tree over five years. Using high quality planting stock with brush control to increase the amount of northern red oak regeneration is less expensive then the use of tree shelters and may be sufficient in areas free of heavy browsing by rabbits and deer.

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RED SPRUCE/HARDWOOD ECOTONES IN THE CENTRAL APPALACHIANS

Harold S. Adams[1], Steven L. Stephenson[2], David M. Lawrence[3], Mary Beth Adams[4], and John D. Eisenback[5]

Abstract: We are currently investigating patterns of species composition and distribution, ecologically important population processes, and microenvironmental gradients along ten permanent transects (each consisting of a series of contiguous 10 x 10 m quadrats) established across the typically abrupt and narrow spruce/hardwood ecotone at seven localities in the mountains of central West Virginia and western Virginia. Primary emphasis of our research is directed toward testing three basic hypotheses: (1) red spruce communities in the central Appalachians are decreasing in areal extent due to encroachment of surrounding hardwood communities, (2) stress-induced growth decline in red spruce is a factor in this decrease, and (3) the direction and rate of successional change can be predicted from models developed from quantitative data obtained from field studies of red spruce/hardwood ecotones. Preliminary data obtained during the 1992, 1993, and 1994 field seasons suggest that central Appalachian red spruce communities presently exist at least in static equilibrium with respect to surrounding hardwoods and exhibit, at some localities, advance regeneration into the hardwood communities. (Supported in part by funds provided by the USDA Forest Service).

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NITROGEN DYNAMICS IN OAK FOREST SOILS ALONG A HISTORICAL DEPOSITION GRADIENT

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Abstract: This study quantified soil nutrient status and N mineralization/nitrification potentials in soils of oak-dominated, unmanaged forest stands in seven experimental forests ranging along a historical and current acidic deposition gradient from southern Illinois to central West Virginia, U.S.A. Among these seven sites (that spanned 8.5° of longitude) soil pH and Ca^{2+} decreased and soil organic C and extractable Al^{3+} increased from west to east. In general, initial soil solution NO_3^-, total N mineralization potential and net NO_3^- accumulation over 30 days of incubation (as measured by aerobic laboratory incubations) also decreased from west to east, whereas initial soil solution NH_4^+ was uncorrelated with longitude. The Fernow Experimental Forest (W.Va.), the eastern-most site, was the exception to this trend. Soils from the Fernow had the highest concentrations of both NO_3^- and NH_4^+ in the soil solution, and the greatest N mineralization potential. Stepwise regressions of N mineralization rate, net NO_3^- accumulation, and proportional nitrification on initial soil properties produced models with overall r^2 of 0.705, 0.772, and 0.708, respectively. Rates of N turnover were positively correlated with initial NO_3^-, pH, and Ca:Al ratio and negatively correlated with extractable Al^{3+} concentrations. Differences in oak growth and mortality may be related to the differences in soil chemical status and soil N dynamics among these seven experimental forests.

INTRODUCTION

Most mature, upland forests of eastern North America are nitrogen (N) limited (Aber et al. 1989). In such ecosystems, the processes that regulate the availability of inorganic N to roots and soil microbes are those involved in the mineralization of low molecular weight organic compounds containing N (e.g. amino acids, polypeptides, nucleotides) to NH_4^+ (i.e. ammonification or N mineralization) and, in some cases, subsequently to NO_3^- (i.e. nitrification). Once the N is in inorganic form, it is subject to competition between roots and microbes, both of which result in the inorganic N being reconverted to biomass N.

Rates of nitrogen mineralization and nitrification are affected by a variety of environmental factors, including soil solution pH, temperature, and moisture (Plymale et al. 1987; Robertson 1982), as well as the rate at which nitrogenous substrates are supplied. Thus, modification of soils by anthropogenically-generated deposition has the potential to alter the rates and patterns of organic N turnover in forest soils, and thereby, to alter the basic functioning of the forests those soils support (Bondietti and McLaughlin 1992).

Over the last 50 years, some forest soils in eastern North America have experienced decreases in pH, increases in NO_3^- deposition and soil solution concentration, increased soil solution Fe^{3+} and Al^{3+} concentrations, and lowered Ca:Al ratio (Bondietti and McLaughlin 1992), all of which have the potential to affect key ecosystem processes (Aber et al. 1989). In N-poor ecosystems, the activity of autotrophic nitrifiers may be inhibited and soil NO_3^- concentrations remain low because of a number of factors, including the inability of such bacteria to compete well against plant roots and heterotrophic soil microbes for NH_4^+ and the direct inhibition by low pH (van Miegrot et al. 1992) and relatively high Fe^{3+} and Al^{3+} availability (Liang and Tabatabai 1978). Although the activity of heterotrophic N-mineralizing bacteria and fungi may not be as sensitive to changes in soil chemistry as that of autotrophic nitrifiers (Legge and Crowther 1987), N-mineralization rates are generally lower on acidified soils (Plymale et al. 1987).

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N-limited forest ecosystems (i.e. ones with low rates of mineralization and little nitrification) are typically dominated by tree species which are dependent on ectomycorrhizae (Vogt et al. 1991) such as oaks (*Quercus* spp.), pines (*Pinus* spp.), and spruces (*Picea* spp.). Because of the ability of their ectomycorrhizae (ECM) to forage for NH_4^+, reduce NO_3^-, and degrade low molecular weight organic N substrates, these tree species can maintain growth even under very low N conditions (France and Reid 1983, Jansen 1991). Heavy, chronic N deposition may greatly alter the N status of forest soils, through its effect on leaching of Ca^{2+}, mobilization of Al^{3+}, and removal of N limitation for tree growth. Under such conditions, ECM-dependent species may decline and be replaced by tree species dependent on vesicular-arbuscular mycorrhizae (VAM) more typical of N-rich forest sites (Jansen 1991; Arnolds 1991) (e.g. maples, *Acer* spp. and yellow-poplar, *Liriodendron tulipifera*). Thus acidic deposition with a significant N component has the potential to affect eastern forest ecosystems over the long term, both in soil chemical status and in tree species composition.

As part of a larger study of the relationship between forest management, tree drought responses/mortality, historical deposition rates, and ecosystem function along the Ohio River Valley of eastern North America, this study was specifically designed to: (1) measure potentials for conversion of organic nitrogen to inorganic forms by soil microbial assemblages in forest soils differing in their historical deposition rates and in site quality; (2) determine relationships between differences in nitrogen mineralization/nitrification potentials and other soil properties (including both static properties such as texture and deposition-related properties such as Ca:Al ratio); and, (3) relate forest soil ecosystem processes such as the ones measured here to variations in tree crown condition, mortality, and drought response among these forests sites. This paper discusses the results of our analyses of the first two of those three objectives.

METHODS

Study Sites and Field Sampling

Seven experimental forests located in the Ohio River Valley were selected for study (Figure 1). These seven sites spanned 8.5° of longitude (Table 1) along a gradient of historical atmospheric deposition which increased from west to east (Work Group One 1983, Lovett 1992). In each experimental forest, two unmanaged, oak-dominated plots on NW-NE facing 15%-25% slopes on non-calcareous substrates were selected. In each plot we established two random quadrats, and took seven random soil samples from each quadrat along a transect running parallel to the contour of the slope during a ten day period in July 1993. Before taking a sample, the forest floor (litter + unconsolidated humus) was cleared from the mineral soil surface over an area of approximately 0.25 m^2. A 2 cm soil corer was then used to extract A-horizon cores of 5-20 cm in length, depending on the forest site. Each sample was a composite of 5-15 such A-horizon cores totalling at least 200 g fresh mass of soil. We limited our sampling to the A-horizon because rates of N-mineralization and nitrification are typically much greater in the A-horizon than in the B-horizon and unconsolidated litter (Boerner and Koslowsky 1989). All samples were transported to the laboratory under refrigeration.

Table 1. Experimental Forest study site names, abbreviations, and locations.

Name (Abbrev.)	State	Latitude	Longitude	Elevation Range
Kaskaskia (KAS)	Illinois	37°32' N	88°20' W	460-500'
Bald Rock (BDR)	Kentucky	36°59' N	84°16' W	1060-1080'
McKee (MCK)	Kentucky	37°27' N	83°59' W	1180-1270'
Robinson (ROB)	Kentucky	37°15' N	83°20' W	1050-1150'
Mead (MEX)	Ohio	39°15' N	83°00' W	760-800'
Raccoon (RAC)	Ohio	39°10' N	82°23' W	790-860'
Fernow (FER)	West Virginia	39°03' N	79°41' W	2650-2730'

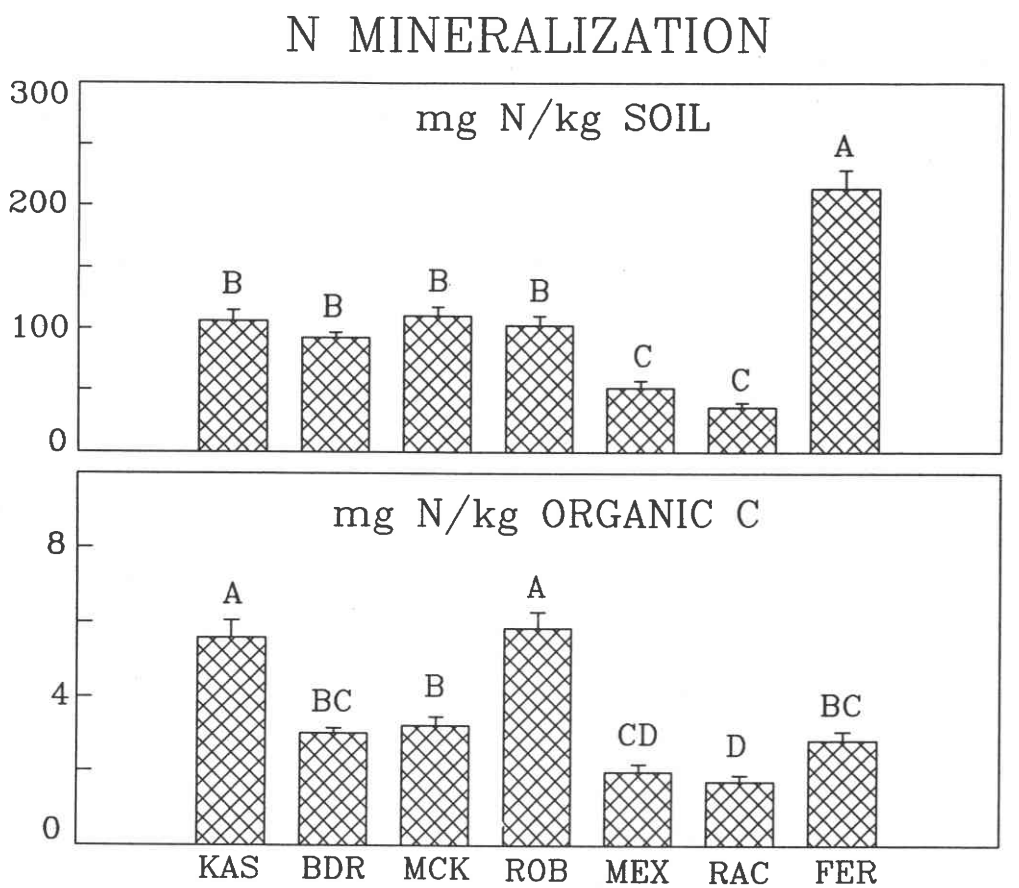

Figure 1. A-horizon N mineralization on a mass basis (mg N/kg dry soil) and on an organic matter basis (mg N/kg organic C), in soils from seven experimental forests along the Ohio River corridor. Forests are arrayed from west to east along the x-axis. Histogram bars represent the mean of N=14, and the error bars represent one standard error of the mean. Bars with the same lower case letter were not significantly different at $p<0.05$ following analysis of variance and Ryan-Einot-Gabriel-Welsch Modified F Test. Forest codes follow Table 1.

Laboratory Methods

Each sample was subdivided into two subsamples. One was sent to Micro-Macro International, Inc., Athens, GA for analysis of soil texture by hydrometer, pH in water, Ca^{2+}, Mg^{2+}, K^{+}, and Na^{+} in 1 M NH_4OAc extracts, and NO_3^{-}, PO_4^{3-}, and Al^{3+} in 1 M KCl extracts. Methods followed Jones (1992). The second subsample was air-dried for 10-14 days, then analyzed for organic C content by Walkley-Black wet oxidation (Allison 1965).

The third subsample was then subdivided into two portions. One portion of approximately 10 g was extracted with 1 M KCl and the extract frozen pending analysis. The second portion was air-dried for 10-14 days, then passed through a 6 mm sieve to remove stones, roots, and leaf fragments. Approximately 100 g of the sieved soil was weighed, placed in a 300 ml styrofoam cup, and brought to 70% of field capacity with "Hubbard Brook Recipe" artificial rainwater (Lee and Weber 1979). We used artifical rainwater rather than deionized water to better simulate field conditions and to prevent the excessive leaching of the soil samples that might occur if deionized water were used throughout the incubation period.

All samples were incubated under aerobic laboratory conditions for 30 days at 23-28° C. Samples were weighed every 2-3 days, and were maintained at 50-70% of field capacity by adding artifical rainwater as needed. At the end of the incubation period, approximately 10 g (dry mass) of soil was taken from each incubated sample and extracted as above.

NH_4^+ and NO_3^- concentrations of each initial and final sample extract were determined colorimetrically on a LaChat QuikChem Autoanalyzer. Relative nitrogen mineralization was calculated as the difference between total inorganic N (i.e. $NH_4^+ + NO_3^-$) in the initial vs final extracts, less the N added in the rainwater solution. Net NO_3^- accumulation was calculated as the difference between initial and final NO_3^- concentrations in each sample. To correct for differences in soil organic matter content among forest sites, we expressed both N mineralization and net NO_3^- accumulation as mg N/g organic C. Percent nitrification was calculated as the percent of the total nitrogen mineralized which was converted to NO_3^- during the incubation period (Campbell et al. 1993).

Initial soil parameters and rates of N turnover were compared among experimental forests, stands within forests, and sample plots within stands by analysis of variance, using the Ryan-Einot-Gabriel-Welsch Modified F Test to post-test differences among means (SAS 1985). Regression models of N turnover rates as a function of initial soil conditions were constructed by maximum-r^2, forward selection stepwise regression (SAS 1985). Significant differences were at $p<0.05$ except where otherwise indicated.

RESULTS

Soil Physical and Chemical Properties

Five of the seven soil chemical parameters we measured varied linearly with longitude (Table 2). Organic carbon content, NH_4^+ and Al^{3+} all increased from west to east, whereas pH and the molar Ca:Al ratio decreased from west to east (Table 2). In contrast, Ca^{2+} was significantly greater in soils from the Kaskaskia E.F. (IL) and Robinson E.F. (KY) than in soils from the other five sites, and NO_3^- concentration was greatest in the two sites at the ends of our longitude gradient, Kaskaskia (IL) and Fernow (WV).

There was no consistent variation in soil texture with longitude (Table 3). The soils of the Mead (OH) E.F. had the greatest proportion of clay of the seven sites sampled, and there was little variation in clay content among the soils of the other six sites. The percentage of sand varied more than did the clay fraction, with the soils of the Fernow E.F. (WV), Bald Rock E.F. (KY), and McKee E.F. (KY) having greater relative sand content than the soils of the Mead E.F. (OH), Raccoon E.F. (OH) or Kaskaskia E.F. (IL) E.F.s. The dominant textural classes ranged from silty clay through silty clay loam and clay loam.

Nitrogen Mineralization and Nitrification

Without considering the Fernow E.F., total N mineralization decreased linearly from west to east, with the soils from the Raccoon E.F. (OH) and the Mead E.F. (OH) mineralizing significantly less ($p<0.0001$) total organic N per unit soil mass than the soils from the four sites to the west (Figure 1). On a mass basis, soils from the Fernow E.F. mineralized >2X as much N over 30 days as did the soils from any of the six other sites, and did not fit the linear longitudinal pattern of the other six sites. When N mineralization was adjusted for the differences in initial soil organic C among sites, the rate of N mineralization at the Fernow E.F. was no longer greater than those from the sites to the west. The lowest rates of total N mineralization/g organic C were still recorded at the Raccoon E.F. (OH) and the Mead E.F. (OH).

Net NO_3^- accumulation on a mass basis was significantly greater at the Kaskaskia E.F. and the Fernow E.F. than at the other five sites (Figure 2). The soils from the other five experimental forests accumulated an order of magnitude less NO_3^-, and did not differ significantly from each other in NO_3^- accumulation rate. When NO_3^- accumulation rate was adjusted for difference in organic C content, the rate observed in the soils from the Fernow E.F. became more similar

Table 2. Chemical properties of A-horizon soils from unmanaged, oak-dominated forest plots in seven experimental forests. Within a column, means followed by the same lower case letter were not significantly different at $p<0.05$. N=28 for each experimental forest and standard errors of the means are given in parentheses. All parameters except pH and organic carbon (%) are in mg/kg dry soil. Experimental forests are listed from west to east.

Experimental Forest	pH	Org. Carbon	NO_3^-	NH_4^+	Ca^{2+}	Al^{3+}	Ca:Al Molar Ratio
Kaskaskia (IL)	5.14a	1.91cd	17.33b	22.42c	683.5a	10.5e	136.20a
	(0.18)	(0.30)	(2.28)	(1.78)	(48.1)	(3.3)	(22.65)
Bald Rock (KY)	4.50b	3.08bc	0.59c	26.58bc	70.6b	147.4c	0.32c
	(0.17)	(0.30)	(0.16)	(1.70)	(8.8)	(9.7)	(0.05)
McKee (KY)	4.28b	3.42b	2.34c	25.41bc	64.3b	190.9bc	0.23c
	(0.06)	(0.52)	(0.32)	(1.90)	(4.4)	(15.2)	(0.03)
Robinson (KY)	4.88a	1.76d	1.96c	24.12c	345.4a	83.8d	27.20b
	(0.11)	(0.05)	(0.48)	(1.53)	(42.0)	(22.4)	(12.00)
Mead (OH)	4.25bc	2.64bcd	0.51c	33.91b	113.3b	311.3a	0.27c
	(0.03)	(0.27)	(0.13)	(2.58)	(14.3)	(17.9)	(0.06)
Raccoon (OH)	4.07c	2.10bcd	0.39c	24.01c	59.7b	212.5b	0.19c
	(0.07)	(0.32)	(0.06)	(1.85)	(7.8)	(11.3)	(0.03)
Fernow (WV)	3.79d	5.59a	50.57a	50.58a	146.3b	244.3a	0.48c
	(0.06)	(0.31)	(6.94)	(3.97)	(19.2)	(13.5)	(0.14)

to, though still significantly different from, those at the five intermediate sites (Figure 2). The percent of total N mineralization which was subsequently nitrified to NO_3^- followed the same pattern (Figure 2).

If one does not consider the Fernow E.F., two general patterns emerged: (1) on a mass basis, total N mineralization decreased 66% over 8.5° of longitude from west to east, and (2) nitrification, measured either as net NO_3^- accumulation or percent of newly mineralized N nitrified, was significantly greater in the site with Ca:Al ratio >135 than in the other five sites. When N turnover rates were adjusted for initial organic C content of the soils, the east-west trend of decreasing total N mineralization existed among all seven sites, including the Fernow E.F.; however, even when differences in organic C were accounted for, the soils from the Fernow E.F. (WV) still accumulated significantly more NO_3^- and nitrified a significantly greater percentage of NH_4^+ than would be predicted by longitude.

Regression Analysis of N Turnover Rates

Because of the inherent weakness in assuming that field plots such as those analyzed here are truly replicates, one must interpret the results of analyses of variance with considerable caution. Given the large variation in initial soil conditions among sites, we felt a multiple regression approach which explicitly addressed the differences among study sites would be a more effective inferential tool in analyzing situations such as this one.

Table 3. Texture of A-horizon soils from unmanaged, oak-dominated forest plots in seven experimental forests. Within a column, means followed by the same lower case letter were not significantly different at $p<0.05$. N=28 for each experimental forest and standard errors of the means are given in parentheses. Experimental forests are listed from west to east.

Experimental Forest	Sand (%)	Silt (%)	Clay (%)	Prevalent Texture
Kaskaskia (IL)	12.0d (1.1)	64.8a (2.2)	23.1b (1.8)	Silty Clay Loam
Bald Rock (KY)	48.8ab (2.5)	27.5c (2.0)	23.7b (0.8)	Clay Loam
McKee (KY)	43.8b (3.5)	31.9cd (2.6)	24.3b (1.2)	Clay Loam
Robinson (KY)	35.3c (3.6)	35.6c (2.5)	29.2ab (1.7)	Clay Loam
Mead (OH)	18.0d (1.9)	47.2b (1.4)	34.8a (1.0)	Silty Clay
Raccoon (OH)	18.9d (1.5)	54.5b (1.2)	26.6b (0.8)	Silty Clay Loam
Fernow (WV)	56.9a (1.7)	21.8e (2.0)	21.3b (1.3)	Clay Loam

Regression of the rate of N mineralization per unit soil mass on the soil chemical and physical properties analyzed produced a linear model significant at $p<0.0001$ with overall model $r^2=0.705$ (Table 4). This model attributed the variance in N mineralization rates among samples and sites to, in declining order, initial NO_3^- concentration in the soils, soil texture, and soil pH, with initial NO_3^- and pH having positive slope coefficients and silt+clay content a negative sign. The robustness of the model can be appreciated when one considers that the highest N mineralization rates were recorded at the Fernow E.F., the site with the lowest pH and silt+clay content of the soil. Without the Fernow E.F. in the model, the r^2 would improve considerably. Organic carbon content was not a significant variance component in this regression, and a similar regression done on mineralization rate per unit organic matter produced essentially identical results.

Regression of the net rate of NO_3^- accumulation on soil properties produced a linear model with equal signficance ($p<0.0001$) and even closer fit ($r^2=0.772$) (Table 4). In this model, the initial concentration of NO_3^- and the Ca:Al molar ratio contributed the largest partial r^2 to the model, both with positive sign for the slope coefficient. Thus, net NO_3^- accumulation increased with increasing initial NO_3^- and Ca:Al ratio. Soil pH (+), initial Al^{3+} concentration (-), and silt+clay content (+) contributed less, but still significantly to model fit. The percent of newly mineralized N which was subsequently nitrified during the 30 day incubation period was also directly proportional to initial NO_3^- concentration and the Ca:Al ratio (Table 4). Adding in initial soil Al^{3+} concentration (-) and silt+clay content produced a linear model with $r^2=0.708$.

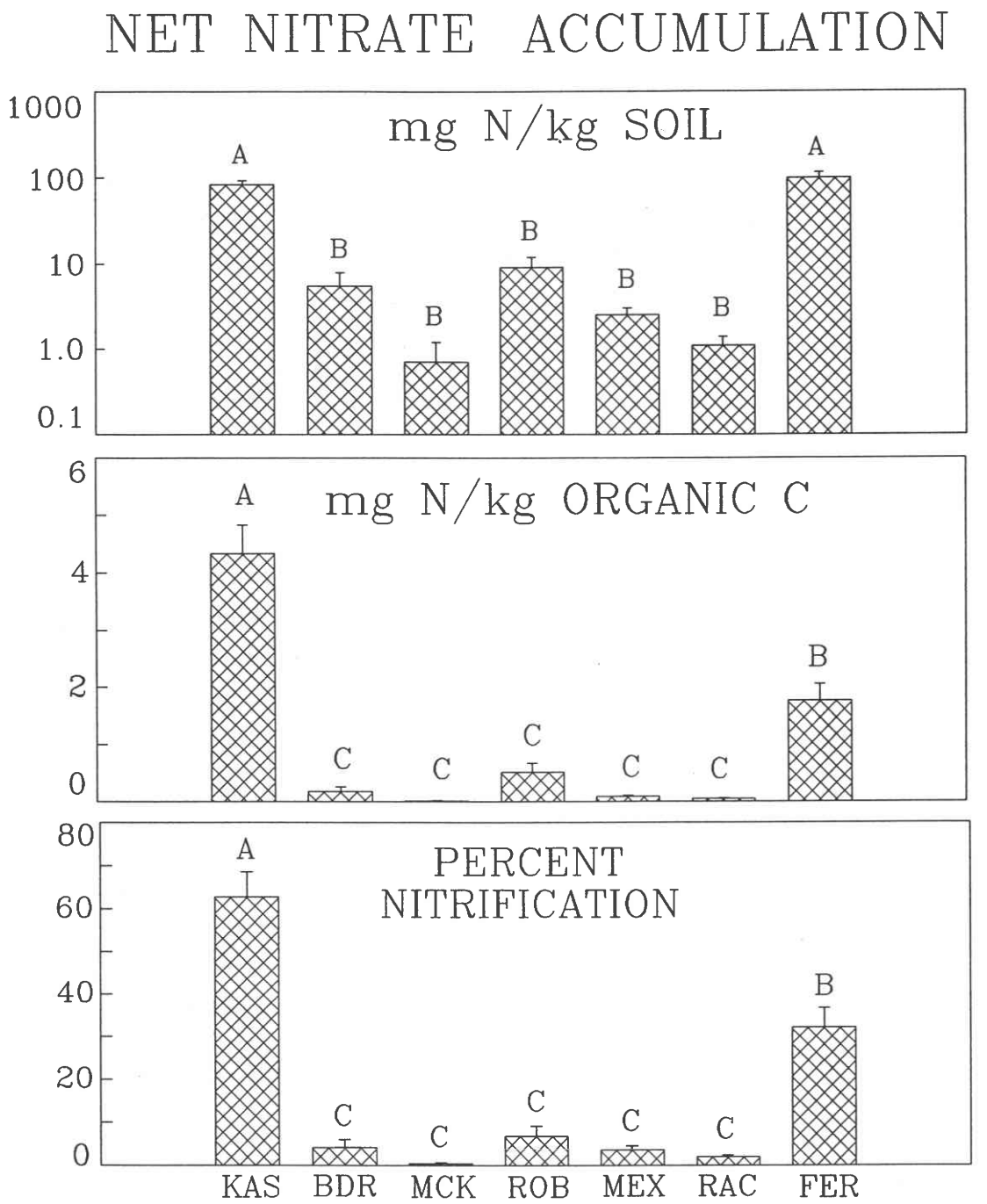

Figure 2. NO_3^- accumulation on a mass basis (mg NO_3^--N/kg dry soil) and on an organic matter basis (mg NO_3^--N/kg organic C), and percent nitrification of mineralized N in soils from seven experimental forests along the Ohio River corridor. Note the logarithmic y-axis for the two NO_3^- plots. Forests are arrayed from west to east along the x-axis. Format follows Figure 1.

Table 4. Forward selection, maximum r^2 regression of N turnover components on initial soil parameters. All independent variables with slope components significantly different from 0 at $p<0.05$ are given, and variables are listed in declining contribution to the total model r^2. N=167 for each regression.

Independent Variable	Slope Coefficient/ Significance of Slope
N Mineralization (mg N/kg soil), total model $r^2=0.705$, $p<0.0001$	
Initial NO_3^-	+2.10 p<0.0001
% Silt+Clay	-1.36 p<0.0001
pH	+12.76 p<0.026
Net NO3 Accumulation (mg N/kg soil), total model $r^2=0.772$, $p<0.0001$	
Initial NO_3^-	+1.94 p<0.0001
Ca:Al Ratio	+0.26 p<0.0001
pH	+7.83 p<0.048
Al	-0.07 p<0.014
% Silt+Clay	+0.27 p<0.048
Percent Nitrification, total model $r^2=0.708$, $p<0.0001$	
Initial NO_3^-	+0.66 p<0.0001
Ca:Al Ratio	+0.18 p<0.0001
Al	-0.05 p<0.0007
% Silt+Clay	+0.24 p<0.0019

Thus, all three measures of organic N turnover were directly proportional to and strongly affected by the initial NO_3^- concentration in the soil samples. Soil texture was also a significant variance component for all three measures of N turnover; however, N mineralization was inversely proportional to the silt+clay content of the soil whereas net NO_3^- accumulation and percent nitrification were directly proportional to that property. Total N mineralization was not affected by Al^{3+} or Ca:Al molar ratio; in contrast both net NO_3^- accumulation and percent nitrification were directly proportional to Ca:Al ratio and inversely proportional to Al^{3+} concentration.

DISCUSSION

As a result of changes in the patterns of fossil fuel use over the last two decades, the relative proportions of NO_3^- and SO_4^{2-} in precipitation and dry deposition have shifted away from predominatly S deposition to relatively greater deposition of N. Currently, upper elevations of the Appalachian Mountains receive as much as 27 kg N/ha/yr in precipitation (Lovett 1992), a level almost an order of magnitude greater than that in pristine precipitation (Aber et al. 1989; Lovett 1992). The geographic range of our seven study sites was designed to encompass a gradient of historical deposition of NO_3^-, SO_4^{2-}, and H^+ increasing from west to east, and a more recent NO_3^- deposition gradient increasing from 2.7-5.6 kg N/ha/yr in Illinois to 13.5-18.1 kg N/ha/yr in West Virginia (Work Group One 1983).

Chronic NO_3^- deposition can have a range of effects on forest soils, including enhanced leaching of Ca^{2+}, increased Al^{3+} solubility, and decreased Ca:Al ratio (Aber et al. 1989; Foster et al 1989). This pattern emerged clearly among our six study sites west of the Appalachian Plateau (i.e. excluding the Fernow Experimental Forest, West Virginia): Ca^{2+} decreased, Al^{3+} increased, and Ca:Al ratio decreased by two orders of magnitude between southern Illinois and southern Ohio. This spatial pattern was consistent with the temporal pattern of change in soil and leachate chemistry

reported from a sugar maple (*Acer saccharum*) forest in Ontario over 1981-1986 caused presumably by heavy N deposition in acidic precipitation (Foster et al. 1989).

Aber et al. (1989) hypothesized that chronic high NO_3^- deposition could reduce microbial activity and, thereby, reduce the rates of N mineralization and nitrification. Although this hypothesis was supported by the rates we observed in our six study sites west of the Appalachian Plateau, the longitudinal patterns of variation in N mineralization and nitrification differed. N mineralization decreased gradually and linearly with longitude, and the pattern of change correlated most strongly with pH. Over the 8.5° of longitude represented by our gradient, the rate of N mineralization decreased by 66%. In contrast, nitrification was significant only in the two study sites in which the Ca:Al ratio was >20; in all other sites nitrification was negligible. These data were consistent with *in vitro* studies demonstrating strong inhibition of nitrification by Al^{3+} and Fe^{3+}, even in the presence of high NH_4^+ concentrations (Liang and Tabatabai 1978), and supported the notion that nitrification is more susceptible to inhibition by low pH and high Al^{3+} than is N mineralization (Legge and Crowther 1987).

The soils of the Fernow Experimental Forest (WV) departed in a number of respects from the patterns which emerged among the other six sites. Extractable NO_3^- and NH_4^+ concentrations and organic matter content were greatest in this site. The greater organic matter accumulation may have been a function of the of higher elevation (therefore lower mean temperature) of this site. Similarly, the higher concentrations of inorganic N in the A-horizon may have been a function of greater mean annual precipitation, higher elevation (therefore lower primary productivity), and differences in the concentration of N in precipitation.

The rate of N mineralization per unit soil mass at the Fernow E.F. was considerably greater than those we observed in soils from the other six sites. However, this appears to have been primarily a function of differences in organic matter content. When N mineralization was expressed on a per unit organic matter basis, the activity of the soils of the Fernow became consistent with the longitudinal pattern suggested by the other six sites. This was not true, however, for either net NO_3^- accumulation or proportional nitrification: both remained higher than would be predicted on the basis of longitude even when expressed on an organic matter basis. This greater production and accumulation of NO_3^- indicates that the degree to which mineralization of organic N to NH_4^+ exceeded uptake by plants, fungi, and bacteria was greater at the Fernow than at all of the other study sites except Kaskaskia. Again, this is likely due to the higher elevation and cooler climate of Fernow producing lower uptake demands. Thus, we feel the departure of the soils of the Fernow from some of the longitudinal patterns we observed among the other six sites was the result of direct and indirect effects of differences in elevation.

Both mineralization and nitrification were strongly correlated with initial soil NO_3^- concentrations. This may have been the result of a simple feedback loop: soils which support high rates of mineralization are likely to produce sufficient NH_4^+ to allow for significant NO_3^- production production. Thus, high rates of mineralization correlate well with high concentrations of the products of prior mineralization (i.e. NO_3^- and NH_4^+). Thus, we feel our finding of strong correlations between organic N turnover and soil pH, Ca:Al ratio, and soluble Al^{3+} have greater significance than does the demonstration of the autocorrelation between activity rates and product accumulation.

The ramifications of these changes in soil N turnover may have significant effects on oak growth and mortality. Oaks, like many other ectomycorrhiza-dependent species, dominate N-poor and N-limited ecosystems (Vogt et al. 1991). Although these tree species may lack the ability to reduce significant amounts of NO_3^- to amine for protein biosynthesis, they can dominate in strongly N-limited conditions because of the ability of their ectomycorrhizal symbionts to forage actively for NH_4^+ and readily mineralizable, low molecular weight N compounds and because of the ability of ectomycorrhizae (ECM) to reduce modest amounts of NO_3^-. Under N-enriched conditions, where P may be as limiting as N, or more so, species of trees which rely on vesicular-arbuscular mycorrhizae (VAM) and which are capable of much greater relative growth rates, such as maples and yellow poplar, may assume dominance (Vogt et al. 1991). Thus, any process which shifts the balance of inorganic N in the soil solution towards a greater NO_3^-:NH_4^+ ratio and increases overall N availability, may negatively affect the competitive ability of oaks and other ECM-dependent species. Furthermore, if the low Ca:Al ratio and high soluble Al^{3+} concentrations we observed in the eastern portion of our gradient can affect ECM fungi directly, then the ECM-dependent oaks would be at even a

greater disadvantage. This sequence of events has been implicated in the decline of ECM-dependent spruces and pines in Europe (e.g. Jansen 1991; Arnolds 1991). Further experimental analysis of the linkages between organic N turnover, soil solution NO_3^-:NH_4^+ ratio, Ca:Al ratio, ECM fungi, and oak growth/mortality in the eastern U.S. is clearly warranted.

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TEMPORAL VARIATION IN PHOTOSYNTHETICALLY ACTIVE RADIATION (PAR) IN MESIC SOUTHERN APPALACHIAN HARDWOOD FORESTS WITH AND WITHOUT *RHODODENDRON* UNDERSTORIES

Barton D. Clinton[1]

Abstract: Understanding spatial and temporal variation in the understory light regime of southern Appalachian forests is central to understanding regeneration patterns of overstory species. One of the important contributors to this variability is the distribution of evergreen shrub species, primarily *Rhododendron maximum* L. We measured photosynthetically active radiation (PAR) in the understory of closed forest canopies in areas with and *without R. maximum*. Measurements were taken from March through November 1993 with a frequency of approximately 3 weeks. In 1994 measurements resumed from March through May. PAR under sub-canopies of *R. maximum* averaged 77% (e.g., 14 vs. 65 μmol $m^{-2}s^{-1}$) lower than in areas without *R. maximum* during the growing season, and 70% (e.g., 179 vs. 641 μmol $m^{-2}s^{-1}$) lower in the dormant season. Variation in PAR during the growing season ranged from 73% to 86% lower in *R. maximum* versus non-*R. maximum* understories. During the growing the season, light levels beneath *Rhododendron* were observed to be < 2% of full sun. Low-light environments associated *with R. maximum* understories are extremely limiting with respect to regeneration of important hardwood species.

INTRODUCTION

The structure of forest canopies is extremely heterogeneous and the spatial distribution of light at the forest floor is a consequence of this heterogeneity (Pukkala and others, 1991). Although soil nitrogen availability commonly limits productivity in most temperate forests (Vitousek and others, 1982), light is typically the most limiting resource under closed canopies in southern Appalachian forests (Wayne and Bazzaz, 1993).

The quantity of light in the understory is a function of the rate of light extinction through the canopy. Utilization efficiency of the residual radiant energy in the understory depends upon the understory tolerance levels of woody plant species and determines their ability to survive and develop under light-limited conditions. It is this difference in understory tolerance among species which determines the compositional and structural characteristics of forests in the southern Appalachians.

The evergreen shrub *Rhododendron maximum* L. is an important component of mesic forest sites in the central and southern Appalachians. This species has been recognized for its influence on microclimatic conditions (Romancier, 1971) and understory composition (Clinton and others, 1994; Monk and others, 1985; Phillips and Murdy, 1985). Even though the mechanisms (eg., competition for water and nutrients, allelopathy, litter depth and quality, light extinction) of its influence on above and belowground conditions are not completely understood, there is little doubt that this species impacts the light environment beneath it. The purpose of this paper is to present data on the temporal variation in the light environments of open versus *Rhododendron* occupied understories in mesic southern Appalachian forests.

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METHODS

Study Design

Sampling was conducted in mature stands of mixed-hardwoods at the Coweeta Hydrologic Laboratory located in the Southern Appalachian physiographic province of western North Carolina. Basal area of these stands is appproximately 25 $m^{-2}ha^{-1}$ most of which is accounted for by *Quercus prinus*, *Q. coccinea*, and *Acer rubrum*. The basal area of *Rhododendron* is approximately 2 $m^{-2}ha^{-1}$. All plots are mid-elevational (800 - 1100 m) and generally have a northerly aspect. Photosynthetically active radiation (PAR) was measured on sunny days between the hours of 1030 and 1330 in open understories and in understories containing continuous *Rhododendron* sub-canopies. Measurements were made on each of 8 plots (6 non-*Rhododendron* and 2 *Rhododendron* plots). Plots were approximately 300 m^2 each containing a permanently marked transect. Along each transect were permanently marked measurement points approximately 2 m apart. Using a portable light meter (Sunfleck Ceptometer, Decagon Devices, Pullman, WA, USA), four measurements of PAR (400 - 700 nm) taken approximately 1 m above the forest floor were averaged and recorded at each point. Measures of full-sun were taken before and after each set of measurements and averaged. Measurements were made at approximate 3-week intervals beginning early in the growing season of 1993 and continuing into the spring of 1994.

Data Analysis

Comparisons were made for PAR between the two understory conditions using PROC ANOVA (SAS, 1987). Significant differences were evaluated at the 0.05 level.

RESULTS

Variability for a given measurement period within each understory condition was low, particularly within *Rhododendron* understories (Fig. 1). However, considerable variability exists between the two understory conditions (Table 1). For each measurement date, total PAR was significantly higher in the non-*Rhododendron* understory (Table 1; Fig. 1). Maximum PAR was observed during the spring before full leafout when non-*Rhododendron* areas had PAR fluxes from 600 - 900 $\mu m\ m^{-2}s^{-1}$ (Table 1). *Rhododendron* areas showed maxima during the same period, but PAR was considerably less (125 - 200 $\mu m\ m^{-2}s^{-1}$) (Table 1). Light levels were most similar, both in terms of PAR and percent full sun during the months of September and November (Table 1; Fig. 1 and 2), and the highest degree of variability occurred during the month of July as evidenced by the high C.V. for those dates (Table 1). Over the entire sampling period, PAR in *Rhododendron* areas ranged from 12% - 33% of observed PAR in non-*Rhododendron* areas. Throughout the growing season, light levels in non-*Rhododendron* areas were observed to be <10% full sun (Fig. 2). Furthermore, levels under *Rhododendron* sub-canopies were less than 2% of full sun (Fig. 2). PAR levels early in the growing season (Fig. 1) may be sufficient for a limited number of species to germinate during that period.

Table 1. Means and associated test statistics for PAR in rhododendron versus non-rhododendron understories by measurement date. Values in parentheses represent the fraction of full-sun.

	PAR(μmols/m^2/s)				
Date	non-Rhodo	Rhodo	F	P>F	C.V.
Apr 16	613.7 (0.393)	206.4 (0.131)	35.07	0.0001	59.32
Apr 4	663.7 (0.440)	169.7 (0.148)	41.88	0.0001	62.4
Jun 2	85.2 (0.052)	12.2 (0.007)	12.06	0.0006	135.56
Jun 22	55.3 (0.043)	14.0 (0.011)	10.99	0.0017	86.51
Jul 6	92.7 (0.060)	13.9 (0.011)	4.46	0.0398	159.81
Jul 21	87.0 (0.076)	23.4 (0.016)	6.64	0.0114	153.38
Aug 16	39.8 (0.031)	10.8 (0.007)	22.67	0.0001	82.70
Sep 1	37.7 (0.027)	10.1 (0.007)	21.95	0.0001	84.44
Nov 22	189.9 (0.324)	22.2 (0.028)	47.43	0.0001	72.65
Mar 4	463.2 (0.392)	127.5 (0.103)	45.18	0.0001	58.15
Apr 1	734.6 (0.496)	191.2 (0.133)	76.8	0.0001	45.73
Apr 18	789.0 (0.539)	126.6 (0.103)	139.8	0.0001	39.68
May 2	396.4 (0.220)	55.2 (0.043)	33.56	0.0001	83.60

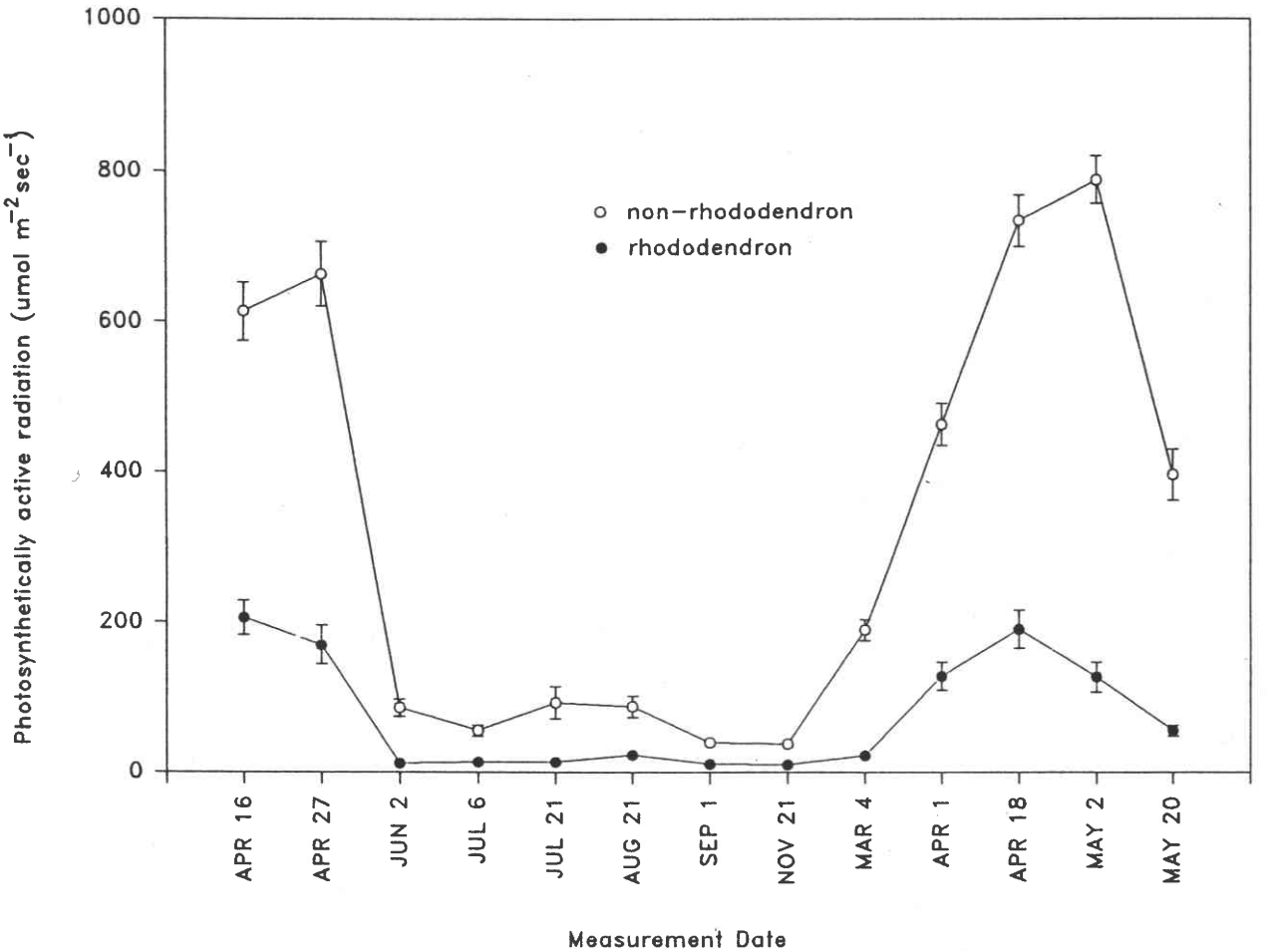

Figure 1. Measured photosynthetically active radiation (PAR) in *Rhododendron* versus non-*Rhododendron* understories. Error bars represent +/- 1 standard error.

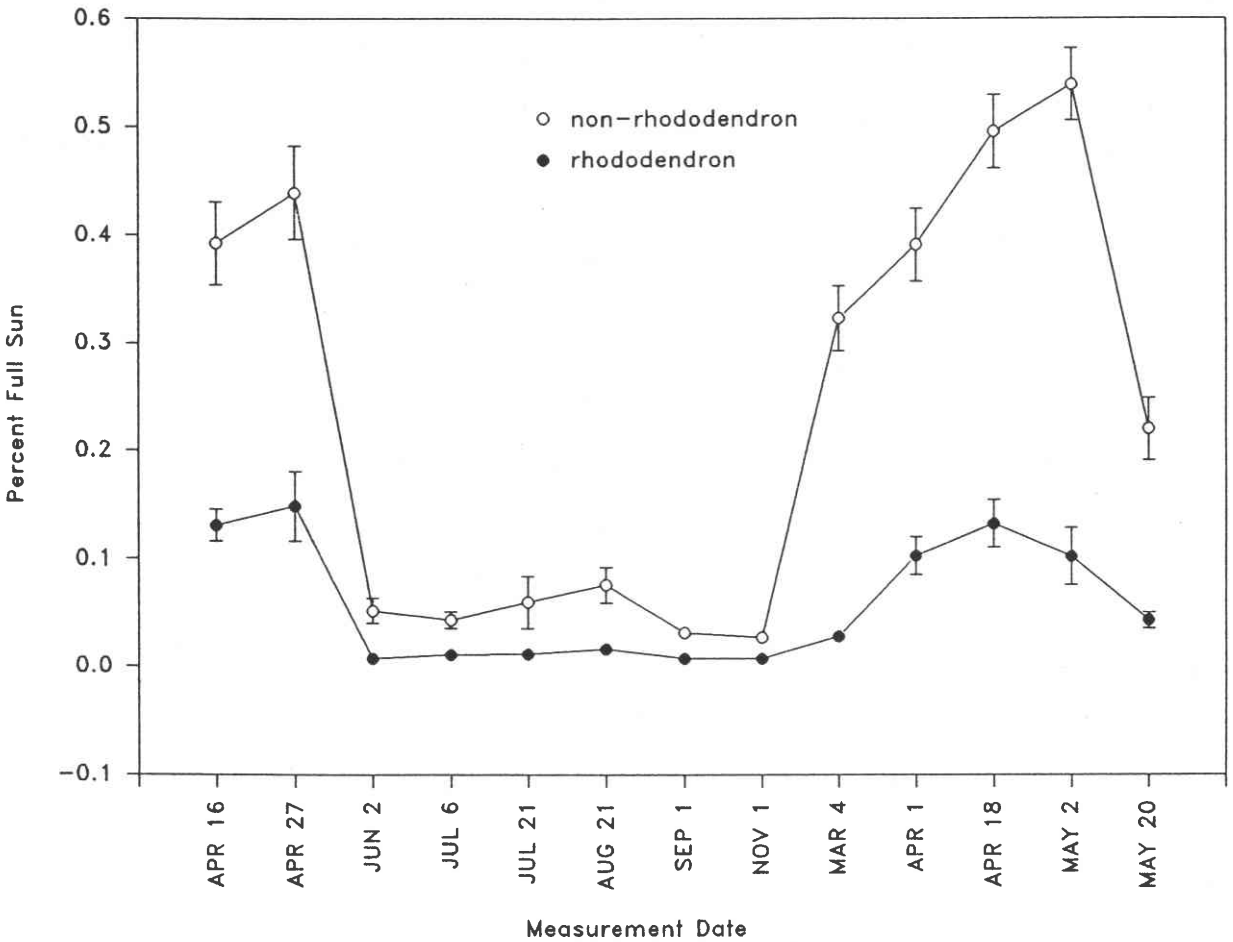

Figure 2. Percent full sun in *Rhododendron* versus non-*Rhododendron* understories. Error bars represent +/- 1 standard error.

DISCUSSION

The light environment beneath *Rhododendron* sub-canopies was observed to be significantly different from non-*Rhododendron* understories throughout the sampling period. These differences may account for the general lack of hardwood regeneration in areas occupied by this species. The light compensation point for most woody species is usually about 2% full-sun (Salisbury and Ross, 1978) which was the level observed for *Rhododendron* areas during most of the growing season in this study. This result suggests that most woody species would simply be unable to become established in such light-limited environments. A few extemely shade tolerant species (e.g., *Acer saccharum* Marsh., *Tsuga canadensis* [L.] Carr.) have limited success in regenerating under *Rhododendron* (Clinton and others, 1994); however, they are not important components of mid-elevation southern Appalachian overstories (Day and others, 1988).

Phillips and Murdy (1985) found that tree regeneration on permanent plots at Coweeta was lower in areas containing heavy *Rhododendron* and that the difference in regeneration between *Rhododendron* and non-*Rhododendron* areas

increased with time. Clinton et al. (1994) observed hardwood seedling density to be significantly less in *Rhododendron* versus non-*Rhododendron* areas (0.5 vs. 2.1 stems m^{-2}). In a germination experiment, Clinton and Vose (unpublished data) found that only 3% of planted *Acer rubrum* L. seed germinated under *Rhododendron* compared with 15% in non-*Rhododendron* areas. Although the low light levels observed under *Rhododendron* in this study may account for these differences, other sources of inhibition may also exist. For example, during the brief period in early spring preceding overstory foliar development, PAR may be adequate, but temperature requirements for germination of important oak species may not be satisfied.

SUMMARY

Rhododendron maximum is an important structural and functional component of southern Appalachian forests. The ecological implications associated with low light environments beneath *Rhododendron* are substantial. Its influence on tree replacement could impact overstory composition (Clinton and others, 1994; Phillips and Murdy, 1985), in addition to potential impacts on nutrient cycling processes. Most overstory species in the southern Appalachians are intolerant to imtermediately tolerant to shade. Where loss of the overstory through natural mortality occurs in areas occupied by *Rhododendron*, what is the probability that the pre-mortality composition of the overstory will be replaced? Given the apparent influence of this species on tree regeneration either due to low light or other phenomena, and the recent observed increase in overstory mortality in the southern Appalachians (Clinton and others, 1993; Starkey and others, 1989; Tainter and others, 1984), it is not likely that overstory composition will be maintained in areas occupied by *Rhododendron*.

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SHORT TERM EVALUATION OF HARVESTING SYSTEMS FOR ECOSYSTEM MANAGEMENT

Michael D. Erickson[1], Penn Peters[2], and Curt Hassler[1]

Abstract: Continuous time/motion studies have traditionally been the basis for productivity estimates of timber harvesting systems. The detailed data from such studies permits the researcher or analyst to develop mathematical relationships based on stand, system, and stem attributes for describing machine cycle times. The resulting equation(s) allow the analyst to estimate productivity and thereby cost, over a range of conditions, as set forth in the mathematical model. The traditional weakness in this process has been the time it takes to collect, synthesize, analyze, and implement the results of the study. For field applications, the turnaround time is such that the original problem or concern often goes unsatisfied or is dealt with by some other means well before the analysis can offer a solution.

The purpose of this study was to develop a methodology for estimating productivity that provides reasonable estimates with quick turnaround time on the analysis. The goal is to collect data in one day and to provide answers almost immediately. The ability to assess productivity of harvesting systems quickly and reliably and to then tie system performance to site impacts, can aid immeasurably in ecosystem management.

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HERBACEOUS VEGETATION IN THINNED AND DEFOLIATED FOREST STANDS IN NORTH CENTRAL WEST VIRGINIA

S. L. C. Fosbroke, D. Feicht, and R. M. Muzika[1]

Abstract: Herbaceous vegetation was inventoried in 1992 and 1993 in eight Appalachian mixed hardwood stands (< 50% basal area/acre in oak species) and eight oak stands (> 50% basal area/acre in oak species) in north central West Virginia. Vegetation was sampled on 20 6-foot radius plots per stand twice each growing season (once during late spring to sample spring ephemeral species, and later in mid summer). Stand disturbance prior to herbaceous sampling included 1) a silvicultural thinning (8 stands) in the winter/spring of 1990, and/or 2) moderate to heavy gypsy moth defoliation (6 stands) in 1990 and 1991. Most overstory mortality occurred within three years of the first defoliation. By the fall of 1993, residual overstory basal area ranged from 30 - 74 ft^2 / acre for (3) thinned, defoliated stands; from 38 - 66 ft^2 / acre for (3) unthinned, defoliated stands; from 78 - 104 ft^2 / acre for (5) thinned, undefoliated stands; and from 117 - 132 ft^2 / acre for (5) unthinned, undefoliated stands.

In 1992, species richness ranged from 22 - 65 in mixed hardwood stands and from 19 - 38 in oak stands. In 1993, species richness ranged from 24 - 70 in mixed hardwood stands and from 23 - 42 in oak stands. Regardless of thinning or defoliation disturbance, the dominant herbaceous species (i.e. the species that had the highest average percent cover) consisted of common greenbrier (*Smilax rotundifolia*), Hayscented Fern (*Dennstaedtia punctilobula*), or New York Fern (*Thelypteris noveboracensis*); the one exception was stand #6, a mixed hardwood stand with the highest species richness values recorded in the study. From 1992 to 1993, 9 of the 11 disturbed stands showed an increase in average percent cover of the dominant herbaceous species; stand #6 (a thinned, undefoliated stand) switched its dominant cover species, and stand #8 (an unthinned, defoliated stand) had its dominant species' coverage drop 0.4% between 1992 and 1993. Four of the five undisturbed stands had no change or a decrease in the average percent cover of the dominant herbaceous species from 1992 to 1993. Stand #12 was the only exception; its dominant herbaceous species increased 1.4% from 1992 to 1993. Detrended Correspondence Analysis (DCA) was used to further examine relationships among herbaceous species and stand/site characteristics.

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DEFOLIATION AND MORTALITY PATTERNS IN FORESTS SILVICULTURALLY MANAGED FOR GYPSY MOTH

Kurt W. Gottschalk and Rose-Marie Muzika[1]

Abstract: Mixed hardwood forests of the Appalachian region support one of the most diverse communities of woody plants in North America, but the composition and relative dominance of the forest changes substantially with slight changes in physiography, soil type, or microclimate. Composition of oak and other species highly preferred by the gypsy moth determines the susceptibility of forests to defoliation. As composition of these species increases, the susceptibility increases. Silvicultural treatments were developed to minimize the effects of gypsy moth defoliation on forests. A research study was conducted to determine the effectiveness of two preoutbreak silvicultural treatments that are applied in different susceptibility stands. Presalvage thinning is designed to reduce the mortality that occurs in stands by taking out the trees with highest estimated probabilities of mortality and increasing the growth and vigor of the residual trees. It is applied in high or very highly susceptible stands (>50 percent oak). Sanitation thinning is designed to reduce the defoliation level (and subsequent mortality) in medium susceptibility stands (20 to 50 percent oak). This reduction is obtained by decreasing oaks and other susceptible species to 20 percent of the stand basal area and increasing vigor of the residual trees. Four replications of each treatment were established in 1989-90, with a paired untreated control stand for every treated stand.

Defoliation and mortality patterns over a five-year period appear to be stand dependent. While the general trend indicates that heavy defoliation of susceptible species occurred in 1990 and 1991, only six of the 16 stands sustained heavy defoliation. Defoliation patterns were nearly identical for both thinned and unthinned stands in a pair, although the thinned stands averaged slightly lower defoliation across all species. Mortality, expressed as percent of original basal area, was greater in the unthinned stands versus the thinned stands of each pair of the six stands that were defoliated. Mortality in stands that were presalvage thinned was 33.8 percent and 63.5 percent as opposed to 69.0 percent and 68.1 percent in the unthinned counterparts. The sanitation thinning mortality was 29.8 percent versus 41.4 percent for the unthinned stands. The majority of the defoliation and mortality occurred in oak and other susceptible species; very little occurred in resistant and immune species. These results represent worst case tests of the silvicultural treatments, as the treatments were completed in April 1990 and defoliation began in May 1990, so the stands had no time to adjust to the treatments and for the residual trees to increase in vigor. The results are encouraging in that they do show potential for silvicultural treatments to minimize gypsy moth effects on forest stands.

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A GENERALIZED INGROWTH MODEL FOR THE NORTHEASTERN UNITED STATES

Linda S. Gribko[1], Donald E. Hilt[2], and Mary Ann Fajvan[1]

Abstract: Ingrowth, the number of trees that periodically grow into the smallest inventoried diameter class, has long been recognized as a basic element of multicohort or, uneven-aged, stand development. However, very little information is available to aid forest managers in the estimation of ingrowth. The purpose of this study was to develop a generalized ingrowth model for the northeastern United States. The model was developed in two stages. First, annual ingrowth trees per acre into the 5-inch dbh class was modeled as a function of average stand diameter, overstory trees per acre, and a generalized stocking percent. Generalized stocking percent and overstory species composition, as a function of relative shade tolerance, were then used in a multivariate logistic model to predict the species composition of the ingrowth trees.

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BLACK WALNUT RESPONSE TO SUBSOILING, IRRIGATION, AND VEGETATION MANAGEMENT ON A SITE WITH A SHALLOW FRAGIPAN

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Abstract: Vegetation management with glyphosate and simazine proved to be more effective than preplant subsoiling or irrigation for achieving acceptable walnut biomass growth on an upland old field site (SI = 70 for white oak). In 1980, we direct seeded germinating black walnut seed on an upland, slightly eroded, old field ridge with a 45 to 60 cm deep fragipan. We tested all combinations of with and without preplant subsoiling (75 cm deep); with and without irrigation for 6 years, and with and without chemical weed control for 7 years. The only treatment effect still persisting after 14 years was vegetation management reducing the tall fescue competition. Preplanting ripping showed benefits for 4 to 8 years after planting only. Irrigation either reduced biomass growth or had no effect in combination with weed control. Our results continue to confirm that with intensive management black walnut can sustain acceptable growth and produce nuts above a 3-m long veneer log on agricultural lands with low site index for black walnut.

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DISTRIBUTION, DISPERSAL AND ABUNDANCE OF HAYSCENTED FERN SPORES IN MIXED HARDWOOD STANDS

Larry H. McCormick and Kathy A. Penrod[1]

Abstract: A study was conducted in 1992 to assess the abundance and distribution of viable hayscented fern spores in the forest floor of central Pennsylvania hardwood stands before and after seasonal spore dispersal. Intact soil samples were collected at various distances and directions from established fern communities and placed in a greenhouse to effect spore germination. Spore abundance was estimated by counting the number of gametophytes which developed. Results indicate that a large viable sporebank exists in many central Pennsylvania hardwood forests. Viable spores were present in over 97% of pre- and post-dispersal samples with estimates of upward to 160,000 viable spores per square meter. Viable spores occurred at all distances within the 50 m from source zone sampled. The number of viable spores generally decreased with distance from the source with the highest abundance occurring within 10 m of the source. There was evidence of directional effects on spore abundance at some sampling locations, however, the effect was variable between sites. These data indicate the potential exists for the establishment of hayscented ferns from a viable sporebank in central Pennsylvania hardwood stands.

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IDENTIFICATION OF CANOPY STRATA IN ALLEGHENY HARDWOOD STANDS

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Abstract: Allegheny hardwood stands typically develop vertical canopy layers, or strata, due to differential species-specific growth rates and tolerance to shade. Across the Allegheny Plateau, black cherry dominates the main canopy, while sugar maple, American beech, and red maple are relegated to subcanopy strata.

Stratification of Allegheny hardwoods is critical in forest management. For example, diameter-limit cutting of the larger and more valuable black cherry results in a residual stand that consists of sugar maple and beech, compromising the economic potential of the site. In addition, because these stands are stratified by species, where each stratum usually consists of not more than two species, the number and arrangement of the strata influence the diversity of and the regeneration potential in these stands.

This study compares several methods used to identify and characterize individual canopy strata, canopy profiles depicting tree silhouettes along horizontal transects, histograms showing the frequency of trees by diameter or height classes, and structural contour charts expressing the distribution of trees by both height and diameter classes. Structural contour charts represent a novel approach to depicting and evaluating canopy strata.

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A METHOD FOR APPLYING GROUP SELECTION IN CENTRAL APPALACHIAN HARDWOODS

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Abstract: Public concern over the adverse visual impact of clearcutting has heightened interest in developing and testing alternative regeneration practices for central Appalachian hardwoods. When applied properly, group selection, which entails making small openings within a timber stand at regular intervals, can meet aesthetic goals while providing suitable light conditions to reproduce shade-intolerant species. Volume control and residual stand density are used to regulate periodic cuts, which include volume removed to create openings plus volume removed between openings to improve the quality and distribution of the residual stand. In central Appalachian hardwoods, openings must have a minimum size of 0.4 acre; all stems 1.0 inch d.b.h. and larger are cut to reproduce desirable shade-intolerant species. Maximum opening size is based on aesthetic requirements or other management constraints. Where reproduction of shade-tolerant species is acceptable, openings can be as small as a space occupied by a few trees. Openings should be located using the *worst first* approach to give the growing space occupied by mature trees or risky trees to faster growing, desirable regeneration. The residual stand between openings should be improved by cutting poor-quality or high-risk trees. The recommended residual basal area in sawtimber-size trees (11.0 inches d.b.h. and larger) is related to northern red oak site index (SI): 70 to 85 ft²/acre for SI 80, 55 to 70 ft²/acre for SI 70, and 40 to 55 ft²/acre for SI 60. These field-tested methods can help forest managers initiate group selection in second-growth Appalachian hardwoods. Guidelines are presented for computing the cut, determining size, location, and number of openings, and marking the stand.

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FIFTY-YEAR RESPONSE OF A 135-YR-OLD WHITE PINE STAND TO PARTIAL THINNING IN CONNECTICUT

Darryl S. Nicholson and Jeffrey S. Ward[1]

Abstract: In the early 1930s a series of plots were established throughout Connecticut to study the effects of thinning on stand growth. Gold's Pine plot (1.0 acre) was established in 1932 to study partial thinning in a mature 125-yr-old white pine/eastern hemlock stand. The plot is located within Housatonic State Forest in western Connecticut. In 1932 the stand volume was estimated to be 49 Mbf, 75% was white pine and the remainder was hemlock. The stand was thinned in 1944 and 1961. Approximately 4 Mbf/acre and 3 Mbf/acre were harvested in 1944 and 1961, respectively. Stand volume in 1994 was estimated to be 53 Mbf/acre. Mean diameter of white pines has increased from 20.7 to 28.8 inches. Over the 60-yr period the height of codominants and dominants has increased from 110 to 128 ft. There were 42 stems/acre in the understory (2-7 inches dbh) in 1932 consisting of eastern hemlock (64%), black birch (24%), white pine (2%), yellow birch (2%), white ash (2%), black cherry (2%), and sugar maple (2%). In 1994 there were 440 stems/acre in the understory consisting of eastern hemlock (55%), black birch (32%), yellow birch (4%), striped maple (4%), and sugar maple (4%). This case study suggests that mature white pine stands can be partially thinned to recover declining material and release stagnating trees without sacrificing continued volume growth and "Big Tree" stand characteristics.

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FOREST STAND DEVELOPMENT ON 6-26 YEAR-OLD CLEARCUTS IN SOUTHEASTERN OHIO[1]

Eric R. Norland and David M. Hix[2]

Abstract: Clearcutting, a commonly-used cutting practice in southeastern Ohio, often results in a forest stand with a different species composition than the parent stand. The time frame during which species shifts occur is unclear. While some studies have documented species composition at specific points in time, none have attempted to examine changes throughout the first decades of stand development. This study focused on examining the early-successional dynamics of young, mixed-species forests of southeastern Ohio. Species compositions were examined across a chronosequence of sixteen stands that developed 6-26 years following clearcutting. The sample was limited to dry-mesic hardwood forest ecosystems found on southerly aspects and on soils derived from residuum or colluvium. Over the 20-year period, stand density decreased from 17,636 stems/ha at age 6 to 2,759 stems/ha at age 26. During the same period, basal area increased from 8.2 m^2/ha to 22.1 m^2/ha. Oaks (*Quercus* spp.) declined from 3,386 stems/ha to 581 stems/ha. When considered as a proportion of the total stand, however, their proportion was relatively stable, averaging 21.3%. Oak importance value (IV=[relative density + relative basal area]/2) in the upper canopy (dominant and codominant crown classes) increased from 33.9 to 77.4% over the 20-year chronosequence. Chestnut oak (*Quercus prinus* L.) was the major species throughout the chronosequence. White oak (*Quercus alba* L.) and black oak (*Quercus velutina* Lam.) were minor components by age 26 years, although they dominated a comparison sample of six mature stands of the same ecosystem type. Yellow-poplar (*Liriodendron tulipifera* L.) was abundant 6-8 years after clearcutting, but nearly absent by age 26 years. Red maple (*Acer rubrum* L.) was the major species in both the intermediate and overtopped crown classes throughout the chronosequence. Predictive models were developed for stand density and basal area. A multiple logit model predicted increasing proportions of shade-tolerant species, although the predicted trends differed between crown classes. Models based on data from a chronosequence of forest stands can be used by resource managers to predict future composition and structure of an individual stand during its early decades of development. By anticipating what a stand may look like 20 years hence, management practices can be prescribed when it is desirable to alter the course of natural development.

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VEGETATION ANALYSIS, ENVIRONMENTAL RELATIONSHIPS, AND POTENTIAL SUCCESSIONAL TRENDS IN THE MISSOURI FOREST ECOSYSTEM PROJECT

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Abstract: The vegetation data set of the Missouri Forest Ecosystem Project (MOFEP, initiated by the Missouri Department of Conservation) in the Ozark Mountains of southeastern Missouri was ordinated by Detrended Correspondence Analysis (DCA) to identify vegetation gradients and potential environmental influences. The first DCA axis represented a strong gradient in black oak importance and segregation of this species from white ash, black walnut, eastern redcedar, sugar maple and bitternut hickory. Some Spearman rank correlation coefficients between Axis 1 and measured environmental variables were significant but not high, suggesting that unmeasured factors (e.g., disturbance history, soil pH) were associated with species distribution along the first axis. The second axis clearly separated mesic species (bitternut hickory, black walnut, white ash and sugar maple) from xeric species (shortleaf pine, eastern redcedar, post oak and blackjack oak). Aspect and aspect x slope variables correlated strongly with Axis 2 scores and species importance, suggesting that this axis represented a habitat moisture gradient. Potential canopy replacement in MOFEP forests was examined by analysis of overstory-understory relationships and density vs. size class distributions. White oak likely will become more important in MOFEP forests with time, as this species is relatively abundant in the understory, while relative abundance of red oak group species and shortleaf pine is substantially reduced compared to that in the overstory.

INTRODUCTION

The Missouri Forest Ecosystem Project (MOFEP) in the Ozark Mountains of southeastern Missouri represents one of the most extensive and comprehensive long-term ecological studies undertaken in the eastern United States (Brookshire and Hauser 1993). There have been few quantitative studies of the vegetation in the region and none of this extent (Nigh and others 1985, Read 1952, Ware and others 1992, Zimmerman and Wagner 1979). The baseline vegetation data set collected for this project offered an opportunity to provide an initial, coherent view of the nature and condition of this ecosystem and its plant population dynamics. Specifically, this study was undertaken to provide an ecological analysis of MOFEP forests with regard to tree species composition and dominance, vegetation-environment relationships, and potential successional trends.

METHODS

Data Analysis

The nine MOFEP study sites are located in Carter, Ripley and Shannon counties in the southeastern Missouri Ozarks and range in size from 260 and 527 ha. Study areas, vegetation plot design and data collection were described in detail by Brookshire and Hauser (1993). For the purposes of the present study, data consisted of 644 0.2 ha circular "tree" plots on which species and diameter of all individuals ≥11.4 cm (4.5 in) dbh were recorded. Additionally, within the boundary of each tree plot, species and diameter were recorded on four nested 0.02 ha plots for woody

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plants ("saplings") that were between 3.8 and 11.3 cm (1.5-4.5 in) dbh. Within each 0.02 ha plot, "seedling" data were recorded by species on a concentric 0.004 ha plot for plants > 1 m tall and < 3.8 cm (1.5 in) dbh.

Indirect gradient analysis of overstory species was conducted using tree plot data and Detrended Correspondence Analysis (DCA) (Hill and Gauch 1980). Tree Importance Value (IV = relative density + relative basal area) by species for each plot was used as an indicator of abundance. As employed in this study, output of the DCA procedure provides gradients in overstory species composition in two or more dimensions. Composition of major forest types on MOFEP study sites was assessed by averaging IV data across plots of the same Ecological Land Type (ELT) (Miller 1981). TWINSPAN analysis (Hill 1979) of the MOFEP data set also was conducted, but provided no detectable advantage in forest classification over that offered by the ELT system (data not shown).

The relationships among DCA axis scores, species importance value and environmental factors were explored by Spearman rank correlation analysis using the CORR procedure of SAS (SAS Institute 1988). Correlation analysis allowed identification of: (1) species that contribute to vegetation gradients extracted in the ordination process and (2) environmental factors that may influence species distribution and dominance across ordination gradients. Spearman rank correlation was used instead of Pearson linear correlation to free the analysis of assumptions of linearity in relationships between species importance and environmental factors and of normality in distribution of vegetation across the ordination gradients (Dollar and others, 1992). Environmental data were obtained from data sets supplied by the Missouri Department of Conservation that contained information for each 0.2 ha plot indicating slope %, aspect (° azimuth) and slope position (1= floodplain, 5=ridgetop). For correlation analysis, aspect was transformed trigonometrically using a function that provided a maximum value (+1) for NE aspects (45°) and minimum value (-1) for SW aspects (225°) (Beers and others 1966):

$$\text{Transformed aspect} = \cos(\text{Aspect} - 45°)$$

This transformation provides a representation of the "mesicness" continuum of topographic orientation, with NE aspects providing the most "mesic," SW aspects the most "xeric" environments, and other aspects being assigned intermediate values along this microenvironmental continuum. Another variable was derived from transformed aspect by multiplying it by slope % based on the tendency of microclimatic influences to be accentuated by the slope of the land (Frank and Lee 1966).

Potential successional trends in these forests were investigated through overstory-understory and size class analysis on groups of plots segregated by ELT classification. Data from the overstory plots mentioned previously and those of the four 0.02 ha sapling plots were merged, and differences between understory (≤19.05 cm dbh) and overstory (>19.05 cm dbh) relative densities of important species of each ELT were assessed by t-tests after arcsin square root transformation (Dollar and others 1992, Sokal and Rohlf 1981).

Plot density (#/ha) of seedlings (from 0.004 ha plots), saplings and trees (as mentioned above) was assembled and segregated by ELT, and data for each species were plotted by diameter class [< 1.5 in (3.8 cm), 1.5-4.5 in(3.8-11.4 cm), 4.5-7.5 in (11.4 cm-19.1 cm), 7.5-10.5 in (19.1-26.7 cm), 10.5-13.5 in (26.7-34.3 cm), 13.5-16.5 in (34.3-41.9 cm), 16.5-19.5 in (41.9-49.5 cm), 19.5-22.5 in (49.5-57.2 cm), 22.5-25.5 in (57.2-64.8 cm), > 25.5 in (64.8 cm) dbh] for further analysis of population structure and potential replacement of canopy trees (Harcombe and Marks 1978, Harcombe 1987, Johnson 1992, McCarthy and Wistendahl 1988). Density vs. size class relationships for different species were compared visually and statistically (using χ^2-based comparison of frequency distributions across size classes) (Sokal and Rohlf, 1981).

RESULTS AND DISCUSSION

Ordination and Classification

Seventy percent of the variation in vegetation composition was displayed in the first two (of four total) DCA ordination axes. The sample ordination graph, using the first two axes, displayed a dense cloud of plots toward the lower left-hand corner, with a far more scattered and sparse distribution of plots to the right and upper portions of the graph (Fig. 1). The DCA species ordination graph (Fig. 2) and correlation analysis of DCA plot scores with

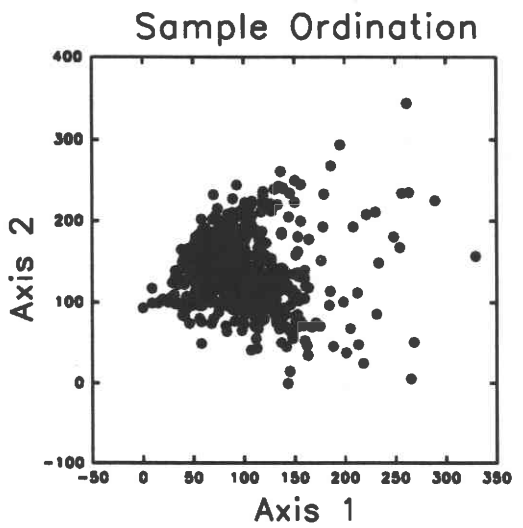

Figure 1. Distribution of plots across two-dimensional ordination space derived from DCA ordination of tree importance values of 644, 0.2 ha plots.

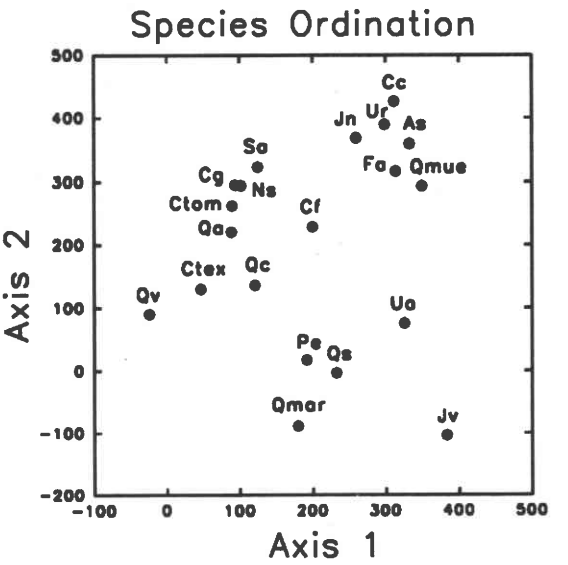

Figure 2. Scatter plot of species across two-dimensional ordination space derived from DCA ordination of tree importance values. Species key: Pe= shortleaf pine; Qc= scarlet oak; Qv= black oak; Qs= post oak; Qa= white oak; Qmue= chinkapin oak; Qmar= blackjack oak; Ns= black gum; Cg= pignut hickory; Cc= bitternut hickory; Ctex= black hickory; Ctom=mockernut hickory; Sa= sassafras; Cf= flowering dogwood; Jv= eastern redcedar; Ur= slippery elm; Ua= winged elm; Fa= white ash; As= sugar maple; Jn= black walnut.

species IV (Table 1) indicated that the first DCA axis represented a strong gradient in black oak (*Quercus velutina* Lam.) importance and segregation of this species from species such as white ash (*Fraxinus americana* L.), black walnut (*Juglans nigra* L.), eastern redcedar (*Juniperus virginiana* L.), sugar maple (*Acer saccharum* Marsh.) and bitternut hickory (*Carya cordiformis* (Wangenh.) K. Koch). Black oak was more abundant in plots with low Axis 1 scores while the latter group of species was important in plots with high Axis 1 scores. In a similar fashion, the second DCA axis clearly separated mesic species such as bitternut hickory, black walnut, white ash, and sugar maple (high Axis 2 scores) from xeric species such as shortleaf pine (*Pinus echinata* Mill.), eastern redcedar, post oak (*Quercus stellata* Wangenh.) and blackjack oak (*Quercus marilandica* Muenchh.) (low Axis 2 scores).

When Axis 1 scores were correlated with environmental factors, some relationships were apparent, with the strongest correlations shown between axis scores and slope position (Table 2A). In practical terms, this correlation indicated that black oak was more important on ridgetop positions. However, in general, environmental correlations with Axis 1

Table 1. Spearman correlation coefficients between DCA axis scores and species importance values. ** $p \leq 0.01$; * $p \leq 0.05$

	Ordination Axis	
Species	Axis 1	Axis 2
ACERSAC*	0.24**	0.25**
CARYCOR	0.18**	0.25**
CARYGLA	-0.11**	0.46**
CARYTEX	-0.24**	0.06
CARYTOM	-0.08*	0.42**
CORNFLO	0.02	0.25**
FRAXAME	0.29**	0.25**
JUGLNIG	0.16**	0.39**
JUNIVIR	0.32**	-0.04
NYSSSYL	-0.02	0.42**
PINUECH	0.44**	-0.53*
QUERALB	-0.03	0.57**
QUERCOC	0.12**	0.02
QUERMAR	0.12**	-0.48*
QUERMUE	0.47**	0.19**
QUERSTE	0.31**	-0.57*
QUERVEL	-0.89**	-0.25*
SASSALB	-0.03	0.27**
ULMUALA	0.39**	0.08
ULMURUB	0.28**	0.35**

*Species codes for Tables 1, 2 and 3. ACERSAC = sugar maple; CARYCOR = bitternut hickory; CARYGLA = pignut hickory; CARYTEX = black hickory; CARYTOM = mockernut hickory; CORNFLO = flowering dogwood; FRAXAME = white ash; JUGLNIG = black walnut; JUNIVIR = eastern redcedar; NYSSSYL = black gum; PINUECH = shortleaf pine; QUERALB = white oak; QUERCOC =scarlet oak; QUERMAR = blackjack oak; QUERMUE = chinkapin oak; QUERSTE = post oak; QUERVEL = black oak; SASSALB = sassafras; ULMUALA = winged elm; ULMURUB = slippery elm.

were not strong (i.e., less than 0.3), suggesting that some unmeasured factor likely was important in determining species composition across this axis. Plausible candidates are disturbance history and soil factors (especially pH, Nigh and others 1985, Ware and others 1992). In contrast to these relatively weak Axis 1-environment correlations, the transformed aspect and aspect x slope variables correlated more strongly (r = 0.43 to 0.46) with Axis 2 DCA sample scores. This result is consistent with the distribution of species shown in the species ordination graph (Fig. 2). Hence Axis 2 of the DCA ordination appeared to represent a habitat moisture gradient.

MOFEP Forest Composition

Forests of the two most common ELT classifications (17 and 18) (Table 3) consisted of nearly equal abundances of black and scarlet oak. White oak (*Quercus alba* L.) was somewhat greater in importance on ELT 18 plots (side slopes N/E aspects; deep soils; Miller, 1981) than on 17 plots (side slopes S/W aspects; deep soils). In contrast, shortleaf

Table 2. Spearman correlation coefficients (with probability level below) between DCA ordination axis scores and environmental variables (A) and between environmental variables and species Importance Values (B). For species codes see Table 1.

(A)

	Axis 1	Axis 2
Slope (%)	0.001 0.98	0.08 0.04
Slope position	-0.29 <0.01	-0.25 <0.01
Aspect*	-0.22 <0.01	0.43 <0.01
Aspect x slope*	-0.19 <0.01	0.46 <0.01

(B)

	ACERSAC	CARYCOR	CARYGLA	CARYTEX	CARYTOM	CORNFLO	FRAXAME	JUGLNIG	JUNIVIR	NYSSSYL
Slope (%)	-0.02 0.59	-0.08 0.04	-0.10 0.01	-0.07 0.10	0.02 0.69	0.15 <0.01	0.001 0.98	-0.07 0.07	0.09 0.03	0.12 <0.01
Slope position	-0.13 <0.01	-0.14 <0.01	0.01 0.76	0.07 0.10	-0.12 <0.01	-0.16 <0.01	-0.13 <0.01	-0.05 0.25	-0.05 0.23	-0.22 <0.01
Aspect*	0.01 0.94	0.03 0.46	0.29 <0.01	0.15 <0.01	0.19 <0.01	0.26 <0.01	0.03 0.43	0.10 <0.01	-0.07 0.06	0.22 <0.01
Aspect x Slope*	0.01 0.90	0.05 0.24	0.33 <0.01	0.15 <0.01	0.21 <0.01	0.28 <0.01	0.07 0.07	0.12 <0.01	-0.10 0.01	0.23 <0.01

	PINUECH	QUERALB	QUERCOC	QUERMAR	QUERMUE	QUERSTE	QUERVEL	SASSALB	ULMUALA	ULMURUB
Slope (%)	0.10 <0.01	0.12 <0.01	0.15 <0.01	-0.04 0.32	0.01 0.81	-0.26 <0.01	-0.07 0.06	0.01 0.99	0.04 0.34	-0.04 0.36
Slope position	-0.01 0.97	-0.20 <0.01	-0.14 <0.01	0.17 <0.01	-0.25 <0.01	0.13 <0.01	0.37 <0.01	-0.07 0.07	-0.22 <0.01	-0.16 <0.01
Aspect*	-0.40 <0.01	0.19 <0.01	0.04 0.37	-0.32 <0.01	-0.09 0.03	-0.30 <0.01	0.04 0.32	0.18 <0.01	-0.04 0.35	0.04 0.37
Aspect x Slope*	-0.45 <0.01	0.17 <0.01	0.02 0.66	-0.32 <0.01	-0.06 0.12	-0.27 <0.01	0.01 0.87	0.19 <0.01	-0.04 0.33	0.04 0.26

* Aspect transformed trigonometrically (see text).

Table 3. Mean Tree Importance Values of MOFEP plots classified by Ecological Land Type (ELT). (n= number of plots in ELT)

	ELT*												
	1	5	6	7	11	15	17	18	19	20	21	22	23
n=	1	28	2	6	89	5	257	213	19	8	4	3	9
SPECIES													
ACERSAC*		1.46	4.10		0.02		0.15	0.10	3.00	1.13			0.42
CARYCOR		3.81	7.70		0.03	0.38	0.05	0.14	0.76				
CARYGLA	19.40	11.77	32.15	6.15	10.87	5.60	3.80	10.98	3.14	5.85	0.55	0.70	1.55
CARYTEX	10.70	9.59	14.10	11.75	12.35	8.14	7.06	8.87	6.28	14.65	8.92	4.13	4.47
CARYTOM	10.80	9.96	15.05	35.15	7.52	5.86	6.22	10.01	4.76	5.77		2.73	0.98
CORNFLO	11.50	1.45		1.56	0.43	0.32	0.97	2.55	2.13	0.48		4.30	0.94
FRAXAME		2.27	2.65		0.22		0.34	0.37	3.89	2.21		1.70	1.36
JUGLNIG		2.92	12.80	1.31	1.71		0.39	0.97	2.59	0.75			
JUNIVIR							0.37	0.03	1.78		10.10	0.60	9.58
NYSSSYL	26.10	6.21	16.40	12.31	1.42	1.04	2.42	4.45	1.35	7.32	0.80	1.03	
PINUECH		11.04		9.41	12.75	9.32	25.25	7.45	31.35	15.36	22.17	13.03	25.35
QUERALB	63.70	43.03	45.95	28.23	30.03	16.58	38.98	47.40	32.63	66.61	20.62	21.16	10.72
QUERCOC	26.10	28.53	24.30	36.00	28.79	14.68	46.88	47.96	35.05	40.26	40.52	15.30	35.76
QUERMAR		0.33		0.36	2.88	9.08	3.05	0.24	0.87	0.47	33.55	17.46	15.52
QUERMUE		5.71	2.40	3.16			0.89	0.49	25.23	3.13	4.15	15.13	20.77
QUERSTE	6.40	25.58		15.56	18.15	16.38	12.48	7.09	29.20	9.97	11.42	46.30	40.67
QUERVEL	20.70	20.80	10.95	30.01	71.80	112.44	49.13	48.28	8.93	18.65	34.00	29.56	26.76
SASSALB		0.49		2.85	0.16		0.15	0.53	0.11	0.85			
ULMUALA		1.52	1.15	0.55	0.01		0.35	0.21	2.56	0.80	13.22	0.73	1.53
ULMURUB		5.82	3.40	0.68	0.25		0.27	0.40	1.77	0.22		0.56	0.23

*Species code: see Tab. 1. ELT Codes (Miller, 1981): 1-Low flood plains/loamy and mixed alluvial soils; 5,6-Upland waterways/deep cherty soils; 7-Toe slopes/deep loamy soils; 11-Narrow ridges/deep cherty soils; 15-Broad ridges/deep loamy and clayey soils; 17-Side slopes/South and West aspects/deep soils; 18-Side slopes/North and East aspects/deep soils; 19-Side slopes/South and West aspects/shallow soils; 20-Side slopes/North and East aspects/shallow soils; 21-23-Side slopes/very shallow limestone soils

pine was more important on ELT 17 plots. Other tree species on these plots generally had importance values greater than 1 and less than 15 and included pignut, black and mockernut hickories (*Carya glabra* (Mill.) Sweet, *C. texana* Buckl., *C. tomentosa* (Poir.) Nutt.), black gum (*Nyssa sylvatica* Marsh.) and post and blackjack oak. These hickory species and black gum tended to be more important on ELT 18 plots, while blackjack and post oaks were more important on ELT 17 plots. Plots of ELT 11 (narrow ridges/deep cherty soils) were dominated by black oak with smaller contributions by scarlet (*Quercus coccinea* Muenchh.), white and post oaks. Lower importance values were associated with shortleaf pine, the three above-mentioned species of hickory, and black gum. Plots of ELT 19 (side slopes S/W aspects; shallow soils) were characterized by the highest shortleaf pine and chinkapin oak (*Quercus muehlenbergii* Engelm.) importance values among all ELT classes, with similar contributions by scarlet, white, and post oaks. Most important on plots characterized as "Upland Waterways" (ELT 5, 6) was white oak, followed by scarlet, post and black oak, and shortleaf pine and previously-mentioned species of hickory. Moreover, there was a small, but notable, contribution on ELT 5 plots of mesic species such as black walnut, black gum, sugar maple bitternut hickory, and slippery elm (*Ulmus rubra* Mühl). Although the sample sizes of remaining ELT classes were quite small, it is worth noting that white oak importance was quite high on ELT 20 (side slopes N/E aspects; shallow soils) and those species that prefer soil high in calcium, such as eastern redcedar and chinkapin oak (Fowells 1965), were important on ELT 21-23 plots.

Correlation of species importance values with environmental parameters (Table 2B) reinforced the qualitative relationships noted above between ELT-based environmental characterizations and species distribution and abundance. Species associated with ELTs categorized as having South and West-facing slopes (shortleaf pine and

blackjack and post oaks) tended to be negatively correlated with the transformed aspect and aspect x slope % variables. In contrast, species that were more important on ELTs associated with North- and East-facing slopes (white oak, species of hickory, black gum, and black walnut) tended to be positively and significantly correlated with these two parameters. Black oak correlated most highly with slope position, being most important on ridgetop positions and showing little correlation with aspect. In contrast, more mesic-site species (for this region) such as sugar maple, bitternut hickory, white ash, black gum, white oak, and winged (*Ulmus alata* Michx.) and slippery elm were associated with lower slope positions.

MOFEP plots show species composition and abundance typical of the generally xeric forests of the region (Braun 1972, Nelson 1985, Read 1952, Ware and others 1992). On ridgetop and upper slope forests a combination of black, scarlet, post and white oaks and shortleaf pine dominate, with relative importance shifting among black oak (ridgetops), scarlet, black and post oaks and shortleaf pine (S- and W-facing slopes), and white oak and hickories (N- and E-facing slopes). On lower slopes white oak tends to attain an importance similar to that associated with mesic (North and East) aspects on upper slopes. On the relatively infrequent mesic sites associated with upland draws and ephemeral streams, mesic species of the region such as sugar maple, bitternut hickory, black gum and black walnut, along with more abundant white oak may be found.

According to Nelson's (1985) scheme, MOFEP forests would appear to variously fit the classifications of xeric to dry-mesic forests on limestone/dolomite, chert and sandstone substrates. This conclusion is consistent with that of Brookshire and Hauser (1993). MOFEP forests similarly fit into Braun's (1972) categorization of Oak-Pine and southern division Oak-Hickory forests. There were some exceptions to Nelson's and Braun's descriptions, however. Particularly noteworthy was the scarcity of northern red oak (*Quercus rubra* L.) in MOFEP forests despite its inclusion in forests of the region by both Nelson and Braun. Similarly, Ware and others (1992) identified northern red oak as a common constituent of upland forests near the Current River in the same general area. Secondly, is the abundance of scarlet oak in the MOFEP study sites and its restriction to the Oak-Chestnut Region by Braun (1972) (she nowhere mentions it as occurring in Oak-Hickory forests). Scarlet oak also fails to appear in the species lists of Ware and others (1992). It might be speculated that, given the morphological similarity of scarlet oak to northern red and black oaks, variation in species identification might be responsible for these discrepancies.

Overstory-Understory Relationships

Figures 3-7 illustrate differences between overstory (>19.05 cm dbh) and understory (≤19.05 cm dbh) relative densities of major species of common ELTs . Some trends were apparent: (1) In every ELT examined, there was evidence of substantial depletion of understory populations of shortleaf pine and major red oak group species (scarlet and black oak); (2) In three of five major ELT classes (11, 17, 18, representing 92% of plots) white oak relative density in the understory was significantly greater than that in the overstory; (3) A number of other species that currently are not very important in the overstory show greater relative abundances in the understory [i.e., black gum, mockernut hickory, black hickory and sassafras (*Sassafras albidum* (Nutt.) Nees)].

Overstory-understory relationships varied somewhat with ELT classification. The understory relative densities of red oak group species and shortleaf pine tended to be greater in plots of ELTs that were more xeric in character (especially ELT 11 and 17, Figs. 4 and 5) compared with more mesic ELTs (5 and 18 Figs. 3 and 6). However, understory relative densities of these species were substantially reduced on ELT 19 plots, which also presumably was quite xeric. As mentioned above, relative density of understory white oak was greater than that in the overstory in three ELTs (11, 17 and 18, Fig. 4, 5 and 6). In the other two abundant ELTs (5 and 19) relative density of understory white oak was less than that in the overstory, but because there were few black or scarlet oak and shortleaf pine plants in the understory of forests of these ELTs, white oak was still the most common understory tree species. Other species that tended to show increased relative density in the understory (black gum, black and mockernut hickory, sassafras) showed little apparent relationship between relative abundance and ELT (and hence microenvironment). Post oak was relatively less abundant in the understory than in the overstory in all ELTs, but the differences were substantially less than those shown by red oak group species and shortleaf pine.

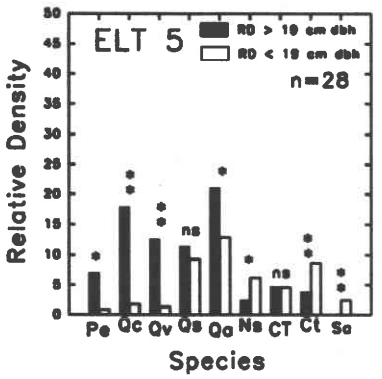

Figure 3. Overstory and understory relative density for major species on ELT 5 plots. Asterisks above each pair of histogram bars indicate highly significant (**p≤0.01) and significant (*p≤0.05) differences between overstory and understory pairs. Bar pairs with "ns" are not significantly different (p>0.05). Species codes: Pe-shortleaf pine, Qc-scarlet oak; Qv-black oak; Qs-post oak; Qa-white oak; Ns-black gum; CT-black hickory; Ct-mockernut hickory; Sa-sassafras..

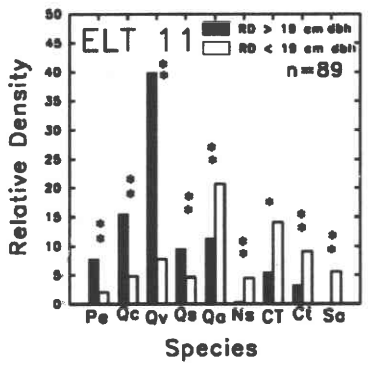

Figure 4. Overstory and understory relative density for major species on ELT 11 plots. Asterisks above each pair of histogram bars indicate highly significant (**p≤0.01) and significant (*p≤0.05) differences between overstory and understory pairs. Bar pairs with "ns" are not significantly different (p>0.05). Species codes: see legend for Fig. 3.

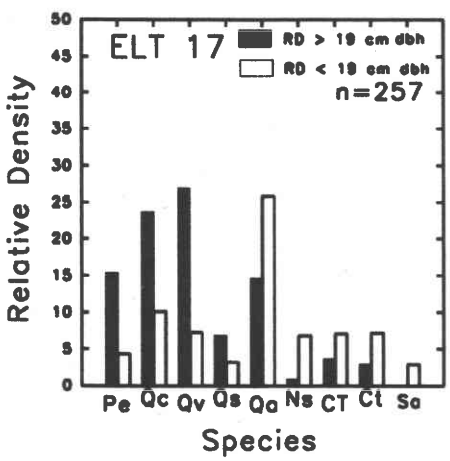

Figure 5. Overstory and understory relative density for major species on ELT 17 plots. All bar pairs are significantly different (p≤0.01). Species codes: see legend for Fig. 3.

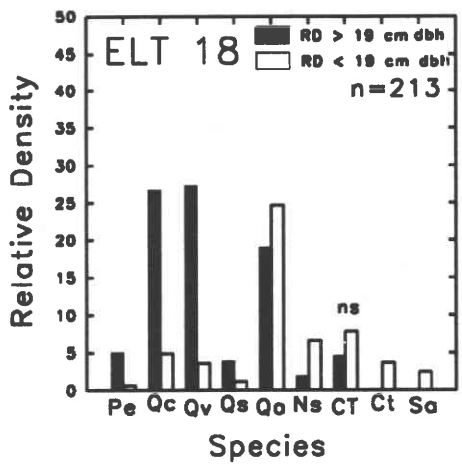

Figure 6. Overstory and understory relative density for major species on ELT 18 plots. All bar pairs except that marked with "ns" are significantly different (p≤0.01). Species codes: see legend for Fig. 3.

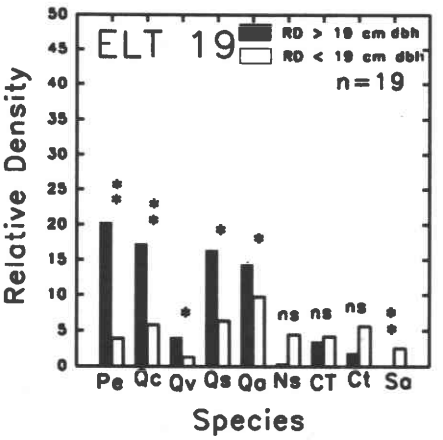

Figure 7. Overstory and understory relative density for major species on ELT 19 plots. Asterisks above each pair of histogram bars indicate highly significant (**$p \leq 0.01$) and significant (*$p \leq 0.05$) differences between overstory and understory pairs. Bar pairs with "ns" are not significantly different ($p > 0.05$). Species codes: see legend for Fig. 3.

Density-Size Class Distribution Relationships

χ^2-tests of distribution differences across size classes indicated that there were highly significant ($p \leq 0.01$) differences among species in every ELT (data not shown). Density vs. size class plots were constructed for major species and ELTs [see Fig. 8 for an example (ELT 17)]. In general, the results offered further confirmation of previously-noted patterns of species dominance and potential replacement across ELTs.

However, several new distribution patterns were shown in these plots and not elsewhere: (1) In all ELTs white oak abundance was greatest in the 1.5-4.5 in (3.8-11.4 cm) dbh size class, decreasing in all larger and the one smaller size class; (2) Most other abundant species in the understory (black gum, sassafras, black hickory, mockernut hickory) had greatest density in the smallest size class; (3) Some species that currently are abundant in the overstory (black, scarlet, and post oak and shortleaf pine) showed shallow negative slopes; (4) Density of red oak group species and shortleaf pine were much reduced in smaller size classes compared with those of white oak and other understory species, supporting the general trends and conclusions shown by overstory-understory analysis (see above). However, red oak group species are present in densities > 50/ha even in smaller size classes in certain ELTs, particularly those that are more xeric (e.g., 17, Fig. 8). Density of shortleaf pine is everywhere less than 25/ha; (5) The differences between ELT 17 and 18 in relative density of red oak group species and shortleaf pine vs. white oak were attributable to differences in absolute abundance in the former group. Absolute density of white oak in the understory was quite similar on both ELTs (data not shown; see also Brookshire and Hauser 1993).

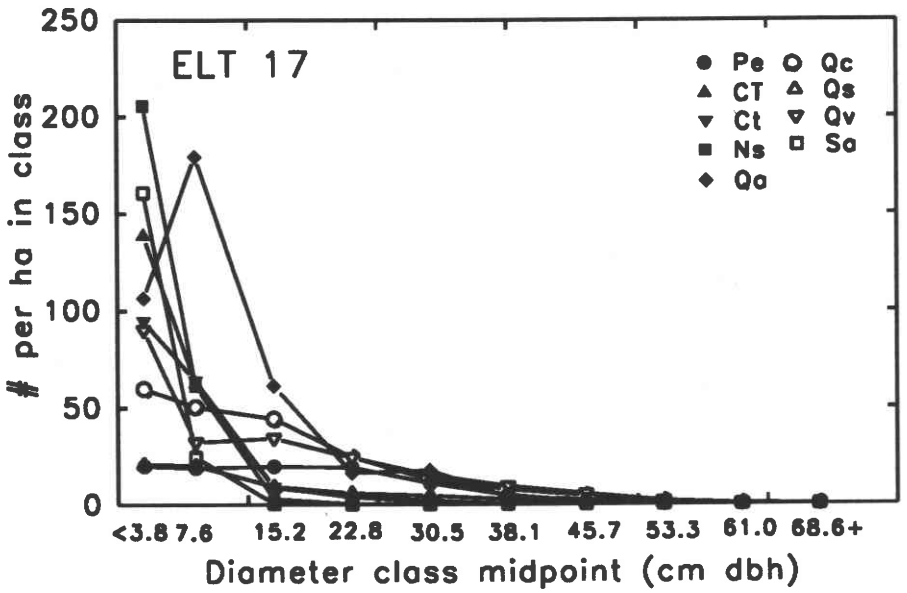

Figure 8. Mean density vs. size class relationships for important species on ELT 17 plots. Species codes: see legend for Fig. 3.

These data are consistent with trends we have seen elsewhere in the Ozarks (Jenkins and Pallardy, 1993). There is strong evidence in the MOFEP data set and from other research in the Missouri Ozarks that white oak has the potential to increase in importance throughout the region. The ecological basis of this potential change *may* be ascribable to white oak's greater longevity and shade tolerance (Dougherty and others 1980, Fowells 1965). The size class distributions further indicate that white oak apparently has experienced a "pulse" of recruitment, with the greatest abundance of this species found in the next-to-smallest size class. Other species, particularly black gum, some of the hickories, and sassafras show great abundance in the smallest size class. One might speculate that observers of long-term forest succession in the Ozark region might see these species becoming more important, but it is debatable whether they (except perhaps for the hickories) have canopy potential given the xeric environment of the Ozarks and the greater microclimatic stress associated with ascent to an unprotected canopy position (Nigh and others 1985, Pallardy and others 1988). On the other hand, the current position of white oak as a canopy dominant throughout much of the MOFEP study area indicates that its short-term increase in abundance is nearly certain.

Future trends in abundance of scarlet and black oak in MOFEP site forests may differ somewhat from those of shortleaf pine. In the former pair there is still sufficient regeneration to anticipate a continuing, if less prominent, contribution of both red oak species to future forest overstories. This is especially true on xeric ELT 11, 17, and 19 sites. In contrast, seedlings and saplings of shortleaf pine are nowhere abundant and one might predict a uniform decline in the abundance of this species through time in canopies of undisturbed forests of the MOFEP project.

ACKNOWLEDGMENTS

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TIMBER MARKING GUIDELINES TO MINIMIZE CHAINSAW FELLING ACCIDENTS

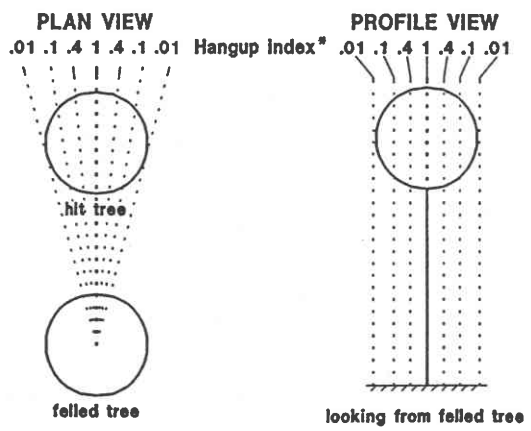
Penn A. Peters[1], Michael D. Erickson[2], and Curt D. Hassler[2]

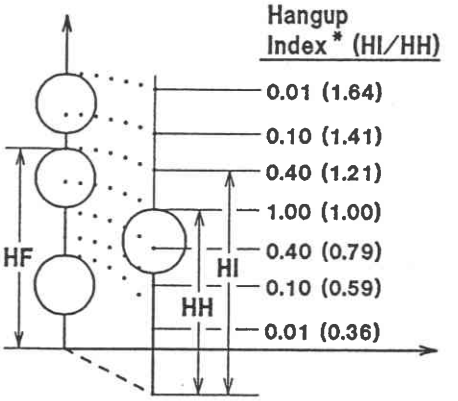
Abstract: Trees are marked by foresters for retention or harvest to satisfy landowner objectives for timber sale revenues, stand improvement, wildlife habitat, water quality, and aesthetics. Another consideration when marking trees is the safety of the chainsaw feller. Can the tree that has been marked for harvest be felled safely? Will the feller be able to select and control a direction of fall such that the felled tree does not produce a hazardous reaction?

When marking timber, the following will have a positive effect on chainsaw feller safety: 1. mark trees to be cut on two sides (for example, uphill-downhill), so the feller can easily see the trees to be cut and can make a safe selection of the felling order. 2. when selecting one cut tree from a pair of trees -- mark the tree that can be felled safely, 3. assume a heavily leaning tree will fall in the direction of lean and assess for hazards, and 4. learn the three factors that affect the likelihood of a hangup, viz., lateral offset, impact height (HI), and felled tree to hit tree dbh ratio. The lateral offset is the perpendicular distance from the direction of fall plane to the centerline of the hit tree. The impact height (HI) is the vertical distance from the base of the hit tree to the top of the falling tree as the falling tree intersects the lateral plane containing the hit tree (see figure). Felled tree height (HF) and hit tree height (HH) are defined in the figure also.

Effect of diameter

Hangup Index*	dbh of hit tree/ dbh of felled tree
0.9	2.07
0.7	1.29
0.5	.99
0.3	.80
0.1	.62
0.01	.48





*A tree with a hangup index of 0.4 is four times as likely to hang up as one with a hangup index of 0.1.

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MATING PARAMETER ESTIMATES OF BLACK WALNUT BASED ON NATURAL AND ARTIFICIAL POPULATIONS

George Rink[1], Guoqiang Zhang[2], Zuo Jinghua[2], and Fan H. Kung[2]

Abstract: Horizontal starch gel electrophoresis was performed on six polymorphic loci in black walnut (*Juglans nigra* L.) embryos from open-pollinated nut collections made in 1987 in a Missouri half-sib progeny test, and Indiana seed orchard and a natural population in southern Illinois. Allozyme data disclosed very high levels of variation (75.0-87.5%), mean heterozygosity (0.198-0.215) and outcrossing (0.880-0.928 in all three seed populations. Mating parameters from seed orchard populations were not significantly different from those of the natural population. The mean fixation index was less than 0.1 in all three seed populations, suggesting very low rates of inbreeding in this species in seed orchards as well as in a natural setting.

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VARIATION AMONG NORTHERN RED OAK PROVENANCES IN BARK THICKNESS:DBH RATIOS

Matthew S. Russell and Jeffrey O. Dawson[1]

Abstract: Differences in bark thickness in relation to diameter at breast height were observed in a 30-32 year old Illinois planting of 32 provenances of northern red oak (*Quercus rubra*) from throughout its natural range. Bark thickness by itself is often a good indication of relative cambial insulation from fire. Fire resistance in trees can largely be attributed to thickness and low thermal conductivity of bark. Western and southern provenances tended to have greater average bark thickness to dbh ratios than northern or eastern provenances because we tested latitude and longitude separately. One possible explanation is that provenances located in the western or southern areas of the range may have developed greater fire resistance through natural selection of genotypes that produce thicker bark at a given dbh.

INTRODUCTION

In mesic deciduous forests of Eastern North America, fires have historically been thought of as detrimental to the quality of standing timber. However, exclusion of fire from these forests has been implicated in the gradual post-settlement replacement of dominant early to mid-successional *Quercus* species with late-successional more shade tolerant *Acer* species (Wuenscher and Valiunas 1967; Abrams 1992). Pallardy and others (1991) noted significant changes in species composition over a 22 year period for three distinct forest types in Missouri. Increases in the prevalence of sugar maple (*Acer saccharum*) and decreases in *Quercus spp.* were attributed to exclusion of fire and succession towards more mesic environments. The continuing trend towards late successional stages has led many resource managers to reconsider fire as a management tool to manipulate natural processes of succession and to control forest species composition.

Northern red oak (*Quercus rubra*) is a commercially valuable species which is widely distributed throughout eastern North America and into southern Canada (Figure 1). The range of this species includes most states east of the Mississippi River except parts of states located in the coastal plains of the South and Florida. Northern red oak is intermediate in shade tolerance and can be found on a variety of topographical locations and soil types. The best development of this species occurs in the Ohio River Valley on deep, well drained loam to silty, clay loam soils (Sander 1990).

Very little is known about the effects of fire on central hardwoods. It is difficult to quantify the effects of fire due to confounding variables such as duration and intensity of burn. Several methods have been developed to delineate factors contributing to fire tolerance of various tree species (Kayll 1963; Vines 1968; Hare 1964; Gill and Ashton 1968; Uhl and Kauffman 1990). There is a consensus among researchers that the vascular cambium of trees will be killed at temperatures in excess of 60°C (Hare 1961). Temperatures at the surface of tree stems often exceed 60°C in prescribed fires (Fahnestock and Hare 1964). Knowledge of the specific effects of fire on individual trees is necessary if fire is to be used efficiently in management. A recent study by Hengst and Dawson (1994) has shown that greater bark thickness and lower thermal conductivity moderate hardwoods' cambial temperature increases resulting from artificial burning. In the same study northern red oak from central Illinois had one of the lower bark thickness:dbh ratios of the common native oak species.

Bark has long been recognized as a factor which provides fire resistance to trees (Starker 1934; Hare 1965; Martin 1963; Vines 1968). Numerous air-filled cells in bark provide excellent insulation and prevent rapid fluctuation of

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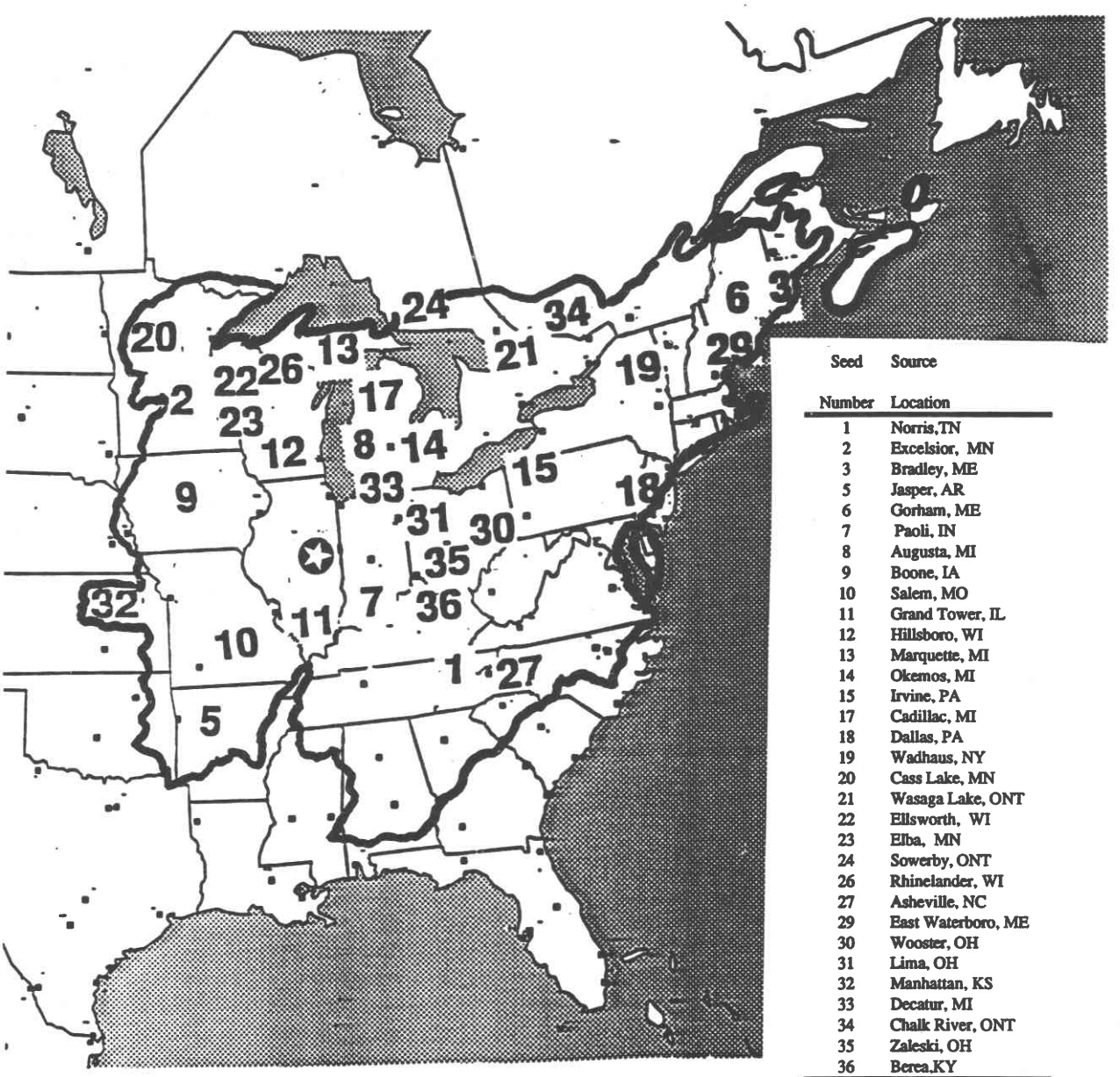


Seed Number	Source Location
1	Norris,TN
2	Excelsior, MN
3	Bradley, ME
5	Jasper, AR
6	Gorham, ME
7	Paoli, IN
8	Augusta, MI
9	Boone, IA
10	Salem, MO
11	Grand Tower, IL
12	Hillsboro, WI
13	Marquette, MI
14	Okemos, MI
15	Irvine, PA
17	Cadillac, MI
18	Dallas, PA
19	Wadhaus, NY
20	Cass Lake, MN
21	Wasaga Lake, ONT
22	Ellsworth, WI
23	Elba, MN
24	Sowerby, ONT
26	Rhinelander, WI
27	Asheville, NC
29	East Waterboro, ME
30	Wooster, OH
31	Lima, OH
32	Manhattan, KS
33	Decatur, MI
34	Chalk River, ONT
35	Zaleski, OH
36	Berea,KY

= Location of provenance study near Monticello, Illinois.

Figure 1. The native range of *Quercus rubra* and locations of 32 seed sources used in an East Central Illinois provenance study.

temperatures at the vascular cambium. Martin (1963) conducted a laboratory experiment examining the thermal conductivity of the bark of several species. His results demonstrated the good insulating qualities of bark and identified some factors contributing to variation in heat conductance between and within the bark of different species.

Northern red oak has been observed to exhibit considerable geographic genetic variation for a variety of traits in past observations of provenance trials (Kriebel and others 1976). McMahon (1988) conducted a study examining the performance of different provenances of Northern red oak in the same plots used in this study. He observed variation and significant differences among provenances for a variety of characteristics such as survival, growth rate, forking, and phenology. The purpose of our study was to determine whether northern red oak provenances differ in bark thickness:dbh ratios.

METHODS

Study Design

Variation in bark thickness was determined in the summer of 1993 in a northern red oak provenance study near Monticello, Illinois (40.1° N latitude; 88.5° W longitude). The planting was one of several established as the U.S.D.A. North Central Cooperative Regional Project (NC-51) entitled "Tree Improvement through Selection and Breeding of Trees of Known Origin." The 1-0 stock was planted during spring of the years 1962-1964. Individual tree seed sources were not separated into families for this study. Bulked seedlots were the source of planting stock. The experimental design encompassed 32 seed sources within 7 replicates of a randomized complete block experimental design. Each replicate contained at least 16 trees from each source upon establishment with a spacing of 3.0 x 2.4m. A double border row of common seed source trees surrounded the plots.

Six trees were measured from 4 x 4 plots in each of 3 replications of each source. Bark thickness measurements were made to the nearest millimeter using a metric bark gauge. Diameter at breast height was measured to the nearest millimeter 1.4m above the ground. Trees that were measured were randomly selected from the total number of surviving trees present. The trees were consecutively assigned a number. and a random number table was then used to select 6 trees. Two bark thickness measurements were made on the north and south sides of each randomly selected tree at a height of 1.4m.

Data Analysis

In a previous study, bark thickness of northern red oak and 15 other hardwood species from a single locale was positively linearly correlated with dbh for each species (Hengst and Dawson 1994). The ratio of bark thickness to dbh was used in analysis rather than bark thickness alone to eliminate the confounding influence that tree size has on bark thickness. The rationale for using the ratio of bark thickness to dbh as opposed to some other function of the two variables is geometrically motivated and is supported by a simple linear regression (Figure 2). Here we see that apart from some residual variance, attributable to seed sources, bark thickness is essentially a multiple of dbh offset by a small intercept very near the origin. Also, in order to avoid the consequences that outliers and violations of the usual assumptions of the standard linear regression model may have on statistical inference, a nonparametric index was used to measure the association of the variables of interest. The mean bark thickness:dbh ratio was associated with latitude and longitude using this ordinal method. Specifically Kendall's tau, a nonparametric index of ordinal association that can be interpreted much like the usual correlation coefficient, was used along with its corresponding test of statistical significance. It should be noted that the usual adjustment to Kendall's tau for data containing ties was used, because there were some ties in the bark thickness:dbh ratio. Though standard linear regression was not used for statistical inference, it was used to provide graphical summaries of the data. Statistical analyses were conducted using S+ (Statistical Science Inc. 1994) and SAS version 6.0 (SAS Institute Inc. 1989). Variation among bark thickness:dbh ratios was also examined for all provenances using ANOVA. The ANOVA was followed by a t-test to examine geographic variation among disjunct seed sources (Table 1).

Table 1. Provenance means for bark thickness:dbh ratios for a 30-32 year old range wide provenance trial of *Quercus rubra* grown in East Central Illinois. A T-grouping was constructed to illustrate statistical differences among seed sources by mean bark thickness:dbh ratios ranked from greatest to least. Means followed by the same letter are not significantly different (α =.05).

Seed Number	Source Location	Mean bark thickness: dbh ratio	# of observations	T-grouping
1	Norris,TN	0.047	17	A
36	Berea,KY	0.045	18	A
2	Excelsior, MN	0.045	12	A
10	Salem, MO	0.045	18	AB
35	Zaleski, OH	0.044	17	ABC
34	Chalk River, ONT	0.042	18	BCD
5	Jasper, AR	0.041	16	CD
9	Boone, IA	0.041	18	CD
23	Elba, MN	0.041	18	CD
27	Asheville, NC	0.040	14	D
12	Hillsboro, WI	0.040	18	D
8	Augusta, MI	0.040	18	D
11	Grand Tower, IL	0.040	18	DE
7	Paoli, IN	0.040	18	DE
32	Manhattan, KS	0.039	34	DEF
6	Gorham, ME	0.039	18	DEFG
15	Irvine, PA	0.036	13	EFGH
33	Decatur, MI	0.036	18	FGH
31	Lima, OH	0.036	18	FGHI
30	Wooster, OH	0.036	13	FGHI
22	Ellsworth, WI	0.035	18	GHIJ
20	Cass Lake, MN	0.035	18	HIJ
21	Wasaga Lake, ONT	0.035	18	HIJ
17	Cadillac, MI	0.035	18	HIJ
29	East Waterboro, ME	0.035	18	HIJ
18	Dallas, PA	0.034	18	HIJK
26	Rhinelander, WI	0.034	18	HIJK
19	Wadhaus, NY	0.033	18	HIJK
14	Okemos, MI	0.033	14	IJK
13	Marquette, MI	0.032	18	JKL
3	Bradley, ME	0.032	18	KL
24	Sowerby, ONT	0.030	12	L

RESULTS

Among the 32 seed sources in the provenance study, the Kendall's tau index of correlation between latitude and the mean provenance ratio of bark thickness to dbh was equal to -0.438, revealing that a negative association exists between bark thickness:dbh ratio and latitude. In other words, trees with seed sources in the south tend to have higher relative bark thickness for a given dbh, a finding consistent with our hypothesis. A central limit theorum based test of the statistical significance of the observed value of Kendall's tau had an association probability value of 0.0002. The mean values of the ratio of bark thickness:dbh are plotted against latitude (Figure 3) along with the least squares regression line. The negative slope of the regression line illustrates the trend.

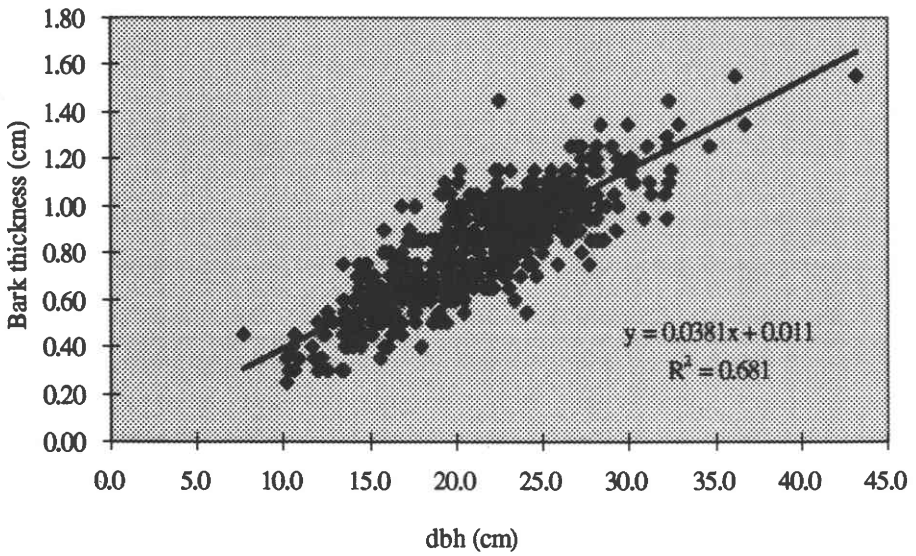


Figure 2. Dbh vs. barkthickness for 32 seed sources planted in a central Illinois provenance trial of Quercus rubra. Individual tree data were used to plot the regression (n=589).

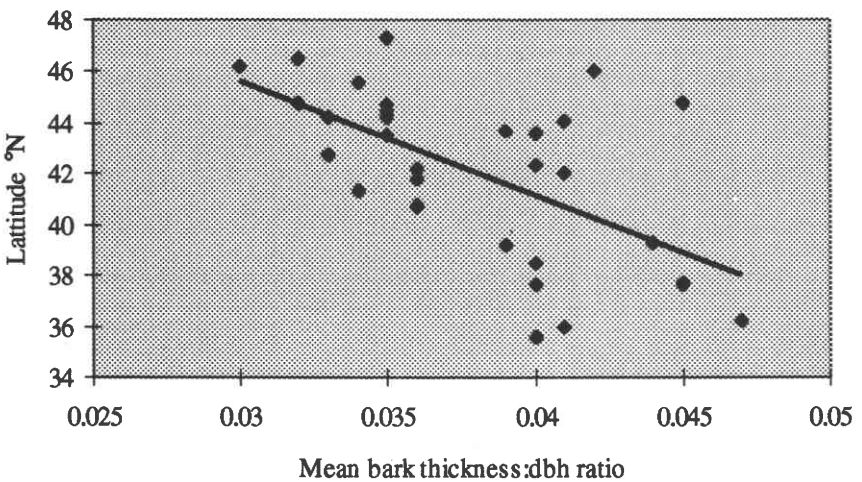


Figure 3. Mean provenance bark thickness:dbh ratios by seed source vs. latitude.

Kendall's tau was also computed to examine the association between longitude and bark thickness:dbh ratio, and was equal to 0.304. While the degree of association implied by this index suggests that latitude is more closely associated with bark thikckness:dbh than is longitude, the positive value of Kendall's tau supports our hypothesis that western seed sources tend to produce trees with relatively thicker bark for a given dbh. A test of significance in this case resulted in a probability value of 0.007, once again lending support to our assertion. The mean values of the ratio of bark thickness:dbh are plotted against longitude (Figure 4) along with the least squares regression line. The positive slope of the regression line illustrates the trend.

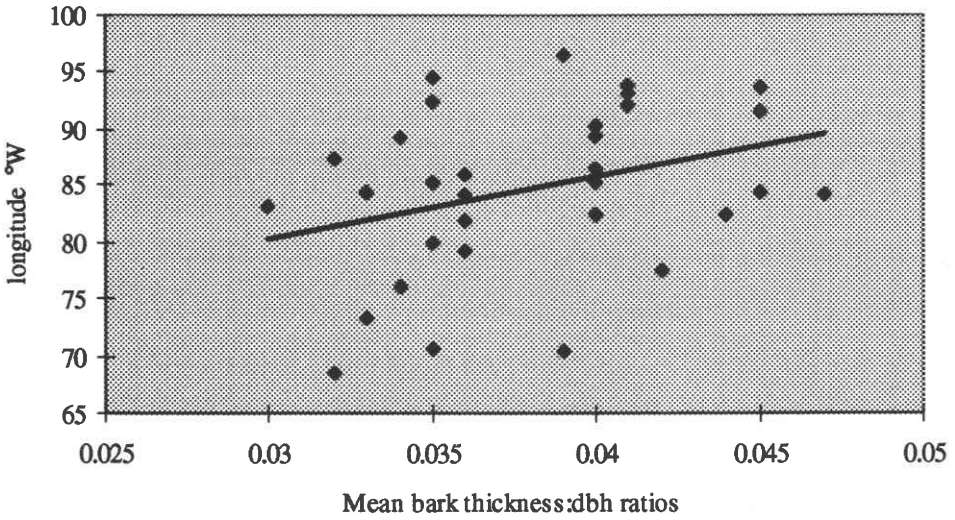


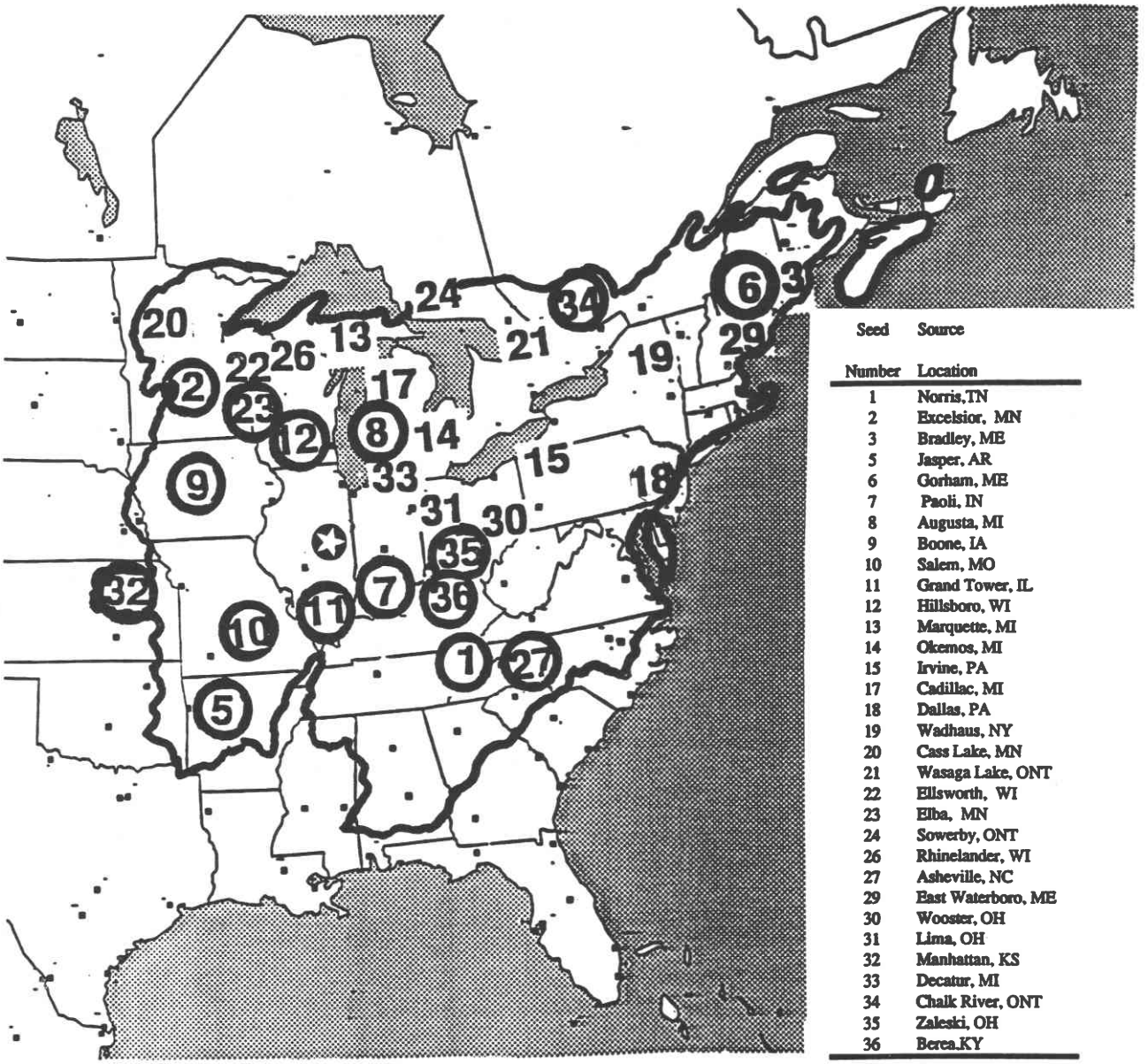
Figure 4. Mean bark thickness:dbh ratios by seed source vs longitude.

Analysis of variance indicated that different mean values for bark thickness:dbh ratios for provenances of northern red oak were significantly different ($\alpha = 0.05$) A T-grouping illustrates specific statistically significant differences in bark thickness:dbh ratios among provenance means (Table 1).

DISCUSSION

The median value of bark thickness:dbh ratios (0.038) for all provenances was used as a division point to separate the two size classes in figure 5. Results indicate that provenances associated with western and southern areas of the range generally had greater bark thickness for a given dbh according to Kendall's tau. Exceptions to the trend are source 6 from Gorham, Maine and source 34 from Chalk River, Ontario. We do not have explanations for these exceptions, though there is a possibility that fire frequented these areas prior to European settlement.

Fire was probably most prevalent in western and southern portions of the range of northern red oak in the millennia following the retreat of continental glaciers from the northern part of its range 12,000 years ago (Abrams 1992).. In these areas long term exposure to increased fire frequency may have led to the natural selection of tree genotypes with greater bark thickness:dbh ratios which consequently could better resist cambial damage from fire. Other factors such as long term patterns in conditions conducive to the prevalence of bark boring insects also may have influenced ecotypic variation in bark characteristics of northern red oak.



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= Location of provenance study near Monticello, Illinois.

Figure 5. Contrast between bark thickness:dbh ratios in the western and southern portions with those in the northern and eastern portions of the native range of *Quercus rubra*. Seed source ratios above the median value of 0.038 are enclosed in a circle.

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PRELIMINARY GUIDELINES FOR THE USE OF TREE SHELTERS TO REGENERATE NORTHERN RED OAK AND OTHER HARDWOOD SPECIES ON GOOD TO EXCELLENT GROWING SITES

Thomas M. Schuler[1], Gary W. Miller[1], and H. Clay Smith[2]

Abstract: Regenerating northern red oak (*Quercus rubra* L.) on good to excellent growing sites is a serious problem for forest managers throughout the eastern and central United States. Natural regeneration using shelterwood methods are still being refined and are dependent on existing natural seedlings, an uncommon situation in many of today's forests. A possible alternative to natural regeneration is the use of plastic tree shelters in conjunction with a planted seedling. Research has shown that tree shelters promote a period of rapid seedling height growth followed by a period of much slower growth after emerging from the top of the shelter. Because the use of tree shelters represents a significant financial investment, judicious use of this technique is warranted. Preliminary guidelines are presented to maximize the likelihood of early seedling survival, growth, and potential competitiveness in the regenerated stand. Recommendations are based on research in the use of tree shelters conducted since 1988 on the Fernow Experimental Forest in north central West Virginia. Considerations include the age and origin of planting stock; size, color, style, and height of the tree shelter; stake material; planting location, density, and timing in relation to overstory removal; the use of herbicides or mulch to retard competing vegetation; maintenance schedules; and costs for installation.

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THE EFFECT OF SITE DISTURBANCE ON NITROGEN AND PHOSPHORUS AVAILABILITY IN INDIANA OAK-HICKORY FORESTS

D. Andrew Scott, Phillip E. Pope, Donald J. Kaczmarek, and Karyn S. Rodkey[1]

Abstract: Nitrogen (N) and phosphorus (P) are nutrients often limiting forest site productivity. Soil water availability, soil temperature, and litter substrate quality interact to control the availability of these mineral nutrients and these factors can be directly altered by site disturbance. This study will examine the effects of site disturbance at three oak-hickory dominated forests of varying productivity. The study will utilize both field and controlled environment experiments. The objectives of the study are: 1) to determine the effects of disturbance on N and P availability, 2) to determine how sites of varying productivity respond to site disturbance, and 3) to determine the effects of varying soil water availability on N and P mineralization rates. The disturbances created are: (1) removal of litter and forest floor organic layers to the mineral soil, (2) incorporation of the forest floor with the mineral soil to a depth of 20 cm, (3) ground fire, i.e. burning the forest floor, and (4) an undisturbed control N and P availability will be determined using mixed bed anion/cation ion exchange resins. In the field, resins will be buried in the mineral soil at a depth of 20 cm. In the controlled environment study, intact soil monoliths measuring 15 cm in diameter and 30 cm in length will be maintained at either high or low soil water potentials to determine the effects of water availability on N and P mineralization rates as determined by ion exchange resins. Temporal changes in ammonium (NH_4^+), nitrate (NO_3^-), and phosphate (PO_4^{2-}) availability in response to these site disturbance treatments will be determined.

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SALAMANDER ABUNDANCE IN SMALL CLEARCUTS

Dana A. Soehn and Dr. Edwin D. Michael[1]

Abstract: Recent research has shown that some timber harvesting methods adversely impact salamander populations. Researchers have documented the virtual disappearance of salamanders in large clearcuts with population recovery times estimated from 20-60 years. The purpose of this study was to determine salamander abundance in small, 3-year old clearcuts at Cooper's Rock State Forest in northern West Virginia. Salamanders were sampled inside and outside 16 clearcuts (0.5, 1.0, 1.5, and 2.0 ha), using a nondestructive monitoring technique. A total of 152 sampling stations were established, each consisting of 12 boards (12" x 6" x 1") placed in a 3 x 4 configuration with 0.5 inches separating individual boards. From April through November of 1994, all boards were lifted and checked for salamanders every 2 weeks. Captured salamanders were given a unique toe clip, sexed, measured, and released. Individual recognition of salamanders allowed for monitoring of movements throughout the season and also a population estimate using a multiple mark-recapture technique. Microhabitat features, including soil temperature, surface temperature, soil moisture, litter moisture, litter mass, and soil pH, were measured at each station. Preliminary results indicate that salamanders are present in the three year old clearcuts, but estimated abundance is less than in the adjacent mature forest. Microhabitat conditions in these small patch cuts may more rapidly allow for recolonization of salamander populations to predisturbance levels, than would occur in larger clearcuts. A total of 742 individual salamanders were captured, with the redback salamander (*Plethodon cinereus*) being most abundant (85% of total captures).

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PRODUCTION AND TRADE FLOWS OF MICHIGAN FOREST PRODUCTS

Dr. James Stevens[1]

Abstract: Michigan hardwoods are an important contributor to the Michigan State economy yet the flow of wood and wood products is poorly documented. This project is an on-going survey of primary and secondary wood processing firms in Michigan. The objective is to determine wood flow from stump to point of final processing (if in Michigan) or to the point where it leaves the state with the intent of evaluating the potential for increasing value-added production in the state. Preliminary data indicate that the flow of raw materials and products is very dynamic. As timber prices have risen in the past few years, firms have tried to capture a larger profit margin by eliminating the use of brokers. The final report will be complete by the end of 1994 and the results will be available for the March conference.

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ECOSYSTEMS MANAGEMENT RESEARCH IN HARDWOOD FORESTS DOMINATED BY DEER

Susan L. Stout, David S. deCalesta, Stephen B. Horsley, Christopher A. Nowak, and James C. Redding[1]

Abstract: Forest ecosystems of the Allegheny Plateau region of Pennsylvania have become impoverished as a result of over-browsing by white-tailed deer for more than 50 years. Research conducted in this region shows a strong, negative relationship between deer density and diversity of woody and herbaceous plant species and intermediate canopy nesting songbirds. This research also shows that deer are strongly associated with difficulties in establishing diverse regeneration of commercially important species. Research to date has been conducted on the stand level, and this research suggests that manipulation of deer forage, through managing at the landscape level, offers some hope of reestablishing diversity in forest ecosystems. Scientists have designed a landscape-level study to test this hypothesis generated from stand-level research. A study design combining observations and experimental manipulation on blocks of 500-1000 acres of forest land has been selected, and initial measurements will be collected before the Conference. While Pennsylvania's forests have suffered over-browsing for more than 50 years, this research has wide regional importance, as many other states are approaching the conditions currently found in Pennsylvania, and mitigation strategies are needed throughout the region.

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Two invited papers, 57 volunteer papers, and 22 volunteer poster summaries presented at the 10th Central Hardwood Forest Conference. Presentation topics included economics, forest amenities, harvesting, logging safety, utilization, physiology, genetics, ecology, regeneration, silviculture, protection, management, hydrology, soils, nutrient cycling, and hardwood markets of central hardwood forests.

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