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Proceedings: Workshop on Fire, People, and the Central Hardwoods Landscape

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Compiled by:
Daniel A. Yaussy

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Preface

This conference was conceived as a vehicle to demonstrate to forest land managers and the interested public, the historical role fire has played as an agent of change and renewal in forests of the area known as the Central Hardwoods Region. However, fire is widely perceived as being potentially dangerous and damaging to trees and ecosystems. This conference, held in Richmond, Kentucky, March 12-14, 2000, explored the historical, present, and future roles of fire and humans in shaping the vast and diverse deciduous forest of the Central Hardwoods Region.

This is a compilation of the 18 papers and 16 poster abstracts presented in the one and a half days of the conference. The workshop was followed by a field trip to Cliff Palace Pond where the pollen and charcoal record evidence of the relationships of fire, people and the forest that existed more than 9500 years ago, challenging our view of the “virgin” forest primeval. A few speakers chose not to submit a final paper to this compilation and the abstract of their presentation is included here.

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Fire In The Central Hardwoods: Why Are We Concerned?

Rex B. Mann[1]

Abstract

For several decades, land managers in the Central Hardwood Region have watched their oak-dominated forests change in the absence of fire. Stand structure has changed as a dense understory of shade-tolerant, fire-intolerant species developed. The forests were not sustaining themselves.

This same phenomenon was occurring in many parts of the United States, resulting in forest health concerns. Severe wildfires erupted in places like Yellowstone National Park and Florida.

Evidence is accumulating that the oak, chestnut, and yellow pine plant communities in the Central Hardwood Region are the result of centuries of burning by Native Americans. To maintain the diversity of our forests, fire must be restored to its vital role.

There are a number of myths relating to the forests that are widely held by the public that present obstacles to reintroducing fire. These myths must be dispelled if we are to restore fire to its rightful role and thus sustain our forests.

Introduction

For years, land managers throughout the Central Hardwood Region have noted definite changes occurring throughout the historically oak-dominated forests under their care. The oaks, which are valued because of their usefulness as wildlife food sources and for a variety of wood products, are being replaced with maple, hemlock, white pine, and other species. Oak seedlings are simply not surviving in the dense understory shade beneath the oak stands. In the absence of fire, the maple, hemlock, white pine, and other species have come to dominate the understories and are posed to capture the overstory.

Indeed, the only way we could regenerate the oak forest was to harvest it and manage the coppice growth. About the only place acorns were germinating and growing to mighty oak trees was at the edge of the forest, where sunlight was abundant. In addition, an ever-increasing number of plants and animals were being listed as endangered or threatened. Something was missing here. Yet, in view of the amazing recovery our Eastern forests made from the massive timbering that occurred during the early 1900's, we were not terribly concerned until fairly recently.

Throughout many other parts of the country, similar challenges have emerged—challenges to restore fire-dependent forest ecosystems that are being replaced in the absence of fire. These conditions are a result of more than 75 years of fire suppression and in some areas have been labeled as forest health problems.

For those of us who serve as firefighters, we have had numerous opportunities to witness the day of reckoning that eventually comes to fire-dependent ecosystems when historic fire is suppressed for long periods of time. Yellowstone in 1988, and more recently the Florida fires of 1998, were classic examples of forest succession in the absence of "normal" fire. Fuel loads and more volatile undergrowth in some areas of the country have increased in the absence of fire to a point where even carefully applied, controlled burns cannot be ignited without disastrous results. Indeed, there will be plenty of gainful employment for wildland firefighters as these past-due bills continue to come due in the years and decades ahead.

Closer to home, in the Central Hardwood Region, evidence is accumulating that our forests are also fire-dependent communities. Consensus is emerging that Native American's fire was responsible for the dominance of the oaks, American chestnut, and yellow pine plant communities we have enjoyed in recent history. Even though wildfire hazards in this region are not nearly as severe as those in the coastal plains or in the West, the exclusion of fire has greatly altered the future of oak and pine-dominated ecosystems. The oak and pine forests are not sustaining themselves in the absence of frequent surface fires except where land managers have interjected silvicultural manipulation.

As we continue to learn more about the ecosystems that make up the Central Hardwood Region, and as we continue to sort out the relationships between individual species of plants and animals and their ecosystems, there appears to be a common thread linking many of these problems we face—the exclusion of frequent surface fires.

As we learn more about these forests and how they came to be, we are dispelling more and more the myths that we as a people have come to believe. For example,

- There is the myth of the pristine wilderness that existed at the time of European settlement, where a squirrel could theoretically travel through the unbroken treetops of an old-growth climax forest from the Atlantic coast to the Mississippi River.

- There is the myth that Native Americans were few in number and had little impact on the land.

- And there is the myth that the forest was very stable, that disturbances were small in scope and natural in origin, and that a great balance existed.

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[1] Timber/Wildlife/Fire Staff Officer, USDA Forest Service, Daniel Boone National Forest, 1700 Bypass Road, Winchester, KY 40391

Now we know that the Central hardwood landscape at the time of European settlement was very diverse with many treeless openings. The pristine wilderness and unbroken canopies existed only in the myth. It is also generally agreed that Native Americans were far greater in numbers than we have been led to believe, until European diseases decimated them. We also know that the tremendous landscape diversity we inherited was a result of their land use, primarily through their use of fire. And finally, we now know that the great “balance of nature” did not exist, that change was the norm rather than stability.

As the land-managing agencies and land owners enter this new millennium, we are challenged to look into and understand the past in order to be able to craft strategies for the future.

Hopefully, during the next two days:

1. We can continue to dispel myths. Myths, which can be the greatest enemy of science; and

2. We can increase our knowledge and understanding of the ROLE OF FIRE IN THE CENTRAL HARDWOODS.

Humans And Fire: A History Of The Central Hardwoods[1]

Ray R. Hicks, Jr.[2]

Abstract

The Central Hardwood Forest is an oak-dominated deciduous forest that stretches from Massachusetts to Arkansas and occurs in hilly to mountainous terrain. It is the largest and most extensive temperate deciduous forest in the world.

During the past 20 million years or so, angiosperms have been gradually replacing gymnosperms as the dominant plant form on earth, and deciduous hardwoods are particularly adapted to the fluctuating seasonal climate and moderate rainfall associated with the Central Hardwood Region. As the glacial ice sheets retreated about 12-14 thousand years ago, forests re-invaded the region. Native peoples populated the area and practiced extensive burning to improve habitat for game and to aid in land clearing for agriculture. By the time the early Europeans arrived, about 450 years ago, the Native People had adopted a sedentary lifestyle and their population had increased to levels comparable to that of Western Europe. The first European explorers brought with them diseases that decimated the native populations, and for almost 200 years the central hardwood forest re-grew to become the "primeval forest" of legend.

By the late 18th century, settlement by subsistence farmers using European methods (draft animals and metal tools) was in full swing. Much of the tillable land in the central hardwood region was cleared, including substantial areas of steep and hilly land. The industrial revolution in the post-Civil War period ushered in a trend away from subsistence farming and marginal land was abandoned to revert back to forest.

The last 100 years has seen an era of exploitive logging in the central hardwood region, followed by a re-growing forest. Human influences have predominated in shaping the present forest and these include logging, burning, grazing, fire control, wildlife management and pest introductions. In the past 30 years, societal attitudes toward land and forests has had a profound effect on governmental policy as it relates to forestry. The maturing central hardwood forest, mostly owned by private individuals, is rapidly becoming a "resource at the crossroads".

Introduction

The historical developments that shaped the central hardwood region are significant in understanding how the present forests came to be and, as always, knowledge of history enables us to learn from past successes and failures. Thus, history provides not only the background, but also the context for future management. For purposes of this discussion, the history of the central hardwoods will focus on the period after the last ice age (the period of human impact). Because the particular forest that exists today is mostly a result of events that have occurred during the life of the trees that presently comprise it, this period will be emphasized.

The overriding factors that predispose a given geographic region to a particular vegetation type are the climate and soil (site), and this has been true throughout the hundreds of millions of years that vegetation has existed on earth. Furthermore the prevailing vegetation creates habitat for animal forms which are adapted to it, and they, in turn, have a biofeedback relationship with the vegetation. This type of association is the basis for what is called an "ecosystem" (Odum 1971).

Evolution and the Deciduous Forest

The ability of plants to capture solar energy by using readily available elements (carbon, hydrogen, oxygen, etc.) has been the key to all biological development on earth. Fossil evidence is our window to the past, and although incomplete, it appears that this process of photosynthesis first developed in primitive marine algae between 400 and 500 million years ago (Eyre 1963). From these evolved the primitive land plants that occupied the earth's surface prior to the Carboniferous Period (more than 200 million years ago). During the Carboniferous Period (215-280 million years ago), ferns and their relatives dominated the earth. Around 190 million years ago, the gymnosperms began to take over as the principal vegetation, and they have given way to the dominance of angiosperms (including deciduous hardwoods) during the last 50 or so million years. The angiosperms, though relatively recent in their development, have shown a remarkable ability to adapt to the changing conditions of the earth. From this point of view, it might be said that the central hardwoods represent the cutting edge of evolutionary change, as compared to the boreal conifer forest.

Perhaps the most recent (and most significant) episode of climatic and vegetational change to affect the distribution of present-day vegetation was the so-called ice age. The continental glaciers extended down through North America into southern Illinois, and a zone of tundra extended well down into much of the present central hardwood region (Davis 1983). As the Wisconsin ice sheet began to retreat northward about 14,000 years ago, a boreal forest type replaced the tundra and vestiges of this forest still exist as the high-elevation spruce and spruce-fir communities in the Appalachians (Maxwell and Davis 1972). The deciduous forest community has gradually become the dominant vegetation of the region and now forms the most extensive forest of this type in the world.

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[1] This paper was previously published in the Proceedings of the 11th Central Hardwood Forest Conference, Columbia, MO. 1997.

[2] Professor of Forestry, West Virginia University, Morgantown, WV 26506-6125.

Impact of the Native People on the Pre-columbian Forest

As important as the ice age , and its retreat, has been, the impact of humans has probably been just as significant in shaping the central hardwood forest. The indigenous people occupied the central hardwood region for at least 10,000 years before Europeans came (Lesser 1993). In the southern portion of the region, human occupancy probably dates back as long as 14,000 years ago owing to the more hospitable climate farther south (Buxton and Crutchfield 1985). Archaeologists classify the human occupancy of North America into Late Prehistoric, Mississippian, Woodland, Archaic and Paleo periods (Lesser 1993). Table 1 shows the chronology of these periods and outlines their cultural lifestyle as well as broad climatic and vegetational environments (Buckner 1992). Prior to the Woodland period, indigenous peoples were nomadic hunter-gatherers. Their influence on the forest during this period was most likely through the setting of fires. Accidental human-caused fires no doubt occurred during this time, and perhaps intentional fires were set to drive out game or to retain habitat. MacCleery (1992) speculates that the early Europeans' observations of abundant game (bison, elk, etc.) in the central hardwood region indicates a greater proportion of grassland occurred than currently exists, which implies that burning probably took place. Such burning, most likely would have been the work of the indigenous population.

Around 1,000-500 B.C., the Native Americans adopted a more sedentary lifestyle, and at some point during that time, they began the practice of agriculture (Swanton 1979). They also constructed elaborate burial mounds which can be found in places, such as Moundsville, West Virginia, Marietta and Hillsboro, Ohio, East St. Louis, Illinois, and Cartersville, Georgia. During this agricultural phase, which extended into the Late Prehistoric Period until European contact (about A.D. 1600), the Native Americans generally lived in permanent villages, often on the level land of river floodplains (Davis 1978). The village would often have a structural layout with houses of thatch or pole construction surrounded by gardens and fields. During this time, the Native Americans used tools as well as fire to clear land. MacCleery (1992) states the case as follows: "The south was dominated by fire-created forests, such as long-leaf pine savannas on the Coastal Plain and Piedmont. The hardwood forests of the Appalachian Mountains were also burned frequently by native peoples. Virginia's Shenandoah Valley — the area between the Blue Ridge Mountains and the Alleghenies — was one vast grass prairie. Native peoples burned the area annually." MacCleery further states: "On the western fringe of the eastern forest, fire-dominated forests, such as oak and oak-pine savannas, covered tens of millions of acres." Martin and Houf (1993) indicated that remnants of so-called "balds" still exist in the Ozarks of southwestern Missouri. These grassy glades usually occur on dolomitic limestone soils, and require periodic fires for their maintenance. They tolerate intermittent grazing, as would have occurred when migratory herds of bison occupied the area, but under year-round grazing imposed by the European settlers, eastern redcedar began to overtake the glades. Beilman and Brenner (1951) use several lines of evidence to indicate that extensive forest cover is a relatively recent development in the Ozark region. They refer to Houck's (1908) account of Ferdinand DeSoto's observations of the region in 1541 where he described fertile alluvial bottoms planted in maize, pecans, plums and mulberry trees. The open, park-like countryside was in contrast to the hardwood and pine forests that exist today. Another description of park-like vegetation in the Ozark region was provided by the explorer Coronado, where he noted the hunting of buffalo by the Osage Indians. Although substantial prairie-like areas probably occurred throughout the central hardwood region, Steyermark (1959) indicates that even within the Ozark region, where grasslands were common, substantial areas of hardwood forests existed prior to European settlement.

Because the indigenous people tended to settle on floodplains of major rivers, their influence was probably greatest in such areas. Therefore, in many of the regions that are dominated by uplands where rivers have narrow floodplains, such as interior West Virginia, western Kentucky, southeastern Virginia and into the Cumberland Plateau of Tennessee, the evidence of Native American influence is more limited. However, their use of fire may have enabled them to exert an influence far removed from their settlements. Cronon (1983), with reference to the American Indian influence on the landscape states: "It is tempting to believe that when the Europeans arrived in the New World they confronted Virgin Land, the Forest Primeval, a wilderness which had existed for eons uninfluenced by human hands. Nothing could be farther from the truth.— Indians had lived on the continent for thousands of years, and had to a significant extent modified its environment to their purposes." The population density of Native Americans is still being debated, with estimates ranging from 1 million to more recent higher estimates, up to 18 million (Buckner 1992). At higher levels, their impact on the forest would have been greater, but in any event, they most certainly had a significant effect on the central hardwood region.

Post European Settlement to 1860

The first Europeans in North America, apart from settlers in colonial villages along the Atlantic seaboard, were essentially hunter/trapper/explorers. They had a limited direct impact on the forest, but their indirect impact was profound. They brought very little with them in the way of equipment, livestock or supplies, but they did bring something that proved to be more significant than any of these—disease. As Williams (1989) points out, the epidemic of disease introduced by the Europeans affected the native peoples to such a degree that between 1520 and 1700, there was a return to a more forested landscape in North America (Buckner 1992). It was this forest that most historians refer to as the "impenetrable ancient forests." Early explorers kept few records of their observations, but they opened the way for others including botanists, surveyors, etc. Britain offered huge grants of land to individuals, such as Lord Fairfax and William Penn, to encourage settlement of the new world and to gain control of the land from the native population. By

Table 1—Chronology of Native People from Buckner (1992)

Cultural Event	Date	Years Ago	Climatic/Vegetational Stages
HISTORIC PERIOD			
Modern Times. Settlement Times.	2000	0	Man-made forests widespread.
High Indian mortality.			Exploitation of forests and soil.
America discovered.	1500	500	Indian impacts (cultivation and fire) mold forest character.
MISSISSIPPIAN PERIOD			
Indian cultures largely agrarian, large palisades.	1000	1000	
WOODLAND PERIOD			
Pottery	500	1500	
Corn cultivated; bow and arrow			
Burial mounds	AD 0 BC	2000	Northern pines had moved into Canada while southern pines had moved into Tennessee - their present distributions.
ARCHAIC PERIOD	1000	3000	
Marked increase in Indian population; exchange with other regions.	2000	4000	Sea level rises to modern position.
Beginnings of cultivation with fire as the only feasible tool for land clearing.	3000	5000	“Southern pine rise” results in marked increase in dominance of southern pines in SE.
Archeological evidence that Archaic Indians used total landscape of So. Appalachians.	4000	6000	
	5000	7000	Increased summer warmth and drought.
	6000	8000	
	7000	9000	Central hardwood oak-hickory forests became established.
	8000	10000	Periglacial climate extended as far south as an east-west line thru Ashville, NC
PALEO-INDIAN PERIOD			
Largely hunting/gathering tribes; fire was an available tool.	9000	11000	Temperate, deciduous forests replace Jack pine/spruce/fir/larch in TN and NC.
First evidence of humans in SE	10000	12000	
		15000	Jack pine, spruce and fir are the primary forest types as far south as Tennessee.
		18000	FULL GLACIAL MAXIMA

1600 A.D. waves of European immigrants, looking for a better life and an opportunity to escape the feudal system, made their way to the new world.

Most early settlers were subsistence farmers, a technology the Europeans had adopted 1,000 years earlier (Blethen and Wood 1985). The European model had proved very successful in their homeland, therefore, the immigrants set about clearing the North American forest to plant their crops and graze their livestock, in much the same way as they had “tamed” the European landscape about 1,000 years earlier. They brought with them two things the native people lacked—metal tools (ploughs, axes) and draft animals. Their attitude was that the forest represented an obstacle to be conquered. The forest also harbored wolves, mountain lions and bears which posed a threat to domesticated livestock, such as sheep and hogs.

Although the forest was a challenge to be overcome, forests also provided many of the early settlers’ needs. For example, nearly every family lived in a wooden house and kept their livestock in a wooden barn (usually log construction).

Additionally, wood was the primary fuel for heating. It was not uncommon for a family to use 20-40 cords of wood annually to heat and cook (MacCleery 1992). Wood was used for fencing. MacCleery (1992) estimated that it required about 8,000 fence rails to enclose a 40-acre square field, and by 1850 there were about 3.2 million miles of rail fence in the United States (mostly in the East). Potash was another important product from American forests in the early to mid eighteenth century. The demand for potash in Britain was great during that time. It was used as a soil amendment and as an ingredient in industrial processing (Williams 1989). Hardwoods, especially oaks and maples, produced the highest amounts of potash per unit wood burned and were preferred by potash producers.

From the middle 1700's into the 1800's, many naturalists trekked across North America in pursuit of unknown, unnamed or undescribed species of plants and animals. Some were students of Carolus Linnaeus, the Swedish botanist. Others, such as Andre Michaux (1805) and John Bartram (1751) and his son William (1791) published detailed accounts of their travels through the central hardwood region. From their descriptions and others, such as surveyors' field notes, we can get an idea of what the forest was like at that time.

By this time the Native American populations had been decimated by disease, and the deciduous forests had been virtually free from their effect for between 150 to 250 years. These observers wrote about vast forests of hardwoods. Stephenson (1993) quotes Diss Debar's description of the high-elevation forests of West Virginia on the "table-lands" of the Cheat and Greenbriar mountains: "Here, also neither Oak, Poplar nor Hickory are to be found, but in their room thrive noble specimens of Sugar Maple, Ash, Beech, Birch, Wild Cherry and Black Walnut." But observers who wrote about the Ridge and Valley and Appalachian Plateau forests of lower elevation noticed an abundance of oaks, hickories, maples and yellow-poplar. Using the frequency of mention as an index to species abundance, from John Bartram's (1751) travels through Pensilvania [sic] to Onandaga, the following ranking of species in order of importance was developed (Table 2). Although the age and size of trees and extent of the forest was greater in 1750 than it is today, the species mix looks fairly typical of present-day stands in the northern part of the central hardwood region.

The early immigrants settled along the Atlantic coast and generally used rivers as a primary means of transportation. Westward progress of settlement was relatively slow from 1600 well into the 1700's, mostly due to the lack of a transportation system. The Fall Line of the Peidmont marked the limit of navigability on most eastern rivers; thus, many settlements were established at this position along the frontier. Innovative feats of engineering created water transportation systems, such as the C and O Canal, brainchild of George Washington, but it was only possible to proceed from the Chesapeake Bay to Cumberland, Maryland. The Allegheny Front proved too formidable for the technology of the day. It was the railroads that finally opened the land beyond the Alleghenies. Even though roads (such as the Drover's Road in North Carolina) were used during this time, they were not suitable for transporting heavy materials. Lewis (1995) indicated that it was the construction of the railroads into West Virginia's back counties that allowed a "timber boom" to occur and transformed subsistence farming into "commercial" agriculture in the state.

A fledgling iron industry was developing in the eastern United States, especially in the Appalachian Plateaus. Low-grade iron ore was processed in stone furnaces using local limestone for purification and charcoal for heating. Charcoal was produced by clearing patches of forest 1-3 acres in size (Clatterbuck 1990). Luther (1977) estimated that the 11 furnaces operating in the mid-1800's on the Highland Rim of Middle Tennessee required 375 square miles (240,000 acres) of timber to support them. Similar operations in West Virginia, Pennsylvania and elsewhere combined would have consumed several million acres of forestland in the central hardwood region.

The Industrialization Period, 1861-1929

As is often the case in a post-war era, the Reconstruction period following the Civil War ushered in a period of sweeping change. The steam engine, which could be moved from place to place, was taking the place of water power for milling. The factories of the North were being converted to peacetime production. And the development of a massive rail transportation system was well under way, spurred on initially to supply troops in the war (Fig. 1). With the end of slavery, many large plantation farms were unprofitable. Many subsistence farmers throughout the central hardwood region gave up farming for a more lucrative lifestyle to work in factories, mines or logging camps. Extensive tracts of land were purchased and consolidated by large timber or mineral companies as a speculative enterprise (Eller 1985). The migration of farmers to logging, milling or mining camps was taking place nation wide (Fig. 2), but was more pronounced in areas, such as the central hardwood region, where small subsistence farming predominated.

In addition to the emergence of America as an industrial nation, the Industrial Revolution brought about significant changes in agriculture. Tractors powered by steam, and later gasoline, replaced draft stock, and the use of fertilizers and genetically improved plants and animals increased production per unit area. Up until about 1908, as the population of the United States grew, more land needed to be placed under cultivation, and, conversely, land was taken out of forest production (Fig. 3). The amount of forestland and cropland has stabilized with the small additional withdrawals of forestland coming from urbanization and rights-of-way for highways, pipelines and power transmission. Mechanization of agriculture also caused a shift in the type of land that could be easily cultivated. Much of the steeper land within the central hardwood region became submarginal and was abandoned. The capital investment required to buy large machinery put agriculture on more of a business level than that of a homesteading operation.

Table 2.—Summary of frequency of mention by species (or species groups) from Bartram's (1751) trip through Pennsylvania

Species	Number Times Mentioned	Ranking
white oak and black oak	25	1
white pine	12	2
chestnut	10	3
spruce	10	3
hickory	8	4
sugar maple	8	4
linden	7	5
pitch pine	7	5
elm	6	6
beech	6	6
white walnut (butternut?)	6	6
birch (yellow?)	5	7
poplar (yellow-poplar?)	4	8
ash	4	8
sugar birch (black?)	3	9
great magnolia (cucumber?)	3	9
locust (black?)	2	10
walnut (black?)	1	11
hophornbeam	1	11
plane (sycamore?)	1	11

Several secondary effects of the industrialization period on the central hardwoods were also felt. For example, the factories, machines and vehicles needed fuel which promoted rapid development of the fossil fuel industry. Coal, oil and gas production was spurred on throughout the central hardwood region. Forests were withdrawn from production by surface mining, roads, well sites, pipelines, etc. In addition, the booming economy required wood for construction of factories, new housing for factory workers (either immigrants or relocated farmers), mine timbers, etc.

The booming economy ushered in by the industrial revolution and the vast resource base in North America swept over the timber industry as well. According to Frederick and Sedjo (1991), production of forest products tripled in the United States between 1860 and 1910 (Fig. 4), mostly as a result of increased production of lumber. Lumber production in West Virginia, where hardwoods predominated, showed a similar trend (Fig. 5). The economic boom in North America for consumer goods continued until the beginning of the Great Depression, but the timber boom had peaked in the central hardwood region by 1920, mostly as a result of overcutting. Ahern (1928) in his booklet "Deforested America" stated the situation as follows: "In 1919 the annual drain on our forest resources was estimated at four times the annual growth." Other estimates, although less dramatic, show the same trend where drain exceeded growth during this period (Fig. 6).

The history of the North American logging boom is the subject of several books (Blackhurst 1954, Brown 1923, Clarkson 1964, Fries 1951), and like similar events, such as the California Gold Rush, the timber boom is surrounded by a certain amount of folklore as well as fact. But, in any event, it had a more dramatic effect on our present day central hardwood forest than any single event. Almost all the forests that had any merchantable value were cut over during that period. Only small pockets remained, such as found in the Joyce Kilmer Memorial Forest in North Carolina (Lorimer 1976). The use of steam engines for logging and steam donkeys for skidding exacerbated the effect in causing numerous fires which burned repeatedly through the logging slash and forest floor, exposing the mineral soil to erosion. Eller (1985) quotes Thomas Wolfe of Ashville, North Carolina, as follows: "The great mountain slopes and forest had been ruinously detimbered; the farm-soil on the hillsides had eroded and washed down; high up, upon the hills one saw raw scars of old mica pits, the dump heaps of deserted mines. . . . It was evident that a huge compulsive greed had been at work." Boom towns, such as Davis and Spruce, West Virginia, Tellico Plains and Gatlinburg, Tennessee, and Fontana, North Carolina, developed rapidly; some are completely gone today, and others turned to other industry or tourism to survive. But in spite of the apparent devastation, the forests of the central hardwood region started to regrow, resulting from natural regeneration (stored seed, seed distributed by wind or animals, seedling sprouts or stump sprouts). This period of recovery is the focus of the following section.

The Regrowing Forest, 1930-1996

Well before 1930, several persons in positions of leadership in American forestry had expressed grave concern about the state of the forest in North America. Gifford Pinchot,

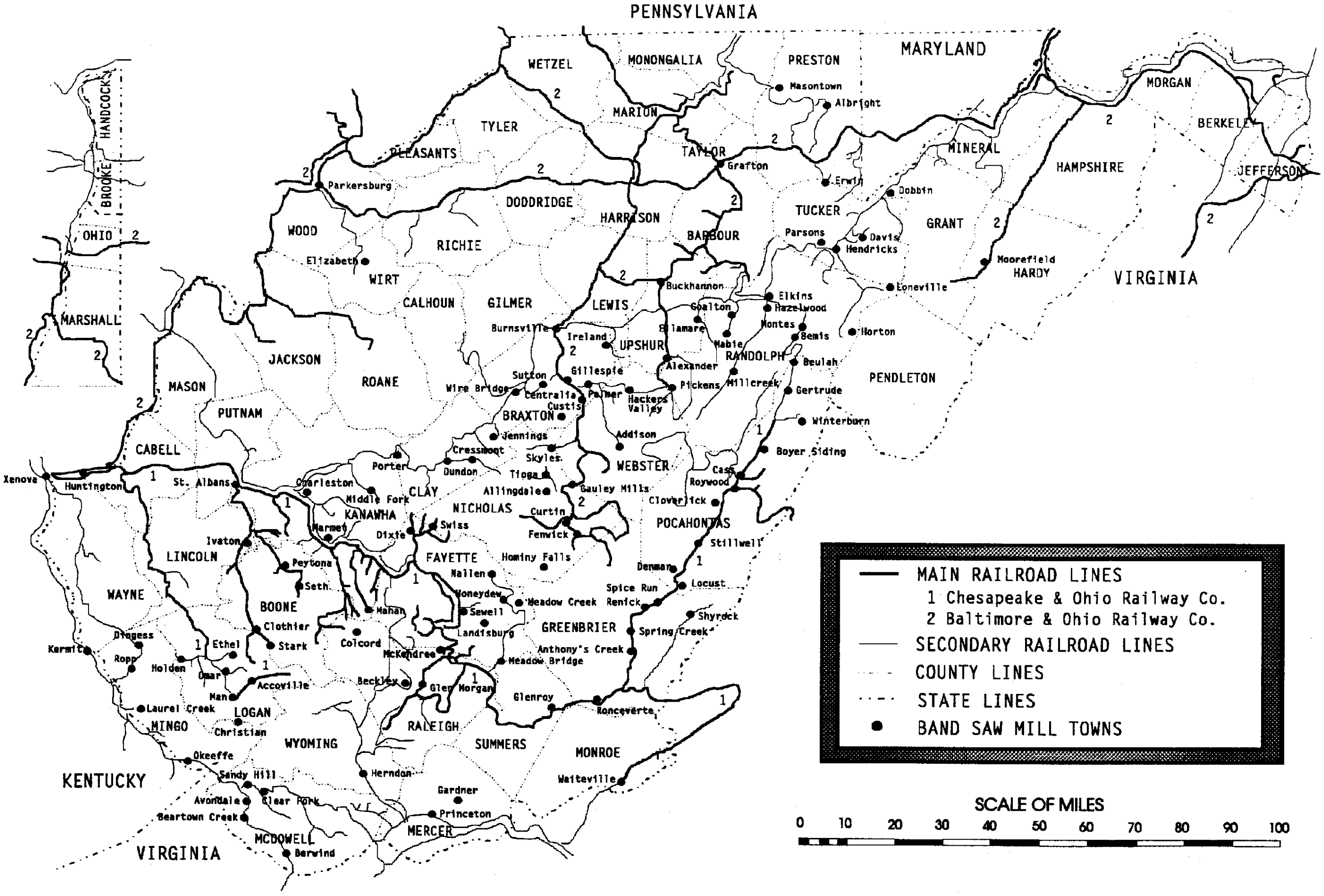
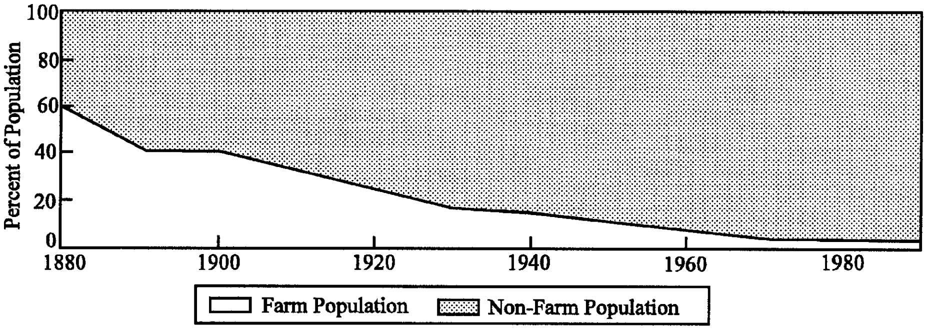


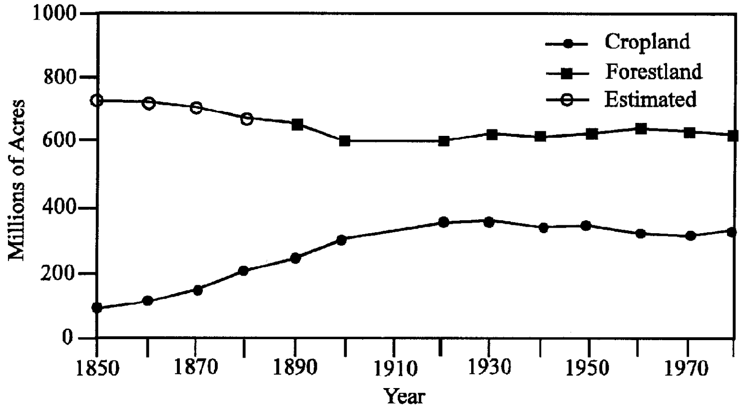
Figure 1.—Clarkson's (1964) map of West Virginia railroads in 1917 and band saw mill towns, 1875-1920.



Source: U.S. Bureau of Census Figures

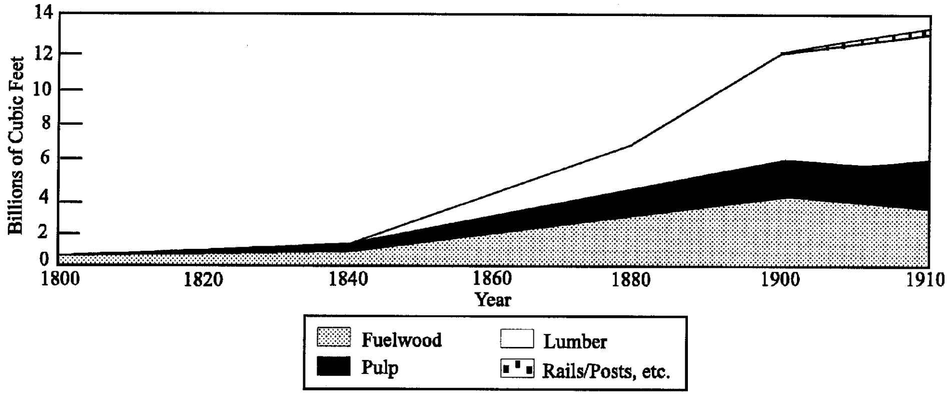
Figure 2.—Farm and non-farm population, 1880-1988, from MacCleery (1992).

generally regarded as the father of American forestry, stated his opinion in Ahern's (1928) publication, as follows: "Forest fires are steadily growing worse in America, and fire prevention is absolutely indispensable. But the axe, carelessly used is the mother of forest fires. The axe and not fire is our greatest danger. Until the axe is controlled there can be no solution of the fire problem, or of the problem of forest devastation in America." Pinchot, who was a consummate politician, had been trained in forestry in Europe. His first job had been as a forester on the huge Vanderbilt estate near Ashville, North Carolina, in 1892 (later to become the site of the Biltmore Forest School), but his greatest accomplishments were in the political arena. He became a Washington political insider and the first Chief of the USDA, Forest Service. He was instrumental in establishment of the Yale School of Forestry and the Society of American Foresters. Pinchot's influence on the central hardwood forest came mostly through the establishment of



Source: RPA Technical Report, RM-175, USDA-Forest Service, 1989.

Figure 3.—Cropland vs forest area, 1850-1980, from MacCleery (1992).



Source: Frederick &Sedjo, RFF (1991)

Figure 4.—Domestic Production, 1800-1910.

the eastern National Forests. Earlier, Pinchot's influence on then President Theodore Roosevelt had resulted in substantial additions to the national forest reserve between 1905-1909. About 80 million acres of federal land in the western states had been set aside as forest reserves as Pinchot and others became increasingly concerned about overcutting in the East.

But the eastern lands had been purchased by private owners and were not so easily annexed by the federal government. The breakthrough came in 1911 with passage of the "Weeks Act" which authorized the federal government to purchase lands in eastern America. Initially, the purpose for acquiring the land was to protect headwaters of navigable streams. Lands which formed the nucleus of many of our current National Forests were purchased under the Weeks Act authority. Almost all the initial purchases were cut-over, and often burned-over, lands belonging to large corporate owners and were bought for prices of $2 to $4 per acre; hence, they were called "the lands that no one wanted."

In 1924, the Clarke-McNary Act added the production of timber to the mandate for National Forest lands. By 1930, the Great Depression was in full swing, and land prices, especially the cut-over forest lands, were extremely low, as corporate owners often viewed the land as a tax burden with no foreseeable harvest of timber in sight. For example, in 1933 a purchase of almost 330,000 acres for the Monongahela National Forest was made at an average cost of $3.43 per acre (USDA, Forest Service 1970). From these beginnings, the eastern National Forests developed (Fig. 7), many within the central hardwood region.

The Weeks and Clark-McNary Acts, in addition to mandating the acquisition of land, also provided funding to states for fire control. The Forest Service also became the lead agency in promoting forest fire control at the federal level. The results came slowly at first, but by the 1960's, forest fires which consumed an average of 40-50 million acres in the 1930's were reduced to 2-5 million acres. (USDA, Forest Service 1987) (Fig. 8). Although the central hardwood region is not as fire prone as conifer-dominated regions of the country, fire has always been a significant disturbance in the oak forests of the central hardwood region (Van Lear and Waldrop 1989). For example, in 1987 more than 50 percent of the land area in Mingo County, West Virginia burned over (Hicks and Mudrick 1993) (Fig. 9).

In 1933, President Franklin D. Roosevelt signed the Civilian Conservation Corps bill into law. The CCC was part of Roosevelt's "New Deal," intended to solve two problems (unemployment and environmental destruction). It was estimated that nearly 250,000 young men who were of employment age, were out of work during the 1930's and Major Stuart, Chief of the USDA, Forest Service, suggested that the army be used to build camps and generally administer the program. But unemployed civilians made up the work force. Many CCC camps were established within National Forests and their activities included planting of trees, fire control, watershed improvements, road improvement and construction of recreational facilities. Although their impact on today's central hardwood forests are largely undetectable, they planted millions of trees (mostly conifers) on denuded lands, and many of the recreational structures they built are still usable today, almost 60 years later.

The Depression drove more people from marginal farms, and the trend in reversion of farmland to forest continued through the 1930's. In 1942, the Depression ended with America's entry into World War II. The war accomplished what governmental programs could not by employing millions of Americans, either as soldiers or in the factories that manufactured war materials. The war put technological development into high gear, and many of the devices that have significantly affected our society were developed during this period. The post-war economy was booming when soldiers returned home, many to take factory jobs, marry and raise families. The impact of these developments on the central hardwood region was a shift in the lifestyle and objectives of land owners. Many people kept their residence in a rural setting while seeking employment in mills and factories elsewhere. The reason for owning land shifted away from utilitarian (production) objectives toward recreational/ aesthetic enjoyment/ residential objectives (Birch and Kingsley 1978) (Fig. 10).

The changing land ownership characteristics have continued into the most recent decades throughout the central hardwood region. This is illustrated by the changes that occurred from 1957 to 1975 in West Virginia, a state that experienced little or no total population growth over that period. In 1957, the West Virginia Forest Industries Committee reported that there were 133,570 private owners of forestland, who owned over 8.9 million acres. The average size of ownership was 66.6 acres in 1957. By 1975, the number of owners had increased to 207,500 and the forestland acres had also increased to 10.3 million acres. But the average size of ownership had dropped to 49.8 acres (Birch and Kingsley 1978). Twenty percent of the land owners in 1975 were not from rural or farm backgrounds. Twenty-two percent of the land owners held their land primarily for aesthetic enjoyment or recreation, and another 23 percent owned the land primarily for their residence. Only 3 percent owned land primarily for timber production. Similar changes have taken place throughout the central hardwood region.

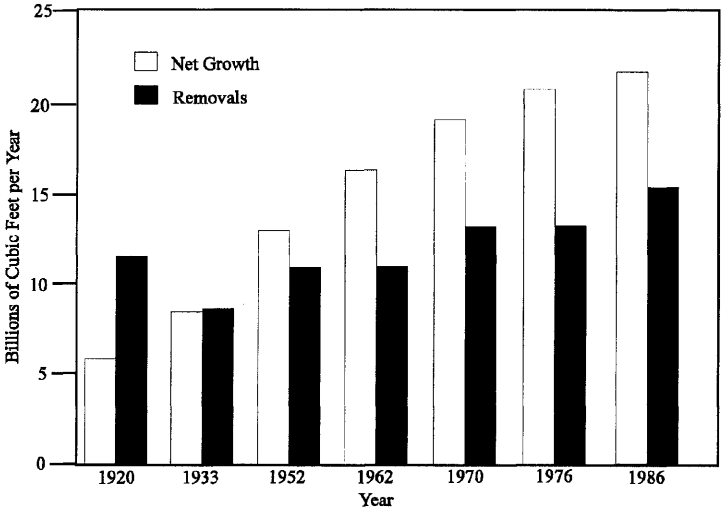


Figure 6.—Timber growth/removals from MacCleery (1992).

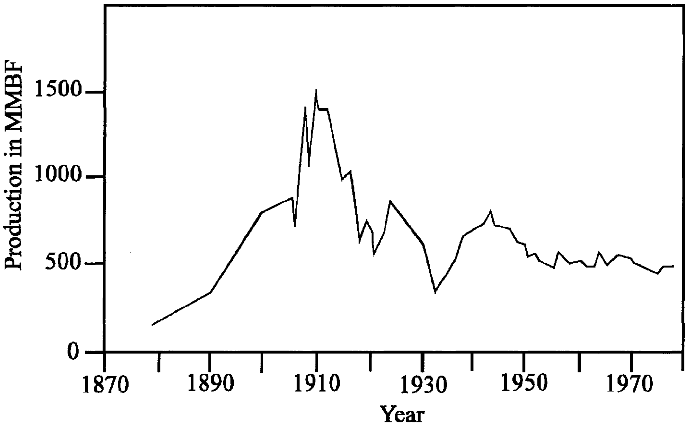


Figure 5.—West Virginia lumber production from Zinn and Jones (1984).

Alig (1990) reported the trends in forested land by states. If the 11 states that constitute the bulk of the central hardwood region are summarized (Table 3), the trend in total forestland is depicted in Figure 11. A decrease of about 2.5 percent in forestland is projected to occur by 2040, mostly due to urban expansion.

Agricultural policy continued to have an impact on forestry throughout the post-war period. The Conservation Reserve Program in the 1960's provided compensation to farmers to convert cropland to forest by planting trees. The Forest Incentives Program provide cost sharing to small private land owners for tree planting and other forest practices. The Stewardship Incentives Program of the 1990's provides federal cost sharing to small forest land owners to develop and carry out planned forest management. Some states have offered additional property tax incentives to land owners enrolled in the Stewardship Incentives Program.

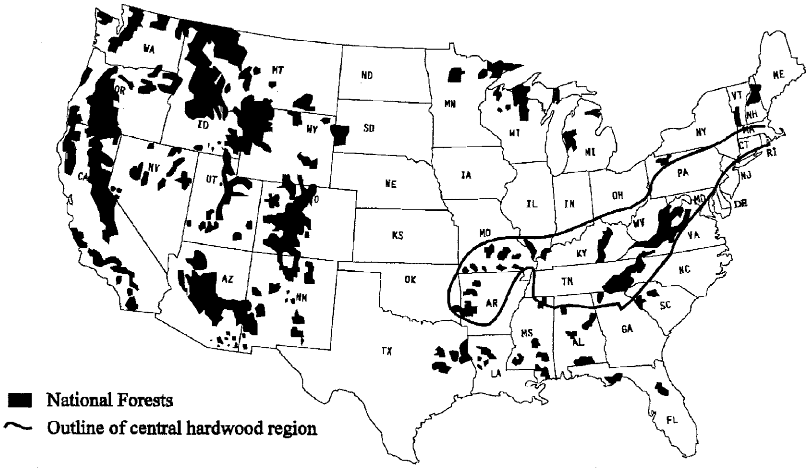


Figure 7.—Distribution of National Forests.

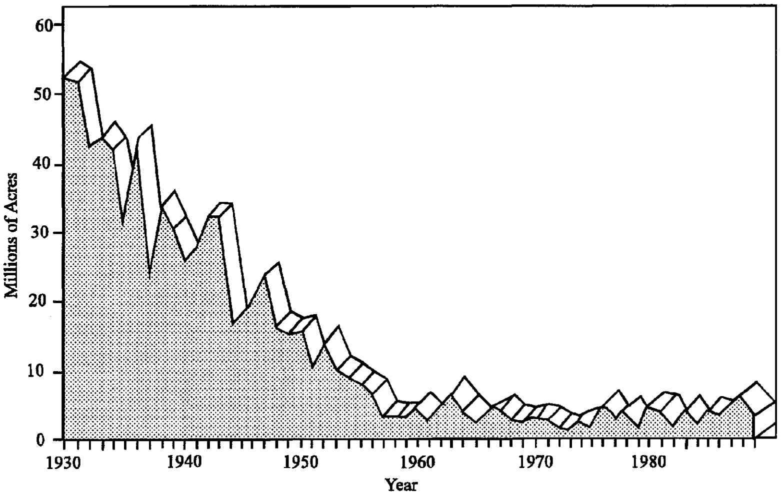


Figure 8.—US wildfire trends, from MacCleery (1992).

These programs are designed to promote good forest management among the many and varied small private land owners, a type of ownership that predominates in the central hardwood region. Technical support for forestry is provided to private land owners through state forestry agencies, Agriculture Extension Agents and the USDA Forest Service, State and Private Forestry. But in spite of this large effort, the level of participation among private forest land owners in the central hardwood region has generally lagged behind government expectations, partly due to the inherent independence of property owners, partly due to a lack of interest or economic incentives and partly due to an inability to inform land owners of the availability of such programs.

The withdrawal of forestland for agriculture leveled off in about 1933, but during the post-war decades, some decline in forest area has occurred. This has mostly resulted from development-related activities. One example is the interstate highway system which was developed during the 1960's and 1970's for the most part. Rights-of-way for gas transmission and electric power transmission also consume large corridors through the forest, and surface mining and urban development have taken up significant amounts of forestland. Topography and economic factors have resulted in the central hardwood region's being an area that is currently among the highest in the eastern United States for coverage with natural vegetation (Kolpatek and others 1979) (Fig. 12).

Some of the most significant changes to the central hardwood forest in recent years have been brought about as a result of human introduction of insects and disease organisms. These changes are often not dramatic, but their impact is unquestionably large. The list of important introduced pests includes dogwood anthracnose, Dutch elm disease, oak wilt and beech bark disease. But the two with the most impact or potential impact on the central hardwood forest are chestnut blight and gypsy moth. The former is a disease that has effectively eliminated a species, which at one time was among the most widely distributed and economically important in the central hardwood region. The latter is an introduced insect that continues to spread southwestward through the central hardwood region and seems to have the potential to spread through the entire region over the next 50 to 75 years.

The chestnut blight fungus (*Cryphonectria parasitica*) was introduced to the New York Botanical Gardens in 1904 (Giddings 1912, Murrill 1904). By 1915 most states east of the Mississippi River reported infected trees. By the late 1930's, most mature chestnut trees were dead throughout the eastern states. The following should provide a frame of reference for the significance of American chestnut. Chestnuts were a source of food and commercial enterprise for mountain farmers. During the settlement period, log cabins were often constructed of chestnut logs due to their durability and decay resistance. Most split-fence rails and posts were constructed of chestnut where it was available. And due to its durability, chestnut lumber was widely used in construction of barns, sheds and even for the manufacture of furniture. The bark of chestnut was an important source of tannin for leather tanning factories.

Chestnut also occupied an important position in the hardwood ecosystem. Braun (1950) used the name "oak-chestnut" to identify one of the most significant forest formations in the eastern deciduous forest. American chestnut was an important source of mast for a variety of wildlife species (animals and birds) most of which served as prey for larger predatory mammals. The impact of the loss of American chestnut can never be fully assessed. But Brooks (1915) indicated the following facts regarding chestnut in West Virginia alone. The annual harvest of chestnut in West Virginia in 1915 was 118 million board feet. The entire volume of chestnut in West Virginia was estimated to be 5 billion board feet. At a stumpage value of $8 per thousand board feet, in 1915, the entire chestnut inventory was estimated to be worth $55 million. At current stumpage prices for red oak that value would be about $1.5 billion. The only evidence remaining of the once important chestnut are occasional root sprouts, which usually survive to the sapling stage before being killed by the blight, and the grey remains of stumps of salvaged trees. Ironically, many of these stumps still have coatings of charcoal, evidence of the fires that burned over the area, probably after the chestnuts had already succumbed to the blight.

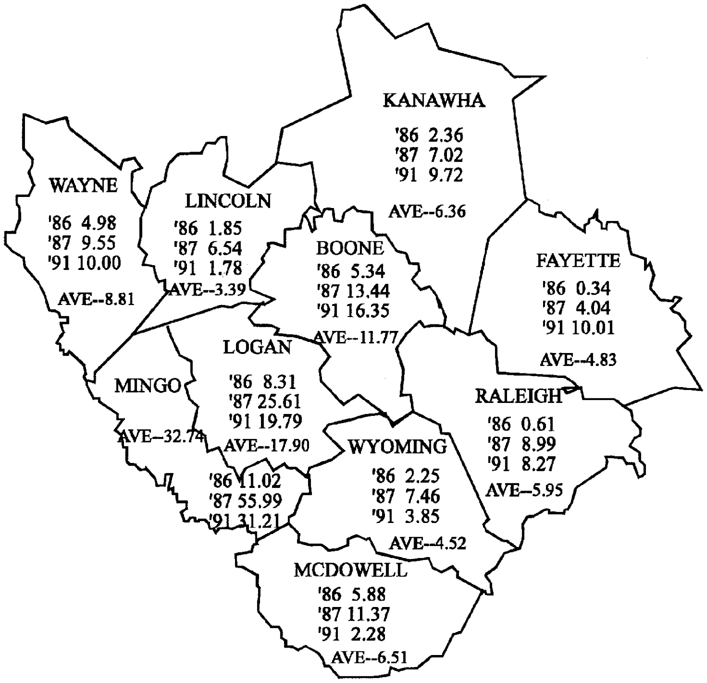


Figure 9.—Percentage of West Virginia counties burned in 1986, 1987 and 1991.

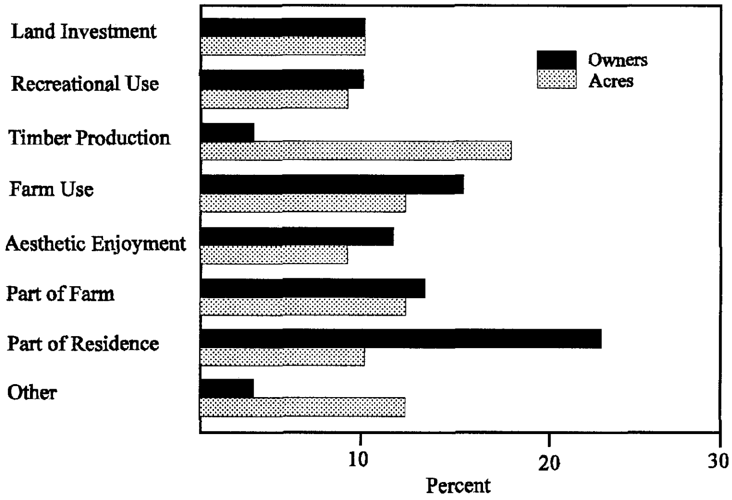


Figure 10.—Reason for owning land in West Virginia (Birch and Kingsley, 1978).

Gypsy moth (*Lymantria dispar*) is an introduced insect that potentially could have as great, or greater, impact on the central hardwoods than chestnut blight. Gypsy moth was inadvertently introduced into the Boston area in 1869 by Leopold Trouvelot (Liebhold 1989), who was interested in hybridizing the gypsy moth with native silkworms. The spread of gypsy moth has continued, and it is potentially capable of becoming established throughout the central hardwood region. Although gypsy moth can defoliate a variety of tree species, including conifers, deciduous species are their preferred hosts. Oaks are among the most preferred hosts of gypsy moth (Bess and others 1947, Gansner and Herrick 1987). Black cherry, hickory and maples are intermediate in preference while species, such as yellow-poplar and ashes, are virtually resistant. The significance of this to the central hardwood region is the fact that susceptible species make up the bulk of the forest composition throughout the region, the exception being yellow-poplar, especially in the mixed mesophytic forest.

Table 3.—Summary of forested lands by state in the central hardwood region

	(Millions Acres)									
						Projected				
States	1952	1962	1970	1977	1987	2000	2010	2020	2030	2040
NC	19.6	20.0	20.1	19.4	18.4	17.7	17.6	17.5	17.5	17.5
VA	15.5	15.7	15.9	15.9	15.4	15.0	15.0	15.0	15.0	15.0
AK	19.6	19.9	18.0	16.8	16.6	15.6	15.4	15.1	15.1	15.1
KY	11.5	11.6	11.8	11.9	11.9	12.1	12.2	12.3	12.5	12.5
TN	12.5	13.4	12.8	12.9	12.6	12.4	12.3	12.3	12.3	12.3
MD	2.9	2.8	2.7	2.5	2.5	2.4	2.3	2.2	2.1	2.1
MA	3.3	3.0	2.8	2.8	3.0	3.0	2.9	2.8	2.7	2.6
PA	14.6	16.3	16.1	15.9	16.2	16.3	16.3	16.0	16.0	15.8
WV	10.3	11.4	11.4	11.5	11.8	11.8	11.8	11.8	11.8	11.8
OH	5.4	6.0	6.4	6.9	7.1	7.3	7.3	7.1	7.0	7.0
MD	14.3	13.5	12.5	12.3	12.0	12.0	12.0	11.9	11.9	12.0
Total	129.5	133.6	130.5	128.8	127.5	125.6	125.1	124.0	123.9	123.7
Change (from 1987)	+2.0	+6.1	+3.0	+1.3	0	-1.9	-2.4	-3.5	-3.6	-3.8
Percent Change	+1.6%	+4.8%	+2.3%	+1.0%	0	-1.5%	-1.9%	-2.7%	-2.8%	-2.9%

Currently, gypsy moth has become established in, and is endemic to, an area stretching from Michigan to North Carolina (Fig. 13). But isolated populations occur throughout the central hardwood region, having been spread by movement of vehicles, logs or other objects containing egg masses.

As a defoliator, gypsy moth's impact on the tree is to reduce its photosynthetic surface. The major defoliation occurs during the spring (May-June) which is a period of critical importance to deciduous trees since the soil moisture is generally high and the temperature is moderate during this time. Usually one defoliation is not lethal to trees, but a single defoliation during a drought year or multiple years of defoliation can lead to extensive tree mortality (Fosbroke and Hicks 1989). The factors that precipitate extensive gypsy moth outbreaks include presence of a suitable host plus climatic conditions that favor buildup. Since gypsy moth is situated in an oak-dominated region, the host is generally suitable, thus outbreaks can occur whenever weather triggers them. As can be seen in Figure 14, cycles of outbreaks seem to occur about every 10 years. The outbreaks in recent years have been affecting more acres than in previous outbreaks, mostly due to the spread of gypsy moth to occupy a larger endemic range.

The gypsy moth story is still unfolding, but its impact on the central hardwood forest has already been experienced throughout the portion of the region where it has spread. As reported by Gansner and Herrick (1987), oak experienced an average mortality rate of 24.2 percent between 1979 and 1984. Quimby (1987) reported a similar result where summer droughts plus gypsy moth defoliation in 1980-1983 resulted in the premature loss of 68 million trees in Pennsylvania, a mortality rate of 27.6 percent. Much of the mortality occurred in oak stands. Hicks and Fosbroke (1987) reported similar rates of tree mortality in southwestern Pennsylvania and western Maryland. The forest invaded by gypsy moth as it entered northeastern Pennsylvania has been dubbed “the new frontier” by Herrick and Gansner (1988), and essentially this new frontier represents the oak-dominated forest of the central hardwood region. The reduction in oak stocking that has occurred in the part of the central hardwood forest where gypsy moth has become established will most likely represent what can happen throughout the entire region.

Another episode in recent history that has had a significant impact on the central hardwood forest is the decline and later reestablishment of wildlife populations across the region. In addition to white-tailed deer, elk and bison were found extensively throughout the region prior to European settlement. Those species were, in part, hunted into extinction but mostly were displaced due to habitat destruction by land clearing for farming. Farmers also eradicated predator species, such as eastern mountain lion, eastern timber wolf and black bear, since these predators were a threat to their livestock. By 1910, bison, elk, mountain lion and wolf had been eradicated from the central hardwood forest, and white-tailed deer and black bear were found only in small isolated populations in remote and mountainous regions.

Wildlife advocates and hunters pushed for, and obtained, stricter hunting regulations, such as seasonal hunting, bag limits and bucks-only hunting. The results for white-tailed deer have been remarkable. In the absence of natural predators, and with the second-growth hardwood forest providing prime habitat, the deer herd has risen sharply throughout the central hardwood region. In fact, deer have generally become a serious problem to forest regeneration in many areas. Deer consume 1-1.5 kg of vegetation daily, consisting of twigs, acorns and some grass and herbs (Smith 1993). Deer are selective and prefer seedlings of oaks and maples to those of black cherry (Marquis 1981). Overbrowsing by deer can result in development of a fern-dominated understory where tree seedlings have a difficult time getting established due to competition and allelopathy (plant toxins). Thus, the long-term effect of deer on the forest may be to retard the process of regeneration, selectively change the species composition and reduce the overall diversity of species.

The modern chapter in forest history also includes a marked change in the social context of forest resources. This has been brought about by a combination of factors including the changing demographics in America, where people today are physically removed from the land. But other factors have come into play. For example, abusive and exploitive land uses, such as exploitive timbering, unregulated mineral extraction, air and water pollution and uncontrolled development are all sources of public concern. The environmental movement began to take shape as a cultural phenomenon during the late 1960's. Rachel Carson's book, *Silent Spring*, did for environmentalism what Harriett Beecher Stowe's *Uncle Tom's Cabin* did for the emancipation movement. Against a backdrop of Kennedy altruism and polarizing issues, such as the cold war, racism and a budding Viet Nam conflict, it seems, in retrospect, inevitable that the environmental movement, led by altruistic urban youth, should flourish. Most foresters, at that time, were from rural backgrounds and had come through their forestry training with a paternalistic approach toward resource management, being one in which the forester is regarded as the trained professional, and should be trusted to make the appropriate decisions regarding management of the resource. Thus, the elements of conflict were in place. The political arena became the venue for the struggle. Environmental activist groups gained strength. The Wilderness Act of 1964 represented a milestone in environmental policy legislation whereby areas in the National Forests could be set aside for "wilderness," and timber cutting in these areas would be prohibited (Dana and Fairfax 1980). Controversy over the use of clearcutting on public land resulted in the landmark Monongahela Decision of 1970 that recommended uneven-age management as a primary management policy for the Monongahela National Forest in West Virginia. This decision was more significant as a precedent than it was to the Monongahela National Forest. In effect, it imposed the will of the public over that of foresters, who had always been regarded as the experts in forest resource management.

The issue of the 90's seems to revolve around regulation of private land. The so-called "spotted owl" controversy in the Pacific Northwest has set the tone. The perceived situation was one in which an endangered organism's habitat (old-growth forest) was not limited to a single land owner but extended to private land as well as public land.

The debate extending to small non-industrial private forest (NIPF) owners has progressed from the passage of regulations requiring the use of "best management practices" for logging to a more comprehensive and controversial debate over whether or not land use should be controlled at the landscape level. If such control is needed to facilitate what is called ecosystem management, how can it work in areas, such as the central hardwood region, where over 75 percent of the land is owned by NIPF owners? Proposed solutions include a combination of government educational and assistance programs plus incentive programs (Campbell and Kittredge 1996). The use of legislative regulation to achieve the goals of ecosystem management is a controversial approach (Argow 1996) and one that evokes strong emotions. The fear among private owners- that they will lose prerogatives to use their land- is not unfounded. Examples abound where the "general good" of the public has taken precedence over the desires of the individual.

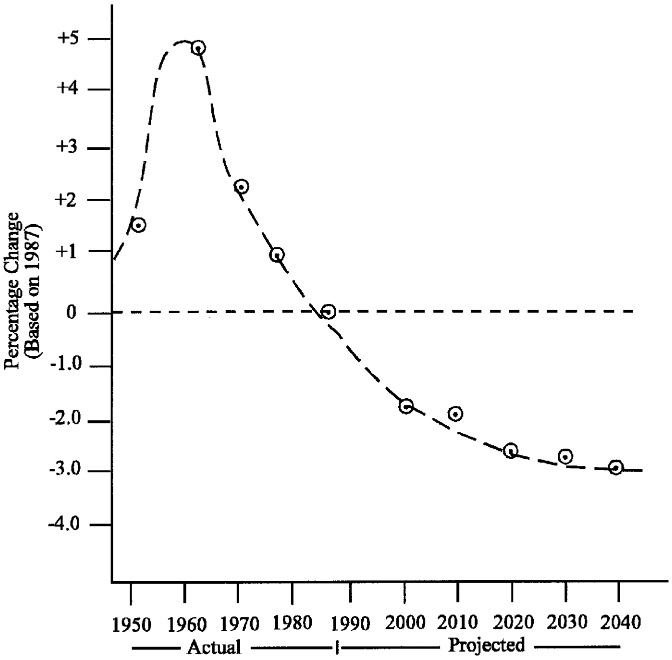


Figure 11.—Change in forestland (actual and predicted) for 11 states with significant area within the central hardwood region (AR, KY, MA, MD, MO, NC, OH, PA, TN, VA WV), 1950-2040, from Alig (1990).

On the other side of the issue are those representing private land owners (Farm Bureau, National Woodland Owners Association, etc.) who contend that the loss of management options to a private land owner due to regulation constitutes a "taking" under the Fifth Amendment of the Constitution, for which the owner is entitled to be compensated. As these debates continue, the outcomes will most certainly affect the way forests can be, and will be, managed and exploited in the central hardwood region in the years to come.

In summary, the history of the central hardwood forests is one that is based on the actions and interactions of climatic and biological factors, and, in more recent times it has been strongly affected by humans and their actions (Table 1).

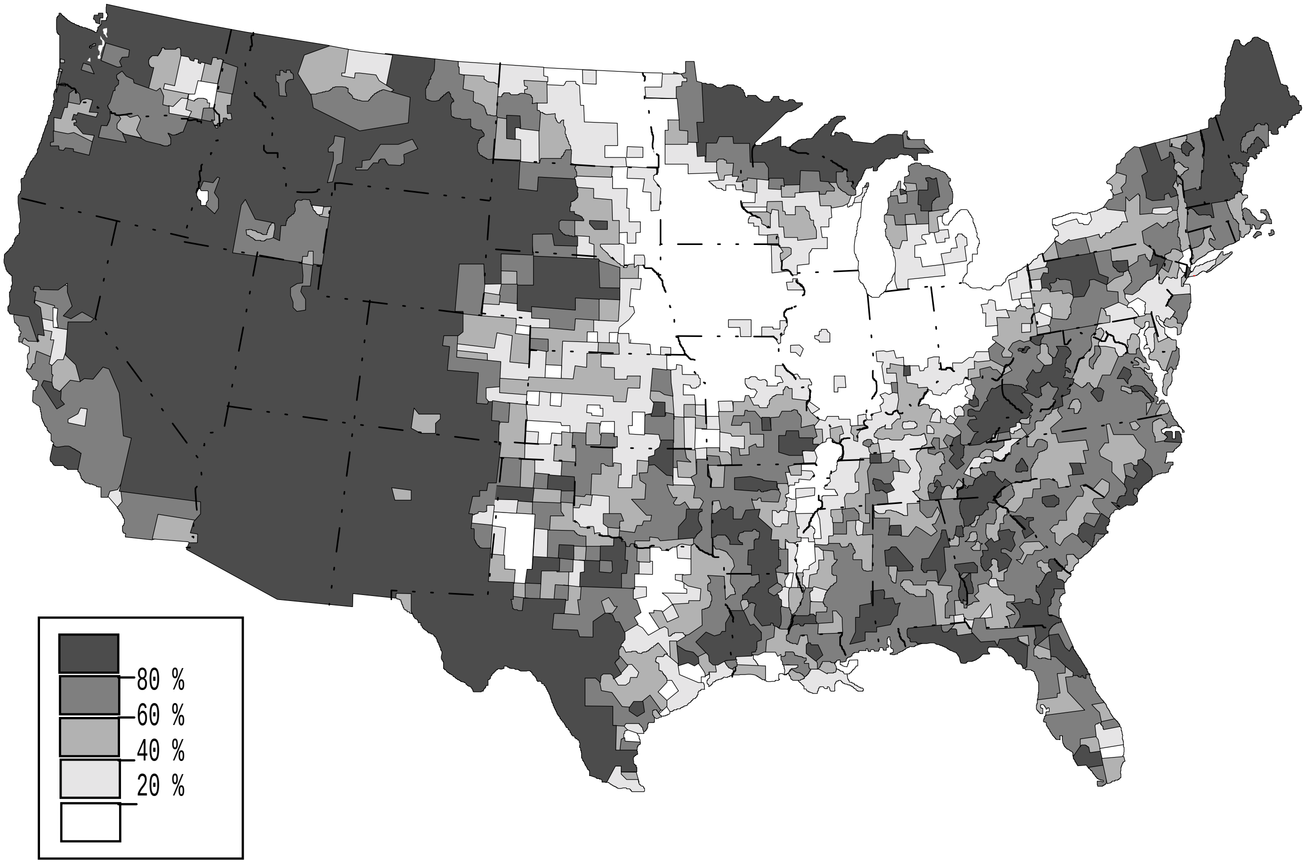
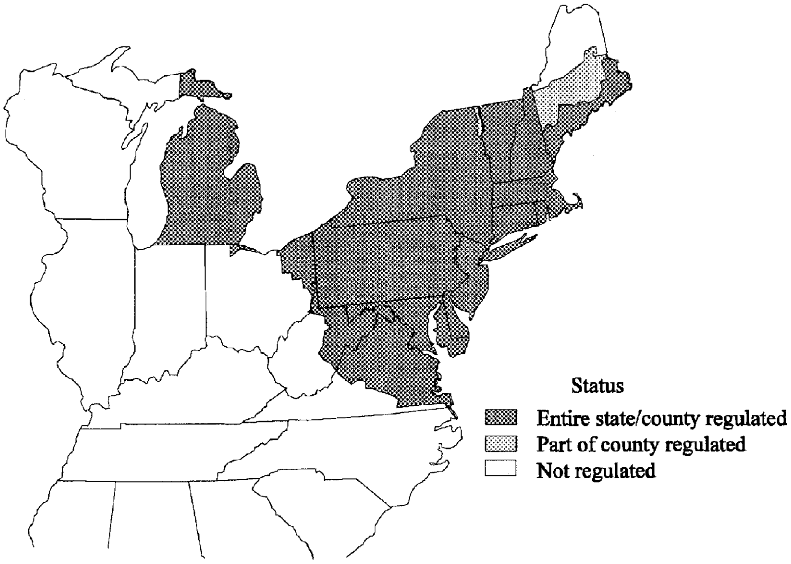


Figure 12.—Coverage of natural vegetation by county in the US (Klopatek and others 1979).

Going back to the retreat of the last glaciers (12,000-14,000 years ago) and the climatic warming trend that has followed, the deciduous forests found an ideal environment and flourished in the central hardwood region. Indigenous people used fire and cultivated land, and their activities peaked about the time European settlement began. The popular notion of an undisturbed "primeval" forest that predated European settlement seems largely to be a myth. After native people were decimated by disease, forests regrew for about 200 years, and then clearing for subsistence farming began in ernest. This proceeded until about the time of the Civil War and the onset of the industrial revolution. This period marked the beginning of a shift in agriculture to more level land, and abandoned hill lands in the central hardwood region began to regrow into forest. A period of exploitive logging around the turn of the 20th century has been followed by a regrowing forest. Forest land owners and the public have changed in their attitudes concerning forest resources in recent times.

Now with many hardwood stands in the central hardwood region approaching 100 years of age, the resource is maturing. Many forces are mustering for a piece of the action. Developers, speculators, industries, agencies, advocacy groups and environmentalists are all becoming aware of this unique and diverse forest. Our actions today will be the history of the future, and foresters should be prepared to participate in, and hopefully guide, the decision-making process at this important crossroad in the central hardwood regions development.



Map provided by the USDA Animal and Plant Health Inspection Service

Figure 13.—Gypsy moth quarantine area as of 1996.

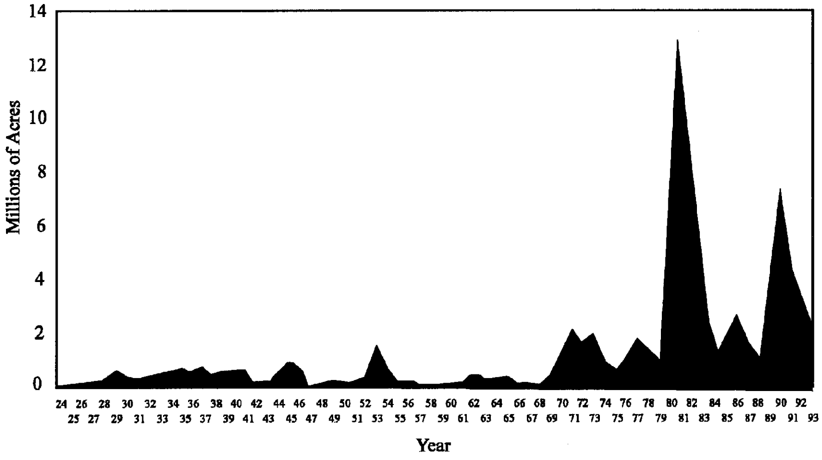


Figure 14.—Outbreak history of gypsy moth in the Northeast.

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Native Burning in Western North America: Implications for Hardwood Forest Management

Charles E. Kay[1]

Abstract

It is now widely acknowledged that frequent low-intensity fires once structured many western forests. What is not generally recognized, however, is that most of those fires were purposefully set by native people, not started by lightning. Data from the Rocky Mountains attest to the widespread use of fire by native people, as does the ecology of aspen, the only common deciduous hardwood in the West. Fire history studies all show that aspen once burned at frequent intervals, yet aspen will readily burn only when the trees are leafless and the understory dry — conditions which occur only early in the spring before leaf-out and understory regrowth, or late in the fall after leaf-drop and the understory has been killed by frost. During both these periods, though, there are few lightning strikes and virtually no lightning-started fires in the Rocky Mountains. Thus if aspen burned frequently in the past, as all evidence indicates it once did, then those fires must have been started by native people, who used fire to modify plant communities for human benefit. Similarly, eastern deciduous forests will readily burn only when leafless, but during that time there are few lightning strikes — one of many indications that aboriginal burning was also common in the eastern U.S.

Introduction

It is now generally acknowledged that frequent low-intensity fires were once common and historically structured many western forests (Barrett and Arno 1982, Kay 1995, Kloor 2000). In addition, it is widely assumed that most, if not all, of those earlier fires were started by lightning (Kloor 2000, Loope and Gruell 1973:434, Romme and Despain 1989). Data from Arizona, New Mexico, Idaho, Yellowstone, and the southern Canadian Rockies, though, all attest to the widespread use of fire by native people, as does the ecology of aspen (*Populus tremuloides*), the only common deciduous hardwood in the West.

Aboriginal Burning in the West

Selway-Bitterroot Wilderness Area

Brown and Bradshaw (1994) and Brown and others (1994, 1995), for instance, compared the U.S. Forest Service's Prescribed Natural Fire Program with pre-European settlement fires in the Selway-Bitterroot Wilderness Area along the Montana-Idaho border. Based on stand-age analyses and fire history maps, Brown and others (1994, 1995) determined how frequently various forest types burned in the past and then compared those data with how frequently the same vegetation types burned from 1979-1990 when lightning-caused fires were allowed to run their course. Brown and others (1994, 1995) reported that, on average, the area burned during pre-European times was nearly twice as great as the area burned by lightning fires alone today. Moreover, low-elevation montane areas that once had the highest fire frequency, now seldom burn. Since the overall climate has not changed, it is unlikely that lightning-caused fires burn less area today then they did in the past. Instead, it is likely that there are fewer fires today because native people no longer use fire to manage the land, as they once did.

Southern Canadian Rockies

A similar situation exists along the east slope of the southern Canadian Rockies. In the past, fires were exceedingly frequent, while today lightning-caused fires seldom occur (Barrett 1996, Heathcott 1999, Kay and others 1999, Kay and White 1995, White 1985). In some vegetation types, fire-return intervals are now 100 times greater than they were in the past. Lower montane valleys that once burned every 5 years or less now do not burn at all. Based on this and other evidence, Parks Canada has concluded that native burning, not lightning-caused fires, was critical in maintaining what heretofore was believed to be the "natural" vegetation mosaic of the southern Canadian Rockies (White and others 1998). That is to say, there simply are not enough lightning-caused fires to account for historical burn and vegetation patterns (Heathcott 1999).

Yellowstone National Park

Prior to park establishment, Yellowstone's northern grasslands had a fire-return interval of once every 25 years (Houston 1973). Yellowstone has had a "let burn" policy for nearly 30 years now, yet during that period, lightning-caused fires have burned practically none of the northern range. In 1988, fire did burn approximately one-third of the area, but according to agency definitions that was "unnatural" because the fire was started by a man, not lightning. Moreover, the fire that burned the largest portion of the northern range started outside the park and was driven by unusually strong winds more than 70 km before it reached the northern range. Besides, the 1988 fires are thought to be a 100-300 year event (Schullery 1989a, 1989b), so similar fires could not have caused the original 25 year fire frequency.

Despite a series of recent droughts, why has Yellowstone's northern range remained virtually unburnt? Park biologists contend that this is because "lightning has chosen not to strike very often on the northern range" (Despain and others 1986:109). That assertion, though, is not supported by data from the Bureau of Land Management's Automatic Lightning Strike Detection System which shows that, on average, lightning strikes the northern range 4 times per km^2/yr (Kay 1990:136-137). So lightning strikes, but why doesn't the

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[1] Adjunct Assistant Professor, Department of Political Science, Utah State University, Logan, UT 84322.

Figure 1—An example of the ponderosa pine communities that were common throughout the West at European contact. These early forests were so open and park-like that it was possible to go nearly everywhere with horse-drawn carriages. These pine forests also supported a lush understory of grasses and forbs, while today, forest ingrowth has created a forest health crisis and understory forage production has fallen to virtually nothing (Uresk and Severson 1989, 1998). Photo taken on the south rim of Arizona's Grand Canyon by R. Arnold (No. 166) in 1905 — courtesy U.S. Geological Survey Photographic Archives, Denver, CO.

range burn? The answer is that when most lightning strikes occur, the herbaceous vegetation is too green to carry a fire (Kay 1995). Thus, it is likely that the park's original 25 year fire frequency was entirely the product of aboriginal burning.

Arizona and New Mexico

At European contact, ponderosa pine (*Pinus ponderosa*) forests in Arizona, New Mexico, and throughout the Rocky Mountains were open and park-like (see Figure 1), but have since developed into impregnable thickets due to the ingrowth of smaller trees, which, in turn, has created the current forest health crisis (Covington and Moore 1994, Fule and others 1997). The open nature of the original forests, as well as the more recent proliferation of smaller trees, is generally attributed to modern fire suppression and the lack of lightning fires. That is to say, it is commonly believed that lightning historically was the primary ignition source, not native people (Seklecki and others 1996, Swetnam and Baison 1996a). This interpretation, though, is not supported by lightning frequency data or time of fire-scar analyses.

In the southwest, over 95% of lightning strikes occur after July 1st (see Figure 2a), while, historically, 85% or more of ponderosa pines were scarred by fire during April, May, and June (see Figure 2b). Now, despite the relatively low incidence of lightning, lightning fires early in the year do burn a disproportional area due to generally dry conditions at that time of year (Baison and Swetnam 1990:1562, Barrows 1978, Swetnam and Betancourt 1990), but lightning fires alone still cannot account for the magnitude of early-season fire scarring seen during pre-European times (Barrows 1978). In many mountain ranges today, there simply are not enough lightning fires to have caused the high fire frequency observed prior to European settlement (Baison and Swetnam 1997:3). Thus, it is logical to assume that a large proportion of the "natural" fire regime in pine forests and

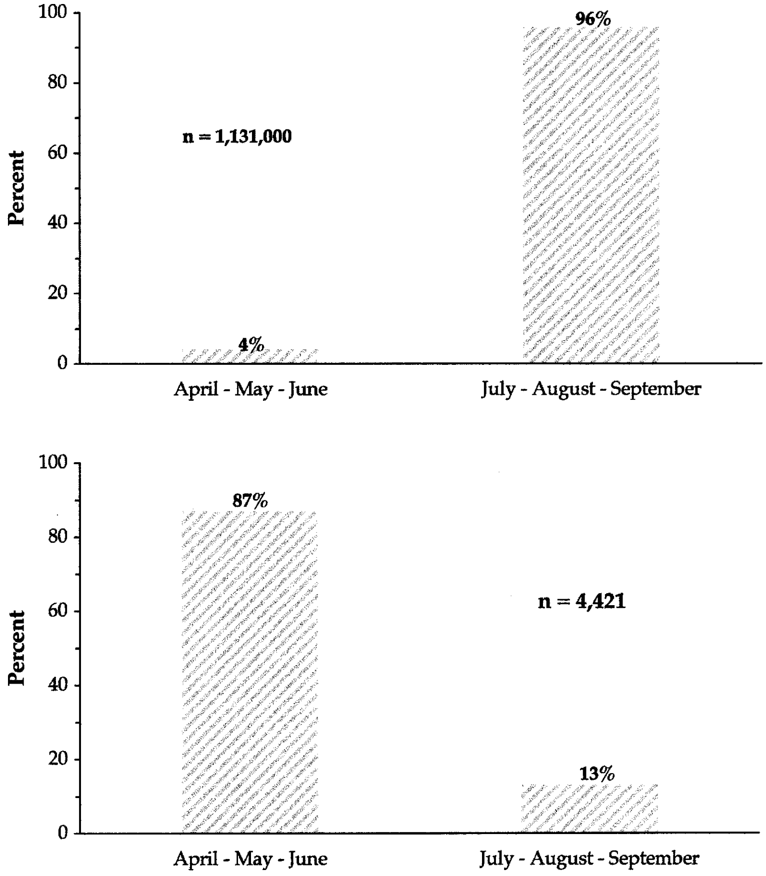

Figure 2—(a) Frequency distribution of cloud to ground lightning strikes in Arizona and eastern new Mexico during 1989 and 1990. From Watson and others (1994:1720). (b) Frequency distribution of fire-scar data for the southern Rockies. Clearly most trees were scarred by fire when there were few lightning strikes. Timing of fire scars determine by microscopic analysis of when individual growth rings were damaged by fire. From Brown and Sieg (1996), Fule and Covington (1999), Fule and others (1997), Morino (1996), Swetnam and Baison (1996b).

other regions of the southwest was actually due to aboriginal burning (Bonnicksen 2000).

Aspen Ecology

Repeat photographs and fire history studies indicate that Rocky Mountain aspen communities burned frequently in the past, yet experience has proven that aspen is extremely difficult to burn (Brown and Simmerman 1986). Terms such as “asbestos type” and “firebreak” are often used to describe aspen (DeByle 1987:75). Even raging crown fires in coniferous forests seldom burn adjacent aspen communities (Fechner and Barrows 1976). DeByle and others (1987:75) noted that “wild fires that had burned thousands of acres of shrubland or conifer types during extreme burning conditions usually penetrated less than 100 feet into pure aspen stands.” Lightning-fire ignition rates for aspen are also the lowest of any western forest type, and overall ignition rates are less than half that for all other cover types, including grasslands (Fechner and Barrows 1976). At current rates of burning, “it would require about 12,000 years to burn the entire aspen type in the West” (DeByle and others 1987:73). Something is clearly different today than it was in the past.

Research has shown that these communities will readily burn only when aspen is leafless and the understory plants are dry, conditions that occur only during early spring and late in the fall (Brown and Simmerman 1986). Prior to May 15th and after October 1st, though, there are few lightning strikes and virtually no lightning fires in the northern or southern Rocky Mountains (Kay 1997a, 1997b; Nash and Johnson 1993) — see Figures 3 and 4. So if aspen burned at frequent intervals in the past, as fire-frequency data and historical photographs indicate it did, then the only logical conclusion is that those fires had to have been set by Native Americans (Kay 1997a, 1997b).

Why did Native People Burn?

Thus, it is increasingly clear that Native Americans had a major impact on ecosystems by repeatedly firing the vegetation (Bonnicksen 2000). They did this to modify plant and animal communities for human benefit and to increase productivity (Pyne 1995). In California, for instance, native peoples had 70 reasons for burning (Lewis 1973), and even in northern Canada, where the vegetation is less diverse, Native Americans still set fires for at least 17 different reasons (Lewis and Ferguson 1988). Native peoples commonly set fires to enhance forage production and to attract prey; to herd wildlife during hunts; to rid the forests of underbrush and to facilitate travel; to enhance plant production such as berry producing shrubs, mast species, and root crops; to destroy poisonous snakes and other vermin; to clear the land prior to farming; to kill trees for firewood; to rid the land of places an enemy could hide; and as an instrument of war, among many others. Although

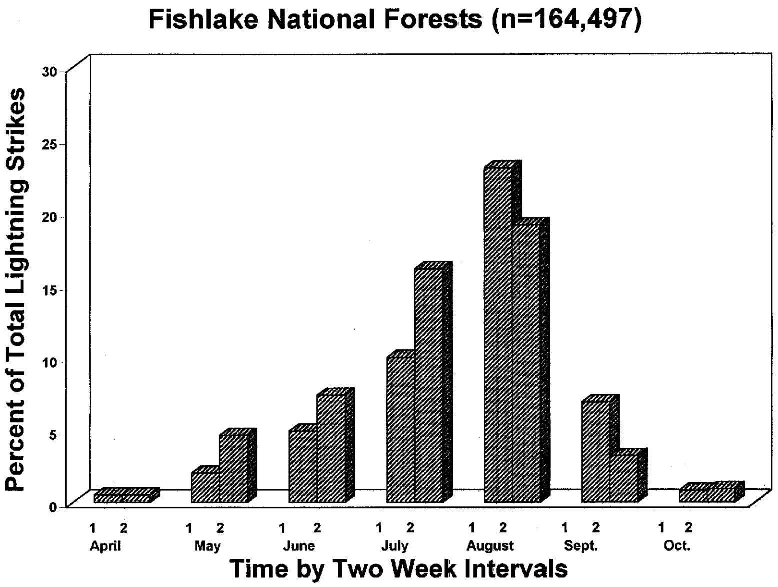

Figure 3—Frequency distribution of lightning strikes on the Fishlake National Forest in southcentral Utah. When aspen is normally dry enough to burn during early spring and late in the fall, there are few lightning strikes. This is true throughout western North America (Kay 1997a, 1997b; Kay and others 1999). Lightning data (1985-1994) from the Bureau of Land Management's Automatic Lightning Strike Detection System, Boise, ID as provided by the Fishlake National Forest, Richfield, UT.

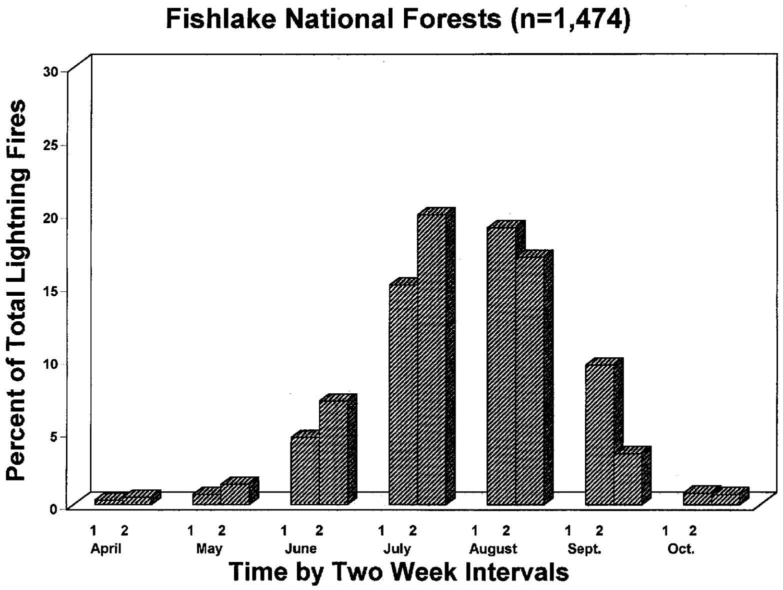

Figure 4—Frequency distribution of lightning-caused fires on the Fishlake National Forest in southcentral Utah. When aspen is normally dry enough to burn during early spring and late in the fall, there are few lightning strikes (Figure 3), and few lightning-started fires. Although, there are virtually no lightning fires capable of burning aspen, historical photographs and fire-history data indicate that aspen burn frequently in the past (Bartos and Campbell 1998). This suggests that the earlier fires had to have been set by native people. Forest fire data (1960-1996) from the Fishlake National Forest, Richfield, UT.

aboriginal burning has been widely reported in the anthropological literature (e.g., Boyd 1986; Gottesfeld 1994; Lewis 1985; Pyne 1993, 1995; Turner 1991), those findings have largely been ignored by ecologists and modern land mangers (Bonnicksen 2000, Kay 1995).

Determining how fires started, though, is critical because “fires set by hunter-gatherers differ from [lightning] fires in terms of seasonality, frequency, intensity, and ignition patterns” (Lewis 1985:75). The majority of aboriginal fires were set in the spring, between snowmelt and vegetation greenup, or late in the fall when burning conditions were not severe (Pyne 1995). Unlike lightning fires, which tend to be infrequent and of high intensity, native burning produced a high frequency of low-intensity fires. Aboriginal burning and lightning fires created different vegetation mosaics, and in many instances, entirely different plant communities (Blackburn and Anderson 1993, Bonnicksen 2000). Moreover, aboriginal burning reduced or eliminated the number of high intensity, lightning-generated fires (Pyne 1993, 1995; Reid 1987). Once aboriginal fires opened up the vegetation, then subsequent lightning fires behaved like those set by Native Americans (Pyne 1993, 1995).

Aboriginal Burning in the East

There are several lines of evidence which indicate that aboriginal burning was also common in eastern deciduous forests (Bonnicksen 2000, Hamel and Buckner 1998). These include the lack of lightning fire, the original structure of the forests, and species composition changes that have occurred since European settlement, among others.

Lightning Fires

Although lightning is common in most eastern forests, lightning-started fires are rare (Barden and Woods 1974, 1976; Bratton and Meier 1998; Harmon 1982; McCarthy 1923). This occurs because when lightning strikes are most frequent during July and August, eastern deciduous forests are too green to burn. Like western aspen communities, eastern deciduous forests will readily burn only when the trees are leafless and the understories dry — conditions that generally occur only early in the spring or late in the fall, and during both those periods there are few lightning strikes and even fewer lightning-caused fires. Fire history studies, however, have shown that prior to European settlement, fires were common in the eastern U.S. — many more than can be accounted for by lightning alone (Bonnicksen 2000:259-269, Bratton and Meier 1998). Therefore, the only logical conclusion is that burning by native people was once widespread in many eastern forests, similar to conditions in the West.

Forest Structure

At European contact, many eastern forests were open and park-like, with little undergrowth (Bonnicksen 2000, Day 1953, Olsen 1996). Like ponderosa pine forests in the West, most eastern forests were once composed of large, widely-spaced trees "so free of underbrush that one could drive a horse and carriage through the woods" (Botkin 1990:51). Like western forests, though, most eastern deciduous forests are now choked with dense underbrush and smaller regenerating trees. The only way to create open park-like stands in either western or eastern forests is for those areas to have been subjected to a high frequency of low-intensity surface fires. In eastern deciduous forests this would have required fires either early in the year before the trees leafed-out or in late autumn after leaf-fall. During both those periods, though, lightning-caused fires seldom occur. Thus, the only way for eastern forests to have displayed the open-stand characteristics that were common at European settlement is if those communities had regularly been burned by native people as part of aboriginal land management activities.

Species Composition Changes

An even more compelling piece of evidence is the species composition changes that have occurred in eastern forests since European colonization (Bonnicksen 2000). For the last 3,000-4,000 years, or longer, much of the eastern United States was dominated by oak (*Quercus* spp.), American chestnut (*Castanea dentata*) and pines (*Pinus* spp.), all fire-tolerant, early successional species (Bonnicksen 2000; Bratton and Meier 1998; Clark and Royall 1995; Cowell 1995, 1998; Delcourt and Delcourt 1997, 1998, 2000; Delcourt and others 1986, 1998; Hamel and Buckner 1998; Meyers and Peroni 1983; Olson 1996). Since European contact, however, oaks and pines have been replaced by late-successional, fire-sensitive species, such as maples (*Acer* spp.) (Abrams 1998, Bonnicksen 2000, Botkin 1990:51-71). That is to say, the species composition of many eastern forests had been maintained for thousands of years by frequent fires — fires, as we have seen, which could only have been set by native people. It is equally clear that aboriginal burning created the many eastern prairies and "barrens" reported by early Europeans (Barden 1997, Belue 1996, Bonnicksen 2000, Campbell and others 1991). Canebrakes (*Arundinaria gigantea*) too likely owed their existence to native burning and other aboriginal land management practices (Platt and Brantley 1997).

In the absence of aboriginal burning, massive vegetation changes have also occurred throughout the West (Bonnicksen 2000). Millions of acres of aspen have been lost to invading, fire-sensitive conifers (Bartos and Campbell 1998, Kay 1997a). Similarly, vast acreages of ponderosa pine and Douglas fir (*Pseudotsuga menziesii*) forests are being replaced by more fire-sensitive, and more shade tolerant conifers. In many areas of the West, native grasslands have declined precipitously as fire-sensitive, woody species have increased (e.g., Miller and Rose 1995). Oak dominated communities in California and Oregon have also been severely reduced by invading conifers, until today, oak woodlands are one of the most endangered habitat types in the West. All because native people no longer employ fire to manage the land (Bonnicksen 2000).

Aboriginal Populations

In addition, aboriginal populations were much larger than commonly believed (Hamel and Buckner 1998). Until recently, it was thought that only about two million natives inhabited North America prior to the arrival of Columbus (Stannard 1989, 1992). Dobyns (1983), however, postulated that native people, who were attempting to escape Spanish exploitation in Cuba, fled to Florida in ocean-going canoes and brought European-introduced smallpox with them to the mainland during the early 1500s. This and other diseases, to which aboriginal inhabitants had no immunological resistance, then ravaged native people, reducing aboriginal populations by >90% before the Pilgrims set-foot at Plymouth Rock. Subsequently, Ramenofsky (1987), Smith (1987), and Campbell (1990) tested Dobyns' hypothesis using the archaeological record and concluded that a major collapse of native populations occurred ca. 1550-1600 throughout North America — 100 to 200 years prior to direct contact of Europeans with native people in many areas; i.e., European diseases were transmitted from native group to native group across all of North America — termed pandemics.

Based on this and other evidence, it is now believed that in 1492 there may have been as many as 100 million native people in North America with perhaps an even larger number in South America (Stannard 1992). Although Dobyns' hypothesis is still debated (Snow 1995), in general, estimates of pre-European native populations have steadily been revised upward. Needless to say, prior to European influence there were more than enough native people to have structured vegetation communities throughout North America (Bonnicksen 2000). This is especially true in the eastern United States where agriculture supported high densities of native people (DeVivo 1991, Hamel and Buckner 1998, Smith 1987).

There was no “wilderness.” There were no unnamed streams, there were no unnamed mountains. The idea that North America was a “wilderness” untouched by the hand of man before 1492 is a myth, a myth created, in part, to justify appropriation of aboriginal lands and the genocide that befell native peoples (Cronon 1995, Denevan 1992, Gomez-Pompa and Kaus 1992, Simms 1992). Moreover, there is no evidence that native people ever purposefully limited their populations to avoid environmental impacts or that, if they tried, they were successful (Blurton-Jones 1986, 1987; Cohen 1977, 1989; Smith and Winterhalder 1992).

Conclusions

Prior to European discovery of the New World, aboriginal use of fire was widespread in both western and eastern forests. In fact, the Americas, as first seen by Europeans, had largely been crafted by native people, not created by nature (Bonnicksen 2000, Kay 1998). Thus, the only way to preserve original vegetation conditions in parks and other natural areas is for modern land managers to reinstitute historical burning regimes. A hands-off or “natural-regulation” approach by today’s land managers will not duplicate the ecological conditions under which eastern deciduous forests developed. Instead, letting-nature-take-its-course creates highly unnatural conditions that have never before existed in eastern or western forests (Bonnicksen 2000, Hamel and Buckner 1998). Unless the importance of aboriginal burning is recognized, and modern management practices changed accordingly, our ecosystems will continue to lose the biological diversity and ecological integrity they once had even in parks and other protected areas (Kay 1998).

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Humans, Topography, and Wildland Fire: The Ingredients for Long-term Patterns in Ecosystems

Richard P. Guyette[1] *and Daniel C. Dey*[2]

Abstract

Three factors, human population density, topography, and culture interact to create temporal and spatial differences in the frequency of fire at the landscape level. These factors can be quantitatively related to fire frequency. The fire model can be used to reconstruct historic and to predict future frequency of fire in ecosystems, as well as to identify long-term changes in anthropogenic fire regimes. Topographic roughness is positively related by a regression equation to the length of mean fire intervals indicating that fires are less frequent in rough than in flat terrain during periods of low human population density. The strength and direction of this relationship diminishes as the frequency of anthropogenic ignitions increases to the point that the fuel environment is pyro-saturated. Human population density is a master variable in understanding anthropogenic fire regimes and topographic effects. The interactions of these factors through time creates at least two stages in anthropogenic fire regimes: an Ignition Limited Stage in which fire frequency is function of human population density, and a Fuel Limited Stage during which fire frequency is limited by fuel production and is independent of increases in human population density.

Introduction

Human-Landscape Interactions Affect Fire Regimes

Humans are by far the most important factors influencing the frequency of fire ignitions in the Missouri Ozarks and elsewhere throughout much of eastern North America. Most fires occur during the fall, winter, and spring in years of normal precipitation. Dry, warm weather of only a few days is sufficient to cure fine fuels and permit the spread of surface fires, especially in the spring and fall when dead ground vegetation and leaf litter are most abundant and exposed to the sun. Fires during the growing season are rare but do occur during very hot and dry summers. High fuel moisture content and precipitation in summer limit fire ignitions and spread. Despite an abundance of thunderstorms in Missouri (50-70 thunderstorm days per year) (Baldwin 1973), natural ignitions are rare (Schroeder and Buck, 1970). Most thunderstorms occur in the spring and summer and are usually accompanied by heavy rainfall. Fire statistics for Missouri (Westin 1992) indicate that less than 1 percent of fires are caused by lightning, all other causes are human-related.

Topography and the fuel environment also are important variables that determine the fire regime by controlling fire spread, intensity and extent (Whelan 1995, DeBano and others 1998). Human culture and land use directly affect the continuity and nature of the fuel environment. We present a quantitative approach to modeling the interaction of humans with landscapes and their combined effects on fire regimes. Three factors, human population, topographic roughness, and culture, interact to create temporal and spatial differences in the frequency of fire at the landscape level.

Succession in Anthropogenic Fire Regimes

Through time, fire regimes undergo a number of changes based upon population density, topography, and culture. A progression of four stages in anthropogenic fire regimes can be identified in the upper Current River watershed fire record. These include: 1) an Ignition Limited stage, 2) a Fuel Limited stage, 3) a Fuel Fragmentation stage and, 4) a Culturally Limited stage. During Stage 1, fire frequency increases as the human population approaches 0.64 humans/km^2 and the number of potential ignitions increases. Fire frequency is a logarithm function of human population density. In later stages, as human population density increases and the landscape is saturated with ignitions, other factors limit fire frequency such as fuel loading and continuity, cultural attitudes toward fire, and human valuation of natural resources. In the Current River watershed, the second stage, a Fuel Limited Stage, is characterized by limits imposed on the frequency of fire frequency by the primary production of fuels. During this stage, human population density exceeds a threshold of 0.64 humans/km^2 and increases in population density and ignitions do not result in increased fire frequency.

This paper will integrate population density and topographic roughness in an empirically derived model that can be used to reconstruct past or to predict future frequency of fire in ecosystems at specific sites. Here, we focus on the development of an empirically based model of fire frequency from over 2,500 tree-ring dated fire scars at 29 sites in the Current River watershed in southeastern Missouri. This data set is used to develop and verify a regression equation that predicts fire frequency based on topographic and population variables. We also examine the interactions of humans and landscapes with respect to fire regimes, and illustrate, using the fire model, the spatial and temporal variability in the anthropogenic fire regime in the upper Current River landscape.

Rational and Use for Fire Regime Models

Quantitative models of fire regimes can be used in many ways. Past fire regimes can be reconstructed for natural areas that have no on-site fire history information. The equations can be used in conjunction with soil, geology, and species data to reconstruct past and potential flora and

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[1] Research Associate Professor, The School of Natural Resources, 203 ABNR Building, University of Missouri, Columbia MO 65211. email: guyetter@missouri.edu

[2] Research Forester, USDA Forest Service, 202 ABNR Bldg. Columbia MO 65211. email: ddey@fs.fed.us

faunal mosaics. Thus, land managers interested in returning small isolated parcels of land, natural areas, or whole ecosystems to pre-European conditions using prescribed fire can use this model to reconstruct the fire regime. Researchers can also use the equation to create a GIS overlay of a fire regime for use with research plots or to make inferences about future fire regimes based on changes in population and climate, and landscape variables. Perhaps the most important aspect of fire regime models and fire history however, may not be the information provided on fire regimes, but the perspective provided on the long-term interactions between humans and their environment.

The Historical Context of Human Populations, Cultures, and Fire in the Current River Watershed

All of the historic changes in human populations due to war, migration, and disease are probably reflected in the fire scar record. Europeans introduced diseases decimated Native American population. War among peoples caused migration and genocide. Depopulation caused by disease (Dobyns 1983; Ramenofsky 1987), warfare, and migration occurred throughout much of North America (DeVivo 1991) and has been linked with abrupt changes in fire history (Guyette and Dey 1995a; Dey and Guyette in press). Indeed, often the highly variable nature of human populations and culture is reflected in the extreme variability of the fire record within a site (Guyette and Dey 1995b, Cutter and Guyette 1994, Guyette and McGinnes 1982).

Anthropogenic ignitions were probably frequent along the rich fertile bottom lands of large rivers including the Current and Jacks Fork Rivers. About 1000 A.D. (Lynott 1989, Price and others 1983) during the Mississippian Cultural Phase, agricultural communities were located in riparian areas in the Current River watershed, and they would have been a source of many ignitions. With the end of this phase (about 1350 A.D.) began an era of limited use of the watershed by humans from downstream reaches of the river system (Price and others 1976). An early source of anthropogenic ignitions may have been the 6,000 Quapaw (Baird 1980) that lived to the southeast of the Current River in Arkansas near the confluence of the White, Arkansas, and Mississippi Rivers before 1680. Disease came first to large riverine cultures such as the Quapaw because of their location and population density (Dobyns 1983), and reduced the Quapaw population by two thirds in 1698 and again reduced their population in 1747 and 1751 (Baird 1980). There were only about 700 Quapaw by 1763. Coincident with a low percentage of sites burned circa 1750 in the Current River watershed (Guyette and Cutter 1997) were epidemic disease in 1747 and 1751 which reduced the Quapaw population.

The Osage people frequented the study site about the time of first European contact and probably provided many ignitions during the 1700s (Guyette and Cutter 1991). Banks (1978) commented that there were about 6,000 Osage circa 1800 in and around the Ozarks. About 1,200 Great Osage lived on the Osage River 280 km west of the Current River (Marriott 1974). A change in the territorial boundaries of the Osage from the late 1600s to about 1803 (Bailey 1973, Wiegers 1985) is coincident with long-term trends in fire frequency in the Current River watershed (Guyette and Cutter 1997). Territorial expansion by the Osage from 1673 to 1770 probably resulted from many factors, one of which included the acquisition of equestrian technology from aboriginal trade with western tribes. The horse gave the Osage new range and mobility to hunt, exploit, and culture (by fire) areas such as the Current River watershed which were distant from their territorial focus. Wiegers (1985) estimated that the Osage acquired horses as early as 1680 while Waldman (1985) sets the date circa 1719. They hunted in the east for bears (perhaps in the Current River watershed) and in the west for buffalo (Chapman and Chapman 1972). Stevens (1991) reported that the Osage moved south and east toward the Current River on extended spring hunts coincident with Missouri’s largest fire season (Westin 1992).

The Cherokee were one of several tribes that migrated into the Ozarks after being displaced from their eastern lands by Euro-Americans (Gilbert 1996). They probably continued the traditional burning practices they had used on their ancestral lands. About 6,000 Cherokee (Gilbert 1996) may have been living in southeast Missouri and northeast Arkansas at the time (1803) of the Louisiana Purchase. In the late 1770s, many of the Cherokee settled to hunt and farm along the St. Francis River, which lies about 60 km east of the Current River. The Osage made war upon the Cherokee (Banks 1978) as the Cherokee infringed upon their hunting grounds. Conflict may have led to wildland burning. A general increase in the percentage of sites burned annually (Guyette and Cutter 1997) in the Current River watershed occurred from 1760 (9 percent) to 1820 (30 percent) coincidently with the migration of the Cherokee (Gilbert 1996) and other eastern tribes.

The Delaware had a tradition of wildland burning before migrating to Missouri from Ohio (Whitney 1994). The Delaware and Shawnee entered Missouri at Cape Girardeau and passed through the Current River watershed on their way west in the late 1700s and early 1800s (Stevens 1991). One estimate of Shawnee and Delaware population west of the Mississippi in 1812 was 400 (Marriott 1974). In 1816, Banks (1978) stated that there were about 840 Delaware and 1,300 Shawnee in all of Missouri. In 1824 there were 1,383 Shawnee in Missouri (Howard 1981). Delaware hunted, lived, and traded in the upper Current River area from about 1815 to 1822. In November 1820, 1,346 Delaware crossed the Mississippi River and made an emergency encampment on the Jacks Fork of the Current River (Weslager 1978). However, they stayed a number of years and this was associated with increased fire frequency (Guyette and Dey 1997). Most of the Delaware and Shawnee had left the Current River watershed by the 1830s and Euro-Americans began to occupy the area.

Old-stock Euro-Americans (Gerlach 1986) from the southeastern U.S., mainly Tennessee, began settling in the area circa 1820 and continued their tradition of wildland burning. Later, circa 1860, Scotch-Irish immigrated into the area. The Current River watershed, like many other forested areas in the eastern United States (Sutherland 1997), experienced an increase in fire frequency with settlement by Euro-Americans. In the Current River watershed the percent of sites burned increased with population density during the 1810 to 1850 period and the average percent of sites burned in a year nearly doubled from 20 percent in 1810 to 39 percent in 1850 (Guyette and Cutter 1997).

Methods

Fire Scars

Fire scars on survivor trees form the data base of this work. In anthropogenic fire regimes, fire scars can be thought of as ecological artifacts. They are objects made or modified by humans. Humans set wildland fires and wounded these woody plants as they attempted to manipulate vegetation with fire for a variety of purposes. As ecological artifacts, they point out the relationships between humans, organisms, and forests. Like artifacts in an archeological context, they can be dated in time and given a location in space. As such, they reflect many human characteristics such as population density and culture that can be used in an anthropological context. The dates and locations of these ecological artifacts are used in this work to examine the relationships between humans, fire, and landscape over a period of extraordinary change and diversity in human history.

The regression equation for reconstructing historic fire frequency is derived from data on fire frequency, topographic roughness, and human population density. The mean fire intervals (MFI) used in the regression equation were derived from fire scar data for three time periods (1700 to 1780, 1781 to 1820,1821 to 1850) at the 29 sites. Much of the fire history information that follows is compiled from more than 2,500 fire scars on shortleaf pine (*Pinus echinata* Mill.) stumps, trees, and remnants in the Current River watershed (Guyette and Cutter 1997, Batek and others 1999). Dendrochronological methods were used to date the fire scars found on over 166 shortleaf pine which are used as the basis for the model and following discussion. The mean fire intervals (MFI) in this study are thought to reflect fire frequency during ecosystem development and a range of human population densities that has occurred over at least the last millennium in the Ozarks. Here, we present the two most significant variables in an equation that characterizes the effects of human population density and topographic roughness on the frequency of wildland fire.

Human Population Estimates

A spatial and a temporal component of human population are combined to reconstruct human population density values for use in the equation. The spatial component reflects the tendency in the watershed for greater population densities and therefore ignitions to occur downstream. We use the distance upstream of a site from Hawes, on the Current River, as a reference variable we call River Mile to model this tendency. Many archeological and historic Native American sites are located in the larger and more fertile bottom lands in the downstream reaches of the Current River (Stevens 1991). For instance, a major Osage trade route to St. Louis intersected the lower Current River. The abundance of these sites decreases significantly (r=-0.53, p=0.03) with their distance upstream. The Mississippian cultural phase (O'Brien and Wood 1998) persisted downstream on the southeast border of the Ozarks until as late as 1700 (Price et al. 1976). Quapaw lived downstream and southeast of the watershed (Rafferty 1981). Later, the density of human built structures in the Current River watershed during the mid 1800s diminishes with their distance upstream (r=-0.69, p=0.002). Even today the population of towns along the Current River and Jacks Fork Rivers decreases as one moves upstream (r=-0.98, p=0.02). Perhaps fertile soils of the lower reaches of the river produces more fuel and forage, which in turn attract more game, hunters, gatherers, and agriculturalist. These are relationships that may persist, to some extent, through time and cultures. Alternatively, fertile soils may be correlated with landscape level fuel and fire dynamics, or land form gradients such as topographic roughness, that may affect the ignition, spread and consequently the frequency of fire.

We derived the temporal component of human population density in the study area from population numbers and trends as described in historical accounts and analyses as reviewed in the introduction. Fire frequency has been found to be strongly associated with human population density (Guyette and Dey 1997, Dey and Guyette in press, Guyette and Cutter 1997). The density of each group in the Current River watershed was calculated by dividing their total population by the area of a circle whose radius was the distance between the population center and the study site. This component of human population density does not take into account the great spatial variation of population density within an area and only reflects changes in population density over time at study sites as a whole.

An Index of Topographic Roughness

The highly dissected topography of the Current River watershed has been shown (Guyette 1995) to affect the spread and frequency of fires during periods of low human population density when anthropogenic ignitions were limited in number. Roughness can slow a low intensity fire by decreasing the rate of spread as fires burn down steep slopes, fuel continuity is broken by creeks, rivers and rocky outcrops, and changes in aspect influence fuel moisture. Topographic roughness is estimated by measuring the surface area of the earth with measuring surfaces of different sizes (Figure 1). Indices of topographic roughness are developed by comparing surface area measurements made with two different sized scales. A circle, 5,000 meters in diameter, is marked on a digital elevation map. The surface area of the earth enclosed by this circle is calculated from pixels that are 30 meters on a side. Their slope and a trigonometric conversion are used to estimate the area of the uneven land surface (Krstansky and Nigh 1999). The pixels are summed to estimate the surface area of the landscape enclosed by the circle. This measure is then divided by the planimetric surface area (the large scale in this case) of the circle that is 5,000 meters in diameter. This ratio of the actual

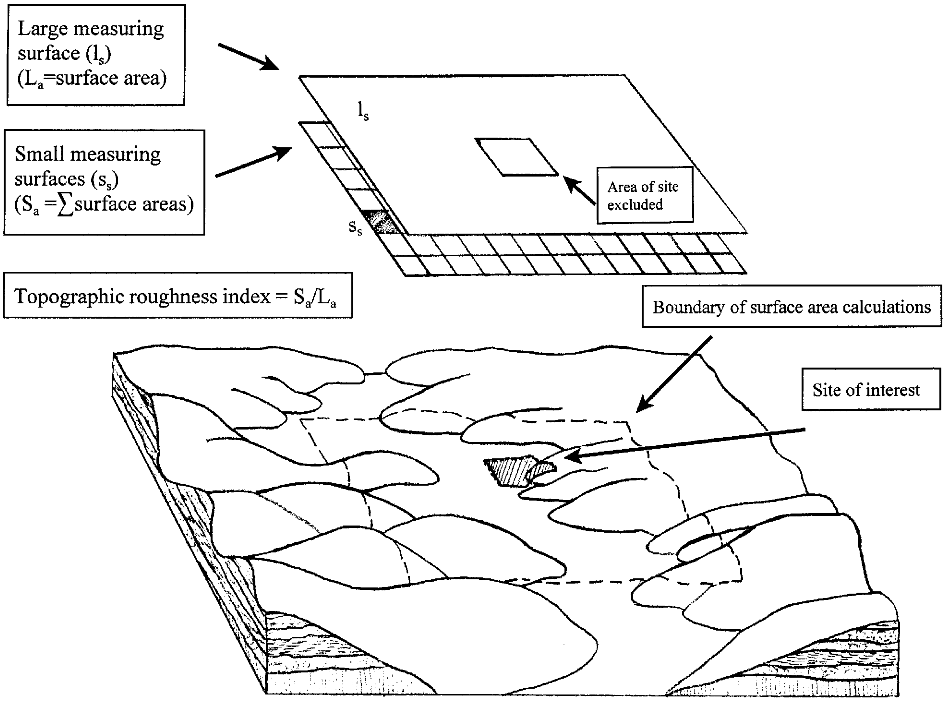

Figure 1.—Topographic roughness is an important variable which interacts with human population density to control the frequency of fire. Topographic roughness can be estimated with an Index of Topographic Roughness. The illustrated example of an Index of Topographic Roughness is calculated by measuring the circumscribed region of the earth of a given dimension and shape with a large measuring surface (l_s) and many small measuring surfaces (s_s). The sum of the areas (S_a) of the small measuring surfaces is then divided by the area (L_a) of the large measuring surface to obtain the Index of Topographic Roughness.

surface area to the planimetric surface area is the Index of Topographic Roughness.

Results and Discussion

Modeling Fire Frequency

We use the Index of Topographic Roughness and estimates of human population density to model the frequency of fire at the landscape level. Central to this model are regression equations derived from and tested with independent fire scar data. The regression used to reconstruct past fire frequency at the study sites for the period 1700 to 1850 is:

$$MFI = -442.1 + (449.9 \times topo) - (0.001 \times pop), \qquad \text{(Equation 1)}$$

where: MFI= mean fire interval
topo = Index of Topographic Roughness for a circle with a 5,000 meter diameter
pop = natural log of human population density times the square of river mile,
n = 78 (three time periods, 29 sites),
$r^2 = 0.51$,
all variables and intercept are significant at $P<0.01$.

This regression explains about half of the variability in fire frequency in time and space. The amount of unexplained variability is not surprising considering the large role chance plays in a complex and highly variable phenomena like wildland fire. In addition, there are methodological problems with the fire scar record such as the number of sample trees at a site, their location, the size of the site, and the hit and miss nature of the scarring process that create errors with respect to the actual spatial and temporal distribution of fire events in the data set. None-the-less, the temporal and spatial depth of the fire scar record allow for the identification of the significant independent variables in this equation. Although the framework of this equation may be applicable in many ecosystems, the calibration of these variables will differ greatly in different ecosystems. Differences in fuels, climate, topographic roughness, and other landscape variables such as the abundance of lakes would require a new calibration.

Human Population Density and Topographic Roughness in Fire Regimes

Anthropogenic fire regimes are the result of complex interactions between human population density and an number of factors, such as culture, fuels, and landscape. Topographic roughness is one of the most important and temporally persistent landscape variables. At low population densities, fire frequency increases as population density does as a result of increasing anthropogenic ignitions (Figure

2) and topographic roughness is an important factor mitigating the frequency of fire (Figure 3). At higher population densities fire frequency may be fuel dependent or diminish as landscape artifacts (i.e., fuel continuity or mosaic that result from human land use), culture, and changes in human economics (ex. real estate, forage, timber) inhibit the ignition and propagation of fire over the landscape. The effects of topographic roughness on fire frequency change with time and human population (Figure 4). Topographic controls on the frequency of fire become less and less important as population density and the frequency of anthropogenic ignitions increases. Just before the era of fire suppression (Figure 4), as population density and agricultural activity become greater, burning increases in topographically rough areas relative to topographically smooth areas as humans attempt to culture vegetation in areas of marginal agricultural productivity. A rule-of-thumb concerning fire history is that at low population densities topographically smooth areas (often prairies and grass lands) burned more often than topographically rough areas (often forested lands). As population density increases, topographically smooth areas burn less often and topographically rough areas burn more often.

Pyro-saturation is a stage in a fire regime when fuels are burned as soon as they accumulate enough to carry a fire. Inorganic decomposition (fire) plays a much greater role than organic decomposition (decay) in controlling the oxidation of biomass. Ecologically, the resistence of an ecosystem to pyro-saturation is a function of human population density and topographic roughness. The resistence of some ecosystems to pyro-saturation from anthropogenic ignitions is very low. For instance, it might take only a handful of humans on horseback or foot to keep a topographically smooth ecosystem, such as a large prairie of tens of thousands of square kilometers, in a state of pyro-saturation. On the other hand, many topographically rough ecosystems, such as the forests of the Central Hardwoods Region, require a relatively high human population density to reach and maintain pyro-saturation. The quantitative relationships expressed in Equation 1 allow us to estimate that in the Current River watershed it takes about 0.64 humans per square kilometer to reach a state of pyro-saturation.

Implications for Landscape Fire Regime Diversity

The fire model (Equation 1) was used to predict the mean fire interval for points throughout the upper Current River watershed (Figure 5). The value to this approach to studying landscape-level fire regimes is that extrapolation of single fire histories of limited spatial and temporal dimensions often underestimate the actual variability within an ecosystem. In contrast, a landscape fire model can represent the diversity of fire regimes as they change across the landscape with variations in topographic roughness and human

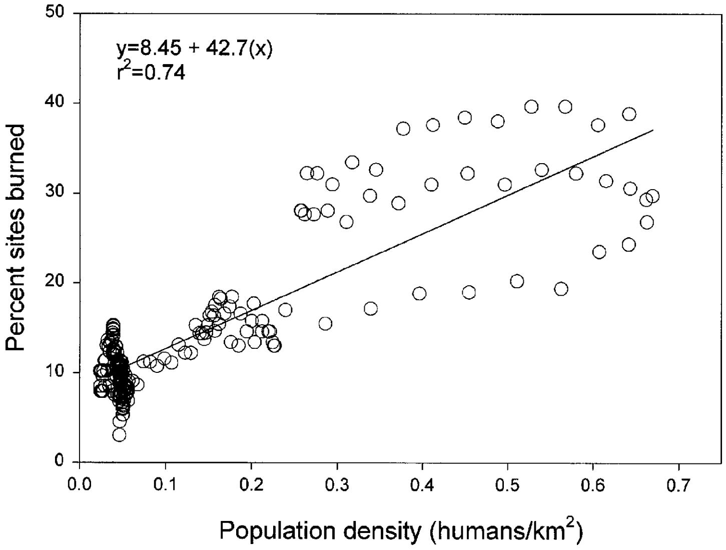

Figure 2.—Human population density is the major factor in the frequency of wildland fire ignitions in many regions. This scatter plot and regression illustrate the change in the percentage of study sites burned each year with human population density. The data is for the period 1620 to 1850 and is from the Current River watershed in Missouri. The percentage of sites burned (y axis) is from annual data that has been smoothed with an 11 year moving average (Guyette and Cutter 1997).

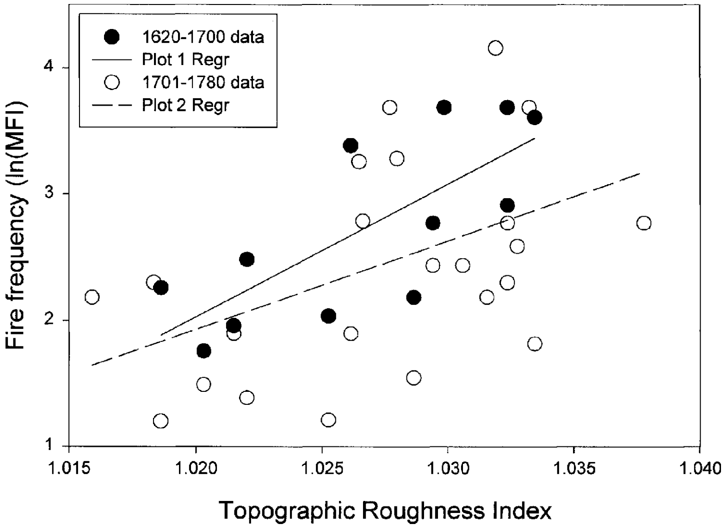

Figure 3.—These two scatter plots and regression lines illustrate the mitigation of fire frequency (MFI) by topographic roughness during two periods of low human population density. Topographic roughness is hypothesized to have reduced the frequency of fire by inhibiting the ignition and spread of anthropogenic fires.

population. The diversity of mean fire intervals present at the landscape level between 1700 and 1780, a period of low population density, may best reflect the fire regime as it was for many centuries before European settlement.

The diversity of fire frequency at specific locations in the Current River watershed is much greater at low population densities than at high population densities as the spread of fires from a limited number of anthropogenic ignitions is mitigated by topographic roughness. The spatial diversity of fire frequency within an ecosystem that results from topographic roughness persists through time and can result in strong species associations. Long-term topographic control of fire regimes can create refugia for fire sensitive species in topographically rough areas as well as creating refugia in topographically smooth areas for fire dependent species. These refugia may provide valuable sources for re-colonization of disturbed landscapes.

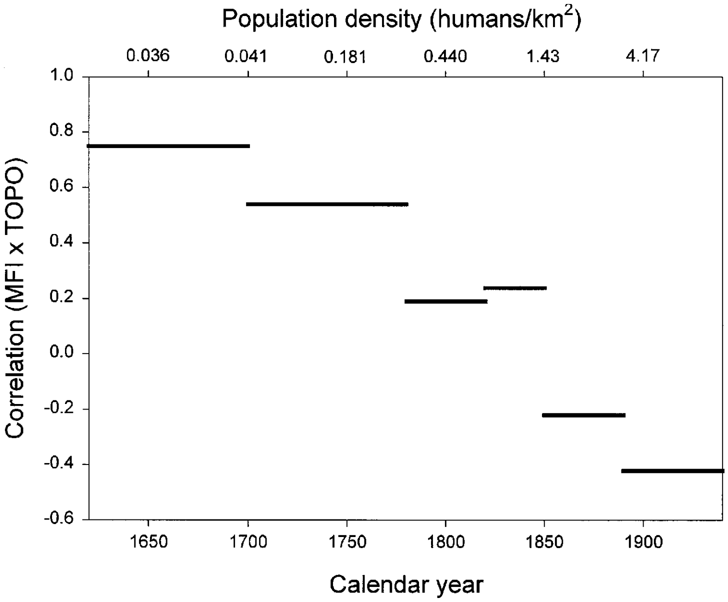

Figure 4.—This graph illustrates how human population density can mitigate the effects of topographic roughness on the frequency of fire. The y axis is the correlation coefficient between fire frequency (MFI) and the Index of Topographic Roughness. The upper x axis shows the mean population density of the period of correlation. The bars indicate the level, sign, and period of the correlation calculation. Correlations for the two earliest and the latest periods are significant ($p<0.05$).

Conclusions

Fire frequency varies spatially and temporally at the landscape-level largely due to changes in topography, human population, and culture. Population density and topographic roughness are master variables in understanding anthropogenic fire regimes that can be quantified and used to model the variability within a fire regime. Although landscape level fire models require a considerable investment in the acquisition of fire scar, site, landscape, and population data, they enable an understanding of long-term fire regimes that can be used in many ways.

Our fire model can be used to estimate the frequency of fire at any or all points in a landscape. Fire frequencies and regimes can be calculated for natural areas, parks, and forests that have no fire scar, historic, or charcoal data base for estimating a fire regime. Future predictions of fire frequency can be made by incorporating changing human population density, values, climate, and fuels into the model. The model enables the mapping of variation within a fire regime in both time and space. Maps of fire regimes and estimates of the long-term frequency of fire can be used in research on flora and fauna, in setting the frequency of fire in prescriptions to restore or maintain fire-dependent ecosystems, and the development of silvicultural prescriptions that mimic “natural” disturbances. Maps of fire regimes and the long-term frequency of fire disturbances can also be used to identify areas in the landscape that are refugia for either fire-dependent or fire-sensitive species.

Anthropogenic fire regimes, composed of complex interactions among fire, humans, fuels, and topography, have altered a continuum of ecosystems in North America for many thousands of years. These fire regimes are dynamic and have a number of identifiable successional stages that change as humans populate and transform their environment by fire and artifact. Human-induced changes in a fire regime are modified by topography and fuels. Stages in anthropogenic fire regimes are marked by changes in factors which limit the frequency of fire such as the frequency of ignitions, the production of fuels, the propagation of fire, and changing cultural values.

Acknowledgments

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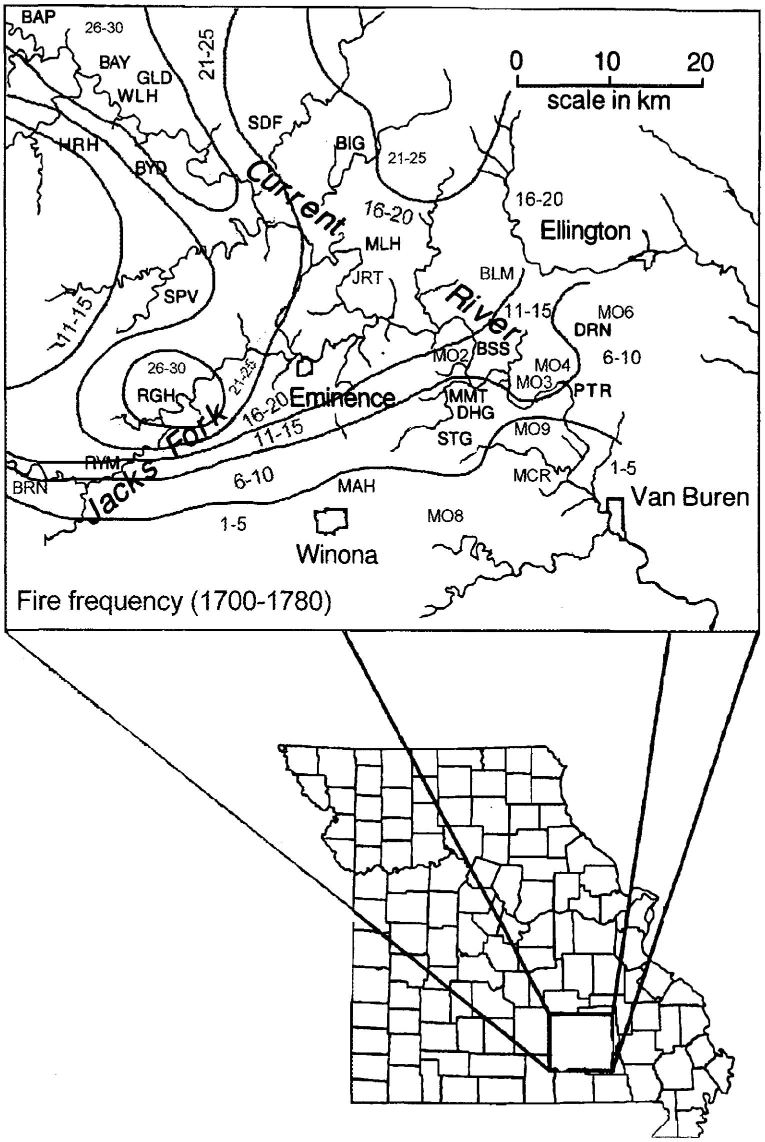

Figure 5.—This map illustrates the diversity of mean fire intervals present at the landscape level during a period (1700 to 1780) of low to moderate population density, which best reflects the fire regime many centuries before European settlement. Fire frequency gradients are represented by the iso-pyro lines. Iso-pyro lines define a gradient of equal fire frequencies. Numbers adjacent to iso-pyro lines give the class of mean fire intervals defined by each line.

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Fire on the Edge: Prehistoric Fire Along the Escarpment Zone of the Cumberland Plateau

Cecil R. Ison[1]

Abstract

Unlike many areas of the United States, anthropogenic fires are the prime agent for affecting changes in plant and animal species composition in the southern Appalachian Highlands. Although the extensive use of fire by the American Indians has been recognized from the earliest European observers, it is somewhat difficult to determine the impact prehistoric fires had on forest structure. By examining the fossil and charcoal record from Cliff Palace Pond with the archaeological record recovered from nearby prehistoric sites, a 9,500-year record of the vegetational development can be established for the escarpment zone of Eastern Kentucky. This record indicates that anthropogenic fires played a central role in shaping the forest structure, especially after the transition from a hunting and gathering economy to one based on swidden agricultural practices.

Introduction

Most federal land managing agencies are now committed to using ecosystem-based approaches in their land management objectives. Unfortunately, until recently, these approaches were primarily concerned with the physical and the biological components of the ecosystem and usually lacked a comprehensive understanding of the human dimension within the environment. Yet the management of federal lands is not set by nature, but by people (Williams 1993). Just as people shape the management direction of ecosystems today so has it been since the first Paleoindians set foot on the New World. Here in the eastern woodlands, people have been a part of ecosystems since the end of the Pleistocene and have been shaped by, and in turn shaped ecosystems in a number of ways.

On the threshold of the 21st century we are striving as never before to bring a historical context into natural resource management. At long last both land managers and the public are beginning to accept that there is a historical relationship between people and nature, that all human history has a natural context and that we shape the land just as much as the land shapes us (MacCleery 1998). Wildland fires (both natural and those created by people) have long been considered to be an integral part of ecosystems across North America and that the recurring fire disturbances are essential to the functioning of these systems (Mutch 1995). While fire certainly played a role in shaping the forests of the southern Appalachians, the extent that anthropogenic fires played has been poorly understood.

Cumberland Plateau Escarpment Zone

In Kentucky, conglomeratic sandstones outcrop in a belt ranging from 8 to 40 km wide along the western edge of the Cumberland Plateau. Geologically, this area is referred to as the Pottsville Escarpment. Erosion within this belt has created deep narrow stream valleys flanked by steep-sided to precipitous valley walls. Little level land is available along the stream margins but the ridges, ringed by high sandstone cliffs are fairly wide and flat. Nearly 5000 kilometers of cliffline snake through the Daniel Boone National Forest, and often stretch for miles without a break. These natural features led Miller, in his Geology of Kentucky (1919) to refer to the escarpment zone as a “Chinese Wall” that greatly inhibited economic development of the area.

According to a tree census of eight “old forest” stands in the Lower Big Sandy River drainage, the dominant trees of Eastern Kentucky were several oak species (especially the white oak), beech, hickories, pines, maple and poplar (Shaler and Crandall 1876:12-13). Along the narrow stream valleys cane thickets was the dominant understory. Nuttall (1821:29) noted that the northern limit of cane was the southern bank of the Ohio River. It is from the archaeological sites and the natural ponds within this narrow escarpment zone that an understanding of prehistoric fire and its affects on forest structure are beginning to emerge.

Fire History

Wildland fire in the southern Appalachians results from two forces. One is natural, those caused by lightening and the other cultural, those which result from people. Martin (1990:56) reports that along the Cumberland Escarpment, lightening-caused fires account for only two percent of the total fires documented within a 50-year period. Here, the annual lightning fire occurrence is less than five fires per one million acres and usually occur after the month of May when the sap is up in woody plants. Consequentially, these fires usually lack the intensity necessary to affect plant species composition.

Since the natural fire regime was not sufficient to account for the fire-adapted ecosystems noted by early observers it was the anthropogenic use of fire that shaped the ecosystems of the eastern woodlands more so than any other force. Fire ecology has been practiced to various degrees for the last hundred years within the pine forests of the southeast. However, resource management agencies have been reluctant to use anthropogenic fires (prescribed fire) as a tool for managing the central hardwood forests. It has only been within the last decade that prescribed fire in hardwood forests has become accepted and this acceptance is far from universal.

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[1] Forest Archaeologist, USDA Forest Service, Daniel Boone National Forest, 1700 Bypass Road, Winchester, KY 40391

A baseline in the fire history of the Cumberland Plateau must begin with the historical literature. Although the pervasiveness of anthropogenic fire makes it a near-universal instrument for analyzing the interaction of people with their environments (Pyne 1994) without the written record as a guide, many of the material conditions observed in the archaeological record may not become apparent. Therefore we must rely on the observations of early writers to provide the foundation of aboriginal fire practices. Wherever early explorers journeyed they remarked on the Indians use fire. These ranged from burning the woods to clear land for cultivation to providing herbage and browse for the white-tailed deer. The deer were considered the cows of the eastern woodland Indians and the burnt woods their pastures (Whitney1994: 109). It has been estimated that this species alone supplied up to 90 percent of the animal protein consumed by the prehistoric inhabitants of eastern Kentucky (Wyss and Wyss 1977:24).

Merely 36 years had lapsed from Columbus' first sighting of the New World until Cabeza de Vaca recorded the first description of the Indian methods of burning the woods (Favata and Fernandez 1993:72). From de Vaca's first written account onward through the 19th century, the historic literature is full of descriptions of Indians burning the woods. Travelers repeatedly remarked on the wildfires they witnessed during the initial settlement of the Ohio Valley in the mid 1700's to early 1800's. In "Images of the Ohio Valley" historian John Jakle (1977) relates how early travelers marveled at the openness of the forest floor and its almost park-like appearance within the oak, hickory and chestnut communities. Fires according to Jakle, "provided the most spectacular forest scenes. Fires burned out of control each fall charring hundreds of thousands of square miles...over a distance of fifty miles the flames reflected on the clouds of smoke (1977:54)."

The earliest historical accounts of wildfire along the escarpment of the Cumberland Plateau were those of Dr. Thomas Walker, an explorer for the Loyal Land Company of Virginia. In 1750, Walker set out from Charlottesville, Virginia to inspect the western country to select the location of the company's 800,000 acre land grant. The first reference to forest fires by Walker was on May 17, 1750 in the vicinity of the divide between the waters of the Rockcastle and South Fork of the Kentucky River in what is now Jackson County, Kentucky. On May 17th he penned in his journal "The woods have been burnt some years past and are now very thick, the timber being almost kill'd" (Johnston 1898). Ironically, this is in the same vicinity of Cliff Palace Pond. The only other reference to wildfire that Walker made during his journey was again along the escarpment. On May 30th while traveling along the headwaters of the north Fork of the Red River Walker notes "The woods are burnt fresh about here and are the only fresh burnt woods we have seen these six weeks" (Johnston 1898).

Sometimes the clues of wildfires come from the most tragic accounts of the early Kentucky pioneers. The rescue party searching for the victims of the 1793 Easter Sunday attack on Morgan's Station for example, noted that two of the victims had been forced to march through a recently burned section of the woods. James Wade recounts that on April 2nd "…just above the head of Little Slate we found Mrs. Becraft and her suckling child 'six or eight months old' lying tomahawked. It was a very plain case. They had marched her that far in her shift, as was visible from the scratches and marks on it from a burnt wood they had passed through. And there she had given out" (Hogan 1991:24-25). This location is very near a large rockshelter known as Newt Cash Shelter, which would later provide the first archaeological evidence of prehistoric anthropogenic use of fire.

With the hundreds of accounts penned by early observers we have no problem accepting the purposeful use of fire by American Indians for a multitude of purposes during the historic era. In fact, it is generally agreed that Native Americans had a sophisticated knowledge of the use of fire (Patterson and Sassaman 1988). The problem arises when we try to provide time depth to the use of fire before the written record. After all, time depth is essential in the overall understanding of the role of fire in ecosystems. Despite this documentary evidence many still hold the view of the early pristine forest, one untouched by the hand of man, where native people lived in the forests without changing the ecosystem in any way. Where do we find the proof to counteract this view? It comes from unlikely sources. It comes from fire-scorched ancient trees, it comes from natural ponds whose sediments hold the record of nearby plant life from the first colonizers at the bottom to the current tenants at the top and it even comes from people's bowel movements preserved in the dry environments of caves and rockshelters over the millennia. But above all else, the proof comes from an interdisciplinary approach. It comes from the cooperative efforts of natural and heritage resource studies.

Archaeological Evidence

Archaeological investigations are essential for providing a diachronic context into the anthropogenic role of fire. In Kentucky, when the field of modern archaeological research was just beginning to become established in the late 1920's (Lewis 1996) early archaeologists realized the importance of the dry rockshelters and caves in recording the story of the human saga. Along the escarpment archaeologists recovered textiles, grass beds, desiccated human feces and other perishable artifacts from the dry rockshelter deposits. These materials provided a wealth of detail to our understanding of the lives of Kentucky's early inhabitants and their effect on landscape ecology. The following brief descriptions of the investigations at four of these rockshelters (figure 1) provide a tantalizing glimpse into anthropogenic fire behavior. The recent analysis of the Cliff Palace Pond sediments has provided a further dimension on fire history studies within the central hardwood region.

Newt Kash Shelter

The earliest clues for the prehistoric use of anthropogenic wildland fire in Kentucky began in the 1930's when Volney Jones, an ethno-botanist at the University of Michigan examined a small sample of vegetal remains recovered the

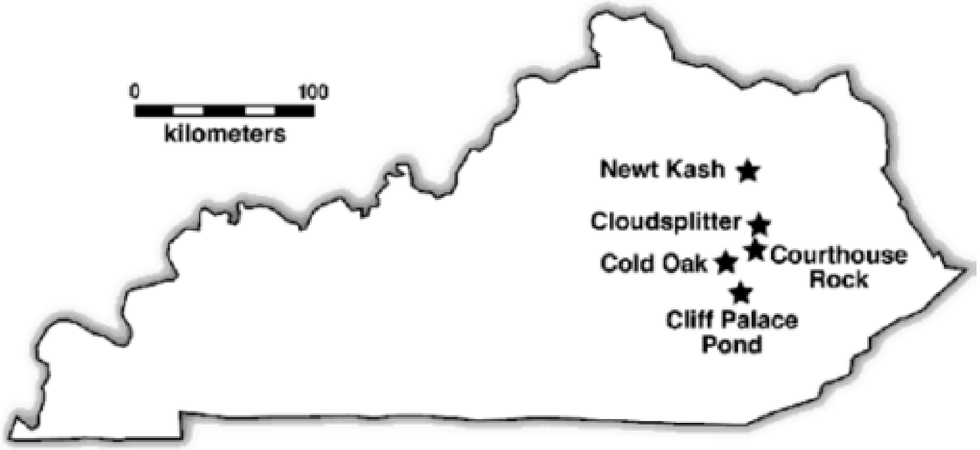

Figure 1.—Escarpment Zone sites discussed in the text.

Newt Kash rockshelter site in Menifee County. Most of the remains analyzed by Jones were from the sleeping beds of the aboriginal inhabitants. These beds were made of a mat-like fabric covering lain over a bed of grass, leaves and straw (Webb and Funkhouser 1939). The bedding also contained all sorts of miscellaneous paraphernalia, which had been accidentally incorporated into the grass mat. Among these were well-preserved seeds, nuts, textile refuse, wood, charcoal and fecal material.

Jones' analysis indicated that the grass, which made up the bulk of the bedding was big bluestem (*Andropogon furcatus*), a robust grass commonly associated with prairie or open woodland environments. Of the identified woods, hickory was the most common followed by chestnut and oak, again species that are uniquely fire adapted. Other trees and shrubs included ash, box elder, sumac and poplar (Jones 1936:160). What intrigued Jones the most about this sample was the preponderance of cultigen seeds including goosefoot (*Chenopodium* sp.), sunflower (*Helianthus annus*), squash (*Cucurbita pepo* var. *melopepo*), marsh elder (*Iva* sp.) and canary grass (*Phalaris caroliniania*).

The paleofecal samples analyzed by Jones also were also quite intriguing. All the cultigens identified within the general contexts of the site with the exception of squash, were represented with the fecal matter. Sunflower for example, while found sparingly elsewhere in the materials examined, comprised a high percentage of the fecal bulk. When combined with the goosefoot, marsh elder and canary grass seeds, plants which were foods grown by simple agricultural methods, comprised the majority of the bulk of the fecal matter.

The study of the ethno-botanical materials from the Newt Kash Hollow rockshelter led Jones to speculate the occupation of the shelter was during a period of transition of the flora from prairie to forest or reverse (ibid p.165). Although archaeologists in the 1930's had no way of absolutely dating the cultural deposits, Jones felt tempted to place the occupation of the site around 4000 years ago. With the advent of radiocarbon dating in the 1950's the intensive occupation of Newt Kash shelter has been determined to be circa 3,000 years ago (Smith and Cowan 1987, Gremillion 1997). Never the less, Jones' work was a keystone for the use of botanical remains from archaeological sites in examining floristic changes upon the landscape.

Cloudsplitter Shelter

It would be over forty years before the well-preserved materials beneath the secluded cliffs of eastern Kentucky once again gained prominence. Although the information gleaned by the early excavations of the 1920s and 1930s had been used to formulate hypotheses concerning the development of horticulture in eastern North America, the precision of the early excavations made it difficult to address synchronic research topics (Cowan and others 1981:60). This changed in 1978 when the National Science Foundation funded excavations at the Cloudsplitter Rockshelter. One of the specific goals of the excavations was to address how human cultural disturbance, especially early horticultural practices affected or altered the local environment. This protected overhang, perched 250 meters above the deeply entrenched Red River floodplain in Menifee County, Kentucky provided the first well-recognized quantitative data sets on early horticultural transitions within the eastern woodlands

The careful excavation of the complex cultural layers at Cloudsplitter provided a 9000-year record of human habitation and interaction with their local environments. At the time of initial occupation the shelter circa 7,000 B.C. the people were faced with a still-evolving postglacial landscape where a dense stand of hemlock shielded the front of the overhang. Today, the mature hemlock first occurs midway down the slope from the site. However, by 1,000 B.C. the deposits of Cloudsplitter, like Newt Kash began to reveal anthropogenic modifications of the local floristic communities. Non-economic plant remains recovered from the Late Archaic deposits (circa 1,000 B.C.) which are indicative of a fringe environment in an otherwise closed canopy forest led Cowan (1985:330-343) to speculate that areas within the immediate vicinity of the shelter had been cleared away. The several cultivars recovered from the Late Archaic deposits including squash (*Cucurbita pepo*), gourd (*Lagenaria siceria*), sunflower (*Helianthus annuus*), and goosefoot (*Chenopodium berlandieri*) probably reflects upland garden plots within the clearings (Ison 1991).

Cold Oak Shelter

In 1984, the Forest Service undertook salvage excavations at the severely vandalized Cold Oak Shelter located at the head of a remote hollow in Lee County, Kentucky. While heavily disturbed by looters, the investigations documented two cultural occupation zones containing a suite of cultigens similar to those recovered from Cloudsplitter and Newt Kash. The pattern that began to emerge from Newt Kash and Cloudsplitter was repeated at Cold Oak. Once again, the earliest component containing the cultivars was a Terminal Archaic horizon dated to circa 1,000 B.C. (Ison 1988, O'Steen and others 1991). It was beginning to become apparent that by the first millennium B.C. the forests surrounding these sites were undergoing major changes at the hands of people.

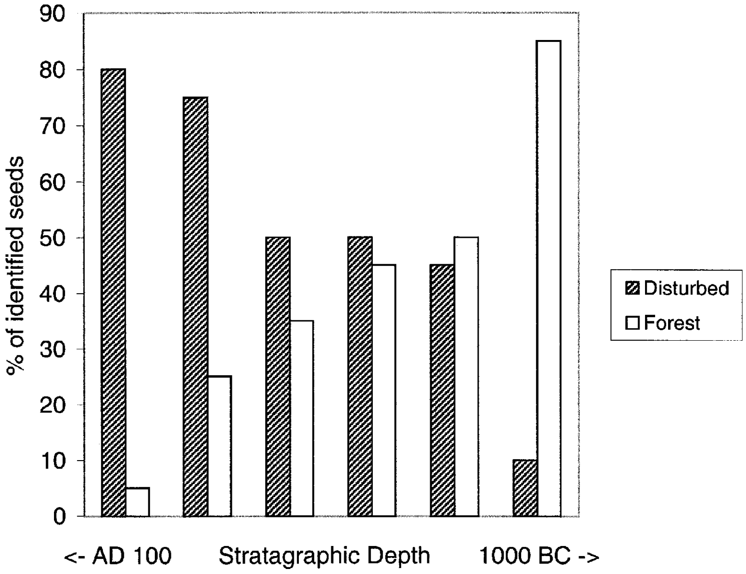

Figure 2.—Forest and disturbed seed taxa from Cold Oak shelter cultural deposits. Adapted from Gremillion 1998.

Ten years later, under the sponsorship of the National Geographic Society, more in-depth excavations were undertaken at Cold Oak (Gremillion 1995, 1998). Building on the earlier investigations, the focus of 1994 data recovery project was to examine the changes in the ecological relationships between people and the plants under their management (Gremillion 1998). The investigations revealed that the quantities of seeds of cultivated plants increased dramatically in the post-1000 B.C. deposits concomitantly with the decrease of seeds represented by taxa of forested habitats (Gremillion 1998). In other word, the analysis of the seed data indicated a general increase in anthropogenic habitats such as gardens and clearings near the habitation sites (figure 2).

Courthouse Rock

The pattern of early plant domestication along the escarpment of the Cumberland Plateau was once again substantiated with the investigations at Courthouse Rock, a sandstone overhang situated approximately 5 kilometers west of Cloudsplitter. The normal suite of seeds representing cultivated plants were recovered from cultural deposits radiocarbon dated between circa 500 and 1000 B.C. While specimens of cultivated plants made up the largest percentage of the identified seeds, the overall assemblage indicated a canopy opening near the site that permitted colonization by sun-loving plants (Gremillion 1999:48). This is most likely the result of preparation of upland garden plots.

Green Sulphur Spring

Even open sites that do not maintain the preservation characteristics of dry rockshelter sites can provide data on anthropogenic forest changes. At the Green Sulphur Spring Site Complex in southern West Virginia differential wood types were noted between the Middle Woodland component (circa A.D. 200 – 400) and the Late Woodland Component (circa A.D. 400 – 800). In sharp contrast to the dominance of hardwood charcoal (99.3 percent) within the Middle Woodland deposits pines represent over 83 percent of the Late Woodland charcoal. This shift in forest composition is thought to have been the result human-induced changes due to land clearing associated with horticulture. As the fields became exhausted or overgrown after several years of cultivation or from periodic abandonment of the site, the fields reverted to forest again with pine being the initial succession species (Rossen 1985, Rossen and Ison 1986).

Hill-side Farming

If these sites are accepted as providing a reasonably accurate indication of prehistoric horticultural endeavors, then the next question is to what extent fire played in the clearing of areas for forest farming. Although there are numerous conditions affecting the decision making process that went into the placement of Kentucky's first garden plots, there are two major underlying cultural mechanisms responsible for their location. These are the "Slug Principal" and "Jones' Law". The Slug Principal dictates that there will be a gravitation toward least effort to produce a food crop (Binford 1983:200-201) while Jones' Law states that "with any given society, the amount of energy obtained in overall subsistence activities must equal or exceed the amount of energy expended in those activities' (Cowan 1988:459). Given these two assumptions, combined with the technology available at the time of early plant domestication, the conditions that most greatly affected development of early horticultural practices along the Cumberland Escarpment were the local topography and the use of fire.

Along the escarpment, little level land is available along the stream margins and where available it is covered by a luxuriant growth of moisture tolerant species. These included the dense stands of cane (*Arundinaria* sp) mentioned by the early explorers of Eastern Kentucky (Filson 1969:318). During the Terminal Archaic Cultural Period (circa 1500 – 500 B.C.) the digging stick was the tool of use, not the hoe. Armed with only this simple implement, it was not practical to create and maintain bottomland fields.

Forest farming depends almost entirely upon the aspect of fire for success unless a sufficient labor bank is available to clear and maintain the plot by other means. Since the direct effect of fire in agricultural endeavors was to clear the surface of refuse and kill most of the plants above the soil, hillside plots were preferred over narrow floodplain plots. The increased fuel moisture along the bottomlands would have greatly reduced the fire's intensity resulting in ineffective burns and therefore more competition with the planted crops. In contrast, the use of fire on hillside plots would have been much more effective for several reasons. Among these are the greatly reduced moisture tolerant understory and a faster curing rate for the one hour and 10 hour fuels. Even with comparable vegetation types within floodplain and hillside plots, the fire will usually be more intense on a slope due to increased heat transfer through convection and radiation to the fuels ahead of the flame front.

Slash and burn agriculture along the Cumberland Escarpment, which had its beginnings over 3000 years ago was a common practice for farming the hill slopes well into the current century (Otto 1983). During the 19th century forest farming was the major cause of wildfires that burned millions of acres of Kentucky's timberlands. In 1880, for example, 10 documented fires burned over 556,000 acres of woodlands in the state. Nearly half of these fires were the result of land clearing exercises (Sargent 1884:491).

It is somewhat paradoxal that while most primitive societies claim a preference for meat, their diet for the most part consisted of plants. As previously noted, Jones' (1936) analysis of the Newt Kash human fecal remains indicated that the inhabitants were chiefly vegetarian. The food supply of these people came primarily from cultivated plants and those gathered from the wild. A similar pattern was reported by Watson (1974) from the fecal data of the Mammoth Cave area where over 50 per cent of the diet were products of horticulture. Thus it seems that by the close of the first millennium B.C. domesticated plants were a very important component of the prehistoric diet. The use of broadcast fire, an essential agent for forest clearing, especially in upland environments of the Cumberland Plateau for agricultural endeavors resulted in a transformation of the forest toward more fire tolerant species.

Palynological Evidence

On of the oldest and still most effective methods used for environmental reconstruction is the retrieval, identification and counting of fossil pollen grains (Butzer 1982:173). By knowing which plants were present and their relative quantities, it is possible to draw conclusions about the climatic and environmental conditions prevailing at the time of deposition (Faegri and Iversen 1989). Quantification of charcoal particles by both size and quantity can be employed as an indicator of past fire histories. For example, charcoal fragments less than 10mm in diameter are considered as indicators of regional fires, whereas those greater than 50mm are assumed to have been generated from forest fires within the immediate vicinity.

Cliff Palace Pond

Cliff Palace Pond is a small woodland pond, perched along the crest of north-south oriented narrow ridge at 424 m elevation in northeastern Jackson County, southeastern Kentucky. This narrow ridge, known locally as Keener Point, is ringed by a high, nearly continuous sandstone cliffline. The sandstone cap is underlain by calcareous shales and limestone that form the lower slopes and stream valleys. Today, the dry ridgetop forest cover is composed by pines (*Pinus rigida*, *P. virginiana* and *P. echinata*) intermixed with scarlet and chestnut oak (*Quercus coccinea* and *Q. prinus* L.). During the historic era chestnut (*Castanea dentata*) occupied the ridgetops on the sandstone derived soils but was absent on calcareous soils (Braun 1950). Mixed mesophytic forests occupied the mesic slopes on calcareous soils below the sandstone cliffs. Since the pond is located within an important center of early-prehistoric plant domestication the deposits offered an opportunity to examine the relationship between prehistoric Native America use of fire and plant production that was retrieved from the surrounding archaeological sites.

The pond was cored in October 1996 resulting in the recovery of a 142-cm sequence of Holocene deposits. The sediment sequence was placed in firm synchronic age classes based on well dated Holocene pollen diagrams elsewhere (Delcourt and Delcourt 1987, 1997, 1998 and Delcourt and others 1998). Charcoal particles, recorded by size class, were also tabulated for each of the distinctive pollen zones. Pollen diagrams were calculated as percentages, representing fire-tolerant trees and shrubs (figure 3) and fire-intolerant trees and shrubs (figure 4) based on the sm totals of pollen grains and spores identified in the sediment core. Figure 5 represents the charcoal diagram in comparison with the major plant taxa represented in the core sample. The following is a brief paleoecological interpretation for each of the distinctive pollen zones.

Interval 9500 to 7300 Years Before Present.

During the early Holocene interval the forests near Cliff Palace Pond were composed of cool-temperate to boreal trees including spruce (probably red spruce, *Picea rebens*), which made up nearly 5 percent of the forest composition. The vegetation was dominated by cedar thought to have been northern white cedar (*Thuga occidentalis*). Cedar pollen, which reaches a maximum of 68 percent within this interval would have been favored by the highly seasonal climate of cold winters and warm summers. Hornbeam (*Ostrya/Carpinus* type) alder (*Alnus rugosa* type) and birch

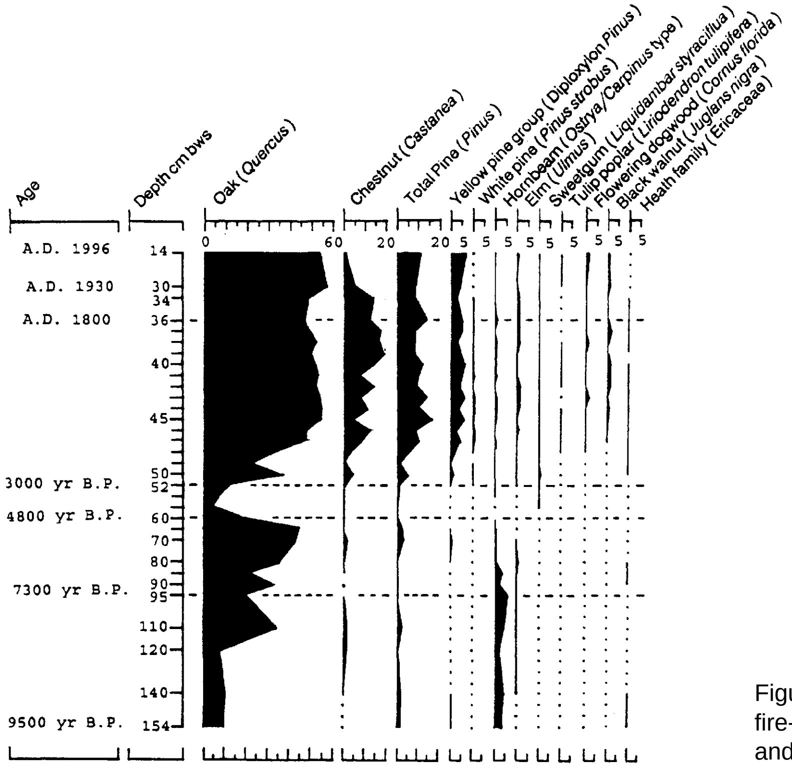

Figure 3.—Cliff Palace Pond pollen diagram, fire-tolerant trees and shrubs. From Delcourt and others 1998.

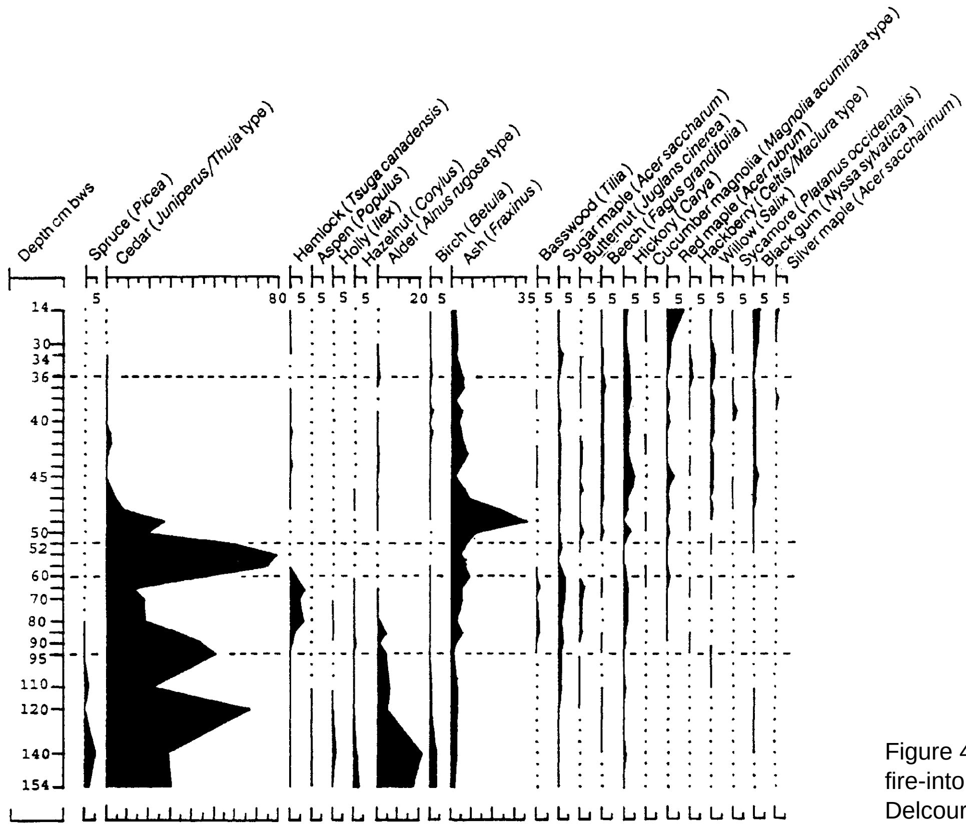

Figure 4.—Cliff Palace Pond pollen diagram, fire-intolerant trees and shrubs. From Delcourt and others 1998.

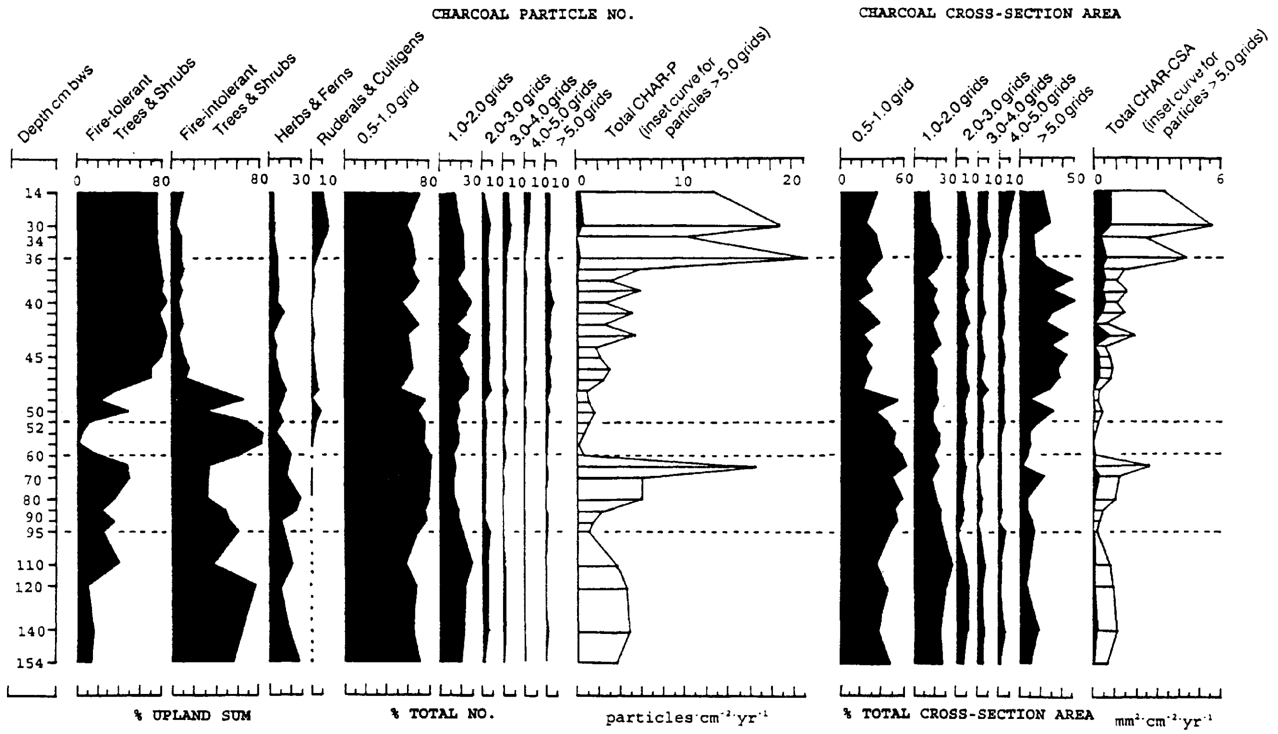

Figure 5.—Cliff Palace Pond charcoal diagram and major plants represented by pollen grains and spores in sediment core. The charcoal particles are tabulated in size classes of cross-sectional area, ranging from 0.5 grid to >5.0 grids (1 square grid = 165.12 μm^2). After Delcourt and others 1998.

(*Betula*) also grew in abundance within the vicinity of Cliff Palace Pond. Pines are represented by up to 2 percent and oaks increase from 10 percent up to 35 percent within this pollen zone. Pollen sums for fire-tolerant trees and shrubs during this interval range from 7 to 20 percent with fire-intolerant species comprising between 19 to 38 percent of the assemblage.

During this period the native people are becoming less nomadic and more territorial. Although they tended to live for longer periods of time in one place, they were still nomadic peoples, never staying in one place longer than a few months. Their camps were placed within areas rich in a variety of natural resources. Anthropogenic fires were used primarily for game drives.

Interval 7300 to 4800 Years Before Present.

The Middle Holocene climate was warm but with lessened extremes in temperature. The shifting storm patterns associated with the “Hypsithermal Interval” resulted in increased precipitation for southeastern Kentucky although it was a time of warmth and drought in the midwestern portions of the U.S. Within the vicinity of Cliff Palace Pond the northern white cedar and spruce became locally extinct and was replaced by the immigration and establishment of temperate trees creating a mixed mesophytic forest community including hemlock (1 to 6 percent), ash (up to 5 percent), basswood (1 percent) sugar maple (up to 4 percent), butternut (1 to 2 percent) and beech (trace amounts). Oaks increase to 45 percent while pines remains at less than 5 percent.

Although this is a time of increased moisture, the total values for fire-tolerant taxa range from 8 to 25 percent. The range of fire-intolerant taxa with a range between 16 to 31 percent shows a slight decrease. These differences are thought to reflect the steady increasing Native American populations’ continued use of fire to promote hunting and foraging.

Interval 4800 to 3000 years Before Present.

This interval begins with a major decline in the hemlock composition of the forest community. Throughout its range in eastern North America a widespread dieback of hemlock has been observed around 4800 years ago. This is thought to have been the result of an infestation of the hemlock looper (Davis 1981). Coincident with the hemlock decline is a large peak in the charcoal accumulation rates at Cliff Palace Pond indicating a major fire event. Following the hemlock decline and the catastrophic fire, extensive stands of eastern red cedar (*Juniperus virginiana*) were established on Keener Point. Cedar pollen dominates the pollen assemblage at 80 percent, with oaks persisting at 3 to 6 percent and ahs at 5 percent. Fire-tolerant trees and shrubs decrease to a minimum of 2 percent, whereas fire-intolerant taxa increase to a maximum of 82 percent.

Interval 3000 to 200 years Before Present.

Cedar declined dramatically in importance falling from 60 percent to trace amounts. Its demise was followed briefly by a succession of ash (increasing from 6 t 36 percent, then dropping to 5 percent). Subsequently, the forests of Keener Point were dominated by oaks, rising from 11 to 55 percent; chestnut, from 1 to 19 percent; and pine from 1 to 17 percent. During this interval pollen grains of both cultigens within the “eastern agricultural complex” and weedy or ruderal species such as ragweed that invade forest openings made by humans activities appear within the record.

Paradoxically, during the Late Holocence time of climatic cooling and increased participation, the forest on and around Keener Point was dominated by fire-adapted taxa. The overall values for fire-tolerant woody plants increase up to 82 percent, while fire-intolerant trees and shrubs diminish to less tan 5 percent. This complete shift in forest composition is considered to have been brought about in large part to Native Americans bringing several plants under domestication. The charcoal record for this interval shows a steady increase in large charcoal particles indicating local fires occurred within the immediate vicinity of Cliff Palace Pond. This is in line with the slash and burn agriculture techniques that would have been employed by the prehistoric framers to clear their upland garden plots.

Interval A.D. 1800 to 1996.

During the last 200 years, a series of changes in forest composition occurred following the total replacement of Native American populations by the EuroAmerican settlement of eastern Kentucky. The chestnut blight of the 1930s results in almost total eradication of this tree. This is offset by an increase for oaks (54 to 58 percent), red maple (7 percent) and black gum (3 percent). Total pine pollen remains at about 10 percent but Virginia pine (*Pinus virginiana*) increases locally. Following deforestation and conversion of large portions of the landscape to agriculture ragweed increases to 12 percent and exotic species introduced from Europe such as plantain, dock and purslane (*Portulaca*) are found within the pollen record for the first time.

For the first 100 years of this period, fire activity increased as forested lands were cleared and burned for new agricultural fields. Slash and burn agriculture, practiced by the American Indians continued only on a larger scale. During the last 100 years fire prevention programs have dramatically reduced large-scale fire occurrences.

Conclusion

As previously noted, the reconstruction of vegetational histories must come from an interdisciplinary approach between natural and cultural resource studies. Historic accounts for example, can offer a multitude of facts about an area with regards to the vegetative cover, fire occurrences, climate, wildlife, land-use practices, etc. but within the New World can only cover the last 500 years. Archaeological sites often contain enormous amounts of ethnobotanical remains that provide insights into the selection and utilization of economically important plants over several millennia. These can range from the selection of firewood and construction materials to the propagation of plant food resources. However, archaeological sites often lack the spatial framework necessary to extract a record of how the human groups affected their environments beyond the immediate vicinity of the site. Pollen and charcoal analysis from geologically deposited sediments on the other hand is a very effective technique for reconstructing the vegetational and fire history of a broader area but in most cases cannot determine the causal factors for the change. Each of these approaches provides a unique dimension to reading the landscape. It is only through a multidisciplinary approach that the tangible effects of fire on the landscape can be viewed within a diachronic perspective.

The interaction of prehistoric human activities and forest dynamics on the landscape took place by the way of the interrelationships among the cultural use of fire, cultivation of plants and forest succession. The changes upon the landscape for the most part, occurred very gradually over hundreds or thousands of years. The transition from on forest type to another was so gradual that for those who didn’t keep diaries, logs, or oral traditions, the change was barely noticed. It is through archaeological and palynological investigations that the murky waters of past forest conditions become clear.

Through the combination of archaeological data from sites within the general vicinity with the pollen and charcoal analysis of Cliff Palace Pond, a link between forest composition, fire history and the interaction of people with their environment is beginning to emerge. The strong coincidence of a temporal link between the domestication of native plants, the prehistoric human use of rockshelters and the increase of local fires as evident from the charcoal record of Cliff Palace Pond argues for a cause-and-effect relationship between Native American activities and changes in forest composition. Within the Escarpment Zone of the Cumberland Plateau prehistoric Native Americans began to concentrate their activities in and near rockshelters on the upper slopes and on ridgetops (Cowan 1985a,b; Ison 1988). The use of fire especially in the preparation of the garden plots adjacent to these campsites resulted in a fine-grained patchwork of vegetation that included fire-adapted and fire tolerant species on the upper slopes and ridgetops while still allowing for the persistence of mixed mesophytic forest communities on the lower slopes.

It should be quite clear that environmental reconstructions cannot be accomplished without including people into the equation. That fire undoubtedly shaped the ecosystems of the southern Appalachians is evidenced by numerous examples. The historical, archaeological and palynological evidence from the Cumberland Escarpment has provided us with the data we have for so long desired. It is now up to land managers to put this data to work. Given the documented fire history for the area we must once again allow fire to play its natural role. If we do not, we will be the progenitors of a forest that has never existed on the landscape before.

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Fire and the Ecological History of Oak Forests in the Eastern United States

Marc D. Abrams[1]

Abstract

Oak distribution and dominance has exhibited major changes since European settlement during the 18th and 19th centuries. Large-scale increases in oak species have occurred as a result of fire exclusion in the central tallgrass prairie and savanna regions, cutting and burning of the northern conifer and hardwood forests of New England and the Lake States region, and land clearing, the charcoal iron industry, and the eradication of *Castanea dentata* following in the mid-Atlantic region. Studies of the dendroecology and successional dynamics of several old-growth forests indicate that prior to European settlement oak grew and regenerated in uneven-aged conditions. At times oak growth was very slow (< 1.0 mm/year) for long periods, which is usually characteristic of highly shade tolerant species. Oak species exhibited continuous recruitment into the canopy during the 17th, 18th and 19th centuries, but stopped recruiting in the early 20th century. Since that time, later successional, mixed-mesophytic species have dominated understory and canopy recruitment, which coincides with the period of fire exclusion throughout much of the eastern biome. Major oak replacement species include *Acer rubrum*, *A. saccharum*, *Nyssa sylvatica* and others. Logging of oak forests that have understories dominated by later successional species often accelerates the oak replacement process.

Basal cross sections were obtained from a partial timber cut in two old-growth forest stands in Western Maryland, USA, in 1986. We recorded 42 fires from 1615 to 1958, which occurred on average every eight years. There were no significant differences in mean return interval between pre-European settlement (before 1750) and the post-settlement (1750-1900) periods. However, there were no major fire years (25% of samples with scars) after 1900. Oaks recruited consistently from the early 1600s to the early 1900s, but there was increasing amounts of *A. rubrum* and *B. lenta* after 1900. Tree-ring dynamics, species recruitment patterns, and long-term fire history reviewed in this paper offer important direct support for the hypothesis that periodic fire played an important role in the historical development of oak forests of the mid-Atlantic region before and after European settlement, and that fire suppression during the 20th century is facilitating a replacement of canopy oak by later successional tree species.

Introduction

Oak species have a long history of domination in eastern North America, and their present distribution in various regions exceeds that recorded in the original forests prior to European settlement (Abrams 1992). Much of the increase in oak during the late 18th and 19th centuries can be attributed to historical changes in disturbance regimes in the eastern biome. Moreover, much of the expansion of oak has occurred on xeric or nutrient-poor sites, which indicates the stress tolerance capabilities of many oak species. However, recent evidence indicates that oak forests, in the absence of periodic understory fire, are often transitional to later successional forest types. The purpose of this paper is to review to linkages in the dynamics for oak species of the eastern United States in relation to the historical changes in land-use and fire regimes.

Historical Ecology of Eastern Oak Forests

Central Plains Example

Prior to European settlement, tallgrass prairie and oak savannas dominated vast areas of the Central Plains, southern Lake States and mid-western regions of the United States (Kuchler 1964, Nuzzo 1986). Much of this region is now part of oak-hickory forest association. Tallgrass prairie and oak savannas in this drought-prone region were maintained by frequent fire at 1-10 year intervals that were initiated by Indian (Native American) activity or lightning strikes (Cottam 1949, Day 1953, Pyne 1983, Abrams 1992).

Eastern Kansas represents the western limit of the eastern deciduous forest, and oak species often grow along streams and ravines forming relatively thin bands of “gallery” forests. A study of the forests in a Kansas (Konza) tallgrass prairie was completed to characterize the composition, structure, development and successional dynamics of this oak-dominated forest type (Abrams 1986). The range of sites on Konza Prairie included mesic riparian benches to xeric limestone ridges. Tree species importance varied with site moisture relations with *Celtis occidentalis* - *Q. macrocarpa*, *Q. macrocarpa*, *Q. muehlenbergii* - *Q. macrocarpa*, and *Q. muehlenbergii* dominating forests along a continuum from mesic to xeric, respectively. In each of the gallery forests studied, oak species represented the oldest and largest individuals, whereas the understory trees and regeneration layers were dominated primarily by *C. occidentalis*, *Ulmus rubra* and *U. americana*, and *Cercis canadensis*. An analysis of the historical records, including the original land survey in 1858 and aerial photographs taken in 1939 and 1978, indicated that the extent of the gallery forests has greatly expanded.

This study exemplifies a major developmental pathway of oak forests in the Central Plains. High fire frequency and intensity during the period of Indian habitation maintained tallgrass prairie species and retarded oak distribution, relegating oak species to savannas and protected woodlands. Following European settlement, the influence of fire decreased due to road construction, expansion of towns, cattle grazing, fire suppression activities, and the elimination

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[1] Professor of Forest Ecology, The Pennsylvania State University, School of Forest Resources, 4 Ferguson Building, University Park, PA 16802

Table 1.—Presettlement forest types in the oak-hickory and oak-pine associations of the eastern United States

Presettlement composition	State	References
Q. alba - Q. velutina - Pinus	MA	Whitney and Davis 1986
Q. alba - Q. velutina - Carya	NY	Glitzenstein and others 1990
Q. rubra - Castanea dentata	NJ	Ehrenfeld 1982
Q. alba - Q. velutina - C. dentata - Carya	NJ	Ehrenfeld 1982
Q. alba - Q. velutina - Carya	NJ	Russell 1981
Q. alba - Q. velutina - Carya	PA	Abrams and Downs 1990
Q. velutina - Q. alba - C. dentata - Carya	PA	Mikan and others 1994
Q. prinus - Q. alba - Pinus rigida - C. dentata	PA	Nowacki and Abrams 1992
Q. alba - P. strobus - Carya	PA	Nowacki and Abrams 1992
Q. alba - Q. rubra	VA	Spurr 1951
Q. alba - Q. rubra - Q. prinus - Carya - Pinus	VA	Spurr 1951
Q. alba - Q. rubra	VA	Orwig and Abrams 1994
C. dentata - Q. rubra	VA	Braun 1950
Q. rubra - Q. prinus - C. dentata	VA	Braun 1950, Stephenson 1986
Q. alba - C. dentata - Carya - Pinus	WV	Abrams and others 1995

of Indian fire activity (Pyne 1983, Abrams 1986). With less fire, oak species expanded into the tallgrass prairie vegetation, with *Q. macrocarpa* and *Q. muehlenbergii* dominating mesic and xeric sites, respectively, in this example. Thus, a significant proportion of the oak-hickory forest in the former tallgrass prairie region is a recent phenomenon in response to fire exclusion following European settlement (Gleason 1913, Kucera 1960).

Lake States Example

Presettlement forests of the upper Lake States and northeast were dominated by *Tsuga canadensis*, *Pinus strobus*, *A. saccharum*, *F. grandifolia* and *Betula allegheniensis*, with generally a very small percentage of *Quercus* (e.g., *Q. alba*, *Q. rubra* and *Q. velutina*) (McIntosh 1962, Siccama 1971, Finley 1976, Whitney 1986). In contrast, *Quercus* species now represent a significant proportion of northern hardwood-conifer forests, and in particular *Q. rubra* has developed prominence (Crow 1988, Whitney and Davis 1986). We studied the presettlement forest records and current forest composition and structure of 46 *Q. rubra* forests along an edaphic gradient in north-central Wisconsin to gain an understanding of their historical development and current and future ecological status (Nowacki and others 1990).

Prior to European settlement, forests on mesic and transitional mesic sites in the study area were dominated by *Tsuga canadensis*, *Betula*, *Acer* and *Pinus* . Dry mesic sites formerly comprised *Pinus*, *Quercus* (*Q. velutina*, *Q. macrocarpa* and *Q. alba*) and *Populus*, while dry-mesic sites were dominated by *Pinus*, *Populus* and *Betula*. In contrast, many forests of the region are presently dominated by *Q. rubra*, with relative importance values of 37-51% (Nowacki and others 1990). Other important overstory trees included *Acer rubrum* on transitional and dry mesic sites, *A. saccharum* on mesic and transitional mesic sites, *Q. alba* on transitional dry-mesic sites and *Betula papyrifera* on dry-mesic sites. Understory trees and reproduction layers were dominated primarily by *A. saccharum* on mesic sites, *A. saccharum* and *A. rubrum* on transitional sites, and *A. rubrum* on dry-mesic sites.

The results of this study indicate that *Q. rubra* had a marked expansion in northern hardwood-conifer forests following Euro-American settlement. *Quercus rubra* on mesic and transitional mesic sites developed following disturbance to the original conifer-northern hardwood forests. Forests on transitional dry mesic and dry mesic sites developed from former oak-pine and pine forests, respectively. A post-settlement increase in *Q. rubra* has been documented in other forests in the northeastern and Lake States regions (Whitney 1986, Whitney and Davis 1986, Crow 1988, Abrams 1992), and appears to be a direct result of widespread cutting and subsequent fire in the middle to late 1800s and early 1900s. Evidence indicates that *Q. rubra* in the overstory was present in relatively low numbers in presettlement forest, but may have been pervasive in the understory of the former pine forests. This coupled with the widespread dispersal of acorns by birds and small mammals facilitated the expansion of this species following large-scale disturbances of the original northern hardwood-conifer forests (Crow 1988).

Mid-Atlantic Region Example

Presettlement forests of southern New England and the mid-Atlantic region were dominated by *Quercus* in combination with other species (Table 1). The leading tree species were *Q. alba*, *Q. velutina*, *Q. rubra*, *Q. pinus*, *Carya* spp., *Castanea dentata* and *Pinus* spp. (including *P. strobus*, *P. rigida*, and *P. echinata*). Evidence from eye witness accounts and charcoal studies indicate that precolonial fires from Indian activity and lightning strikes were pervasive in the region and probably played an important role in the long-term stability of these forest types (Day 1953, Watts 1980, Lorimer 1985, Patterson and Sassaman 1988, Abrams 1992).

As in other regions of eastern North America, disturbances associated with European settlement had a dramatic impact on the original oak-hickory and oak-pine forests. Widespread logging and increased fire associated with land clearing, the charcoal iron industry, tanbark and chemical wood cuts, and lumbering of high quality hardwood and conifers (e.g., *P. strobus* and *Tsuga canadensis*) occurred during the initial settlement period (Pearse 1876, Abrams and Nowacki 1992, Russell and others 1993, Mikan and others 1994). In one example from central Pennsylvania, there were nine active iron furnaces and ten forges in Centre County in 1826, which were responsible for the clearing vast forest acreage each year for charcoal production (Abrams and Nowacki 1992). This type of disturbance regime was responsible for significant changes in species assemblages. In central Pennsylvania, the original *Q. alba* - *P. strobus* - *Carya* forests that were clearcut and burned in the 1800s became dominated almost exclusively by *Q. alba* and *Q. velutina* (Abrams and Nowacki 1992). Cutting for charcoal in New Jersey resulted in the increased dominance of *Quercus* and *Betula*, and decreased dominance of *Tsuga* and *Fagus* (Russell 1980). *Quercus rubra* importance increased from 7% in presettlement *P. strobus* forests in Massachusetts to nearly 20% in present-day forests in response to landclearing and logging (Whitney and Davis 1986). The decrease in *T. canadensis* and *P. strobus* in these examples can be related, at least in part, to their inability to reproduce vegetatively.

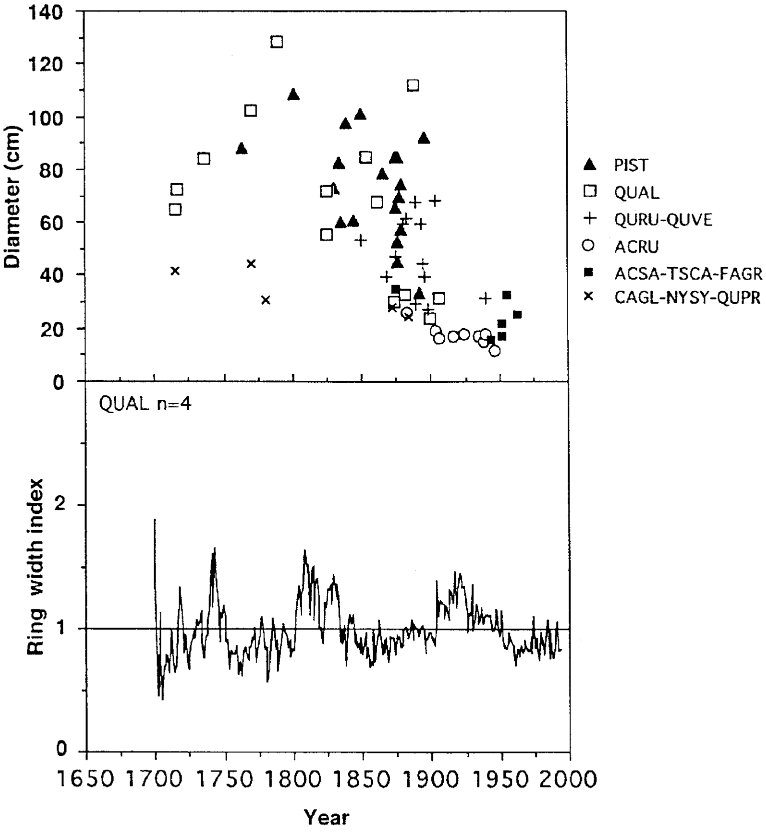

Figure 1.—Age-diameter data for all cored trees and mean ring width index for white oak and white pine and in an old-growth white pine-mixed oak forest in southern West Virginia (adapted from Abrams and others 1995). PIST = *Pinus strobus*, QUAL = *Quercus alba*, QURU = *Q. rubra*, QUPR = *Q. prinus* ACRU = *Acer rubrum*, ACSA = *A. saccharum*, TSCA = *Tsuga canadensis*, FAGR = *Fagus grandifolia*, CAGL = *Carya glabra*, NYSY = *Nyssa sylvatica*.

Another major anthropogenic influence to eastern *Quercus* forests has been the introduction of the chestnut blight fungus (*Endothia parasitica*) during the early 1900s. This fungus has been responsible for the elimination of overstory *C. dentata* thought the eastern biome. The changes to former chestnut dominated forests has been the subject of several studies, most of which indicate that *Quercus* species were one of the major beneficiaries of this disturbance. For example, former oak-chestnut forests in North Carolina became dominated by *Q. rubra*, *Q. prinus*, *Q. alba* and *Carya* spp. (Keever 1953) (Tables 1 and 2). In southwestern Virginia, *Q. rubra* represented 69% importance in forests where *C. dentata* formerly comprised up to 85% of the canopy (Stephenson 1986). In the ridges of central Pennsylvania, *Q. prinus*, *Q. rubra* and *Acer rubrum* increased where *Castanea* and *Pinus* were previously important (Nowacki and Abrams 1992). Thus, post-settlement disturbances to eastern forests via land clearing, the charcoal iron industry, lumbering and the chestnut blight have led to increases in *Quercus* above levels estimated in the original forest.

Q. alba and *P. strobus* dominated the original forests on mesic valley floor sites within the eastern Ridge and Valley Province, which extends from southeastern New York to southern Tennessee (Braun 1950). The composition, diameter and age structure, and radial growth chronologies were studied in one of the few remaining undisturbed remnants of this forest type located in southern West Virginia (Abrams and others 1995). The forest is presently dominated by *P. strobus* (34%), *Q. alba*, *Q. rubra* and *Q. velutina* (26% total) and *Acer rubrum* (24%), and is uneven-aged with *Q. alba* (max. age = 295 years) and *P. strobus* (max. age = 231 years) representing the oldest and largest trees (Fig 1). *Q. alba* exhibited continuous recruitment into the tree size classes from 1700-1900, whereas peak recruitment of *P. strobus* occurred between 1830-1900. Interestingly, the increase in *P. strobus* was followed by a wave of *Q. rubra* and *Q. velutina* recruitment, suggesting possible facilitation of these red oaks by *P. strobus* (cf. Crow 1988, Abrams 1992). After 1900, *Pinus* and *Quercus* recruitment stopped, while that of *A. rubrum*, *A. saccharum*, *F. grandifolia* and *T. canadensis* greatly increased.

Table 2.—Overstory and understory composition reported for *Quercus* forests in the major vegetation associations of the eastern United States

Overstory	Understory	State	Reference
Oak-hickory			
Q. macrocarpa - Celtis occidentalis	*C. occidentalis - Ulmus*	KS	Abrams 1986
Q. muehlenbergii	*Ulmus - Cercis canadenis - Q. muehlenbergii*	KS	Abrams 1986
Q. velutina - Q. alba	*Prunus serotina - Acer negundo - Ulmus*	WI	McCune & Cottam 1985
Q. marilandica - Q. stellata	*Q. marilandica - Q. stellata*	OK	Rice & Penfound 1959
Q. alba - Carya ovata	*A. saccharum*	IL	Rodgers & Anderson 1989
Q. alba	*A. saccharum*	MO	Pallardy and others 1988
Q. alba - Q. velutina	*A. rubrum - A. saccharum - Betula lenta*	NJ	Little 1974
Q. alba - Fagus grandifolia	*A. rubrum - A. saccharum - P. serotina*	PA	Abrams & Downs 1990
Q. alba - Q. velutina	*A. rubrum - P. serotina*	PA	Abrams & Nowacki 1992
Q. prinus - Q. rubra	*A. rubrum - Q. prinus - B. lenta*	PA	Nowacki & Abrams 1992
Q. prinus	*A. rubrum - B. lenta - B. allegheniensis*	PA	Mikan and others 1994
Oak-pine			
Q. alba - Liriodendron tulipifera	*Nyssa sylvatica - Cornus florida*	VA	Orwig & Abrams 1994
Q. alba - Q. coccinea	*N. sylvatica - A. rubrum - C. florida*	VA	Orwig & Abrams 1994
Q. alba - Q. prinus - Q. coccinea	*N. sylvatica - Carya - C. florida*	VA	Farrell & Ware 1991
Q. rubra	*A. saccharum - A. rubrum*	VA	Stephenson 1986
Q. alba	*A. rubrum*	NC	Christensen 1977
Q. rubra - Q. prinus - Q. alba	*Q. rubra - A. rubrum - Carya*	NC	Keever l953
Maple-beech-basswood			
Q. alba - Q. velutina	*P. serotina - A. rubrum*	MI	Dodge & Harman 1985
Q. macrocarpa - Q. alba	*A. saccharum - Fagus grandifolia*	OH	Cho & Boerner 1991
Q. alba - Q. rubra - Q. velutina	*A. saccharum - F. grandifolia - A. rubrum*	OH	McCarthy and others 1987
Q. alba - Q. velutina - Q. rubra	*A. saccharum - F. grandifolia*	IN	Schmelz & Lindsey 1970
Q. rubra - Q. alba - Q. macrocarpa	*A. saccharum - Aesculus glabra*	IN	Parker and others 1985
Northern hardwood - Conifer			
Q. velutina - Q. rubra - Pinus strobus	*A. rubrum*	MA	Lorimer l984
Q. prinus - Q. rubra	*A. rubrum - A. saccharum*	NY	McIntosh 1972
Q. velutina - Q. alba	*A. rubrum*	MI	Host and others 1987
Q. rubra - Q. alba	*A. saccharum - A. rubrum*	WI	Nowacki and others 1990
Q. ellipsoidalis	*P. serotina - A. rubrum*	WI	Reich and others 1990
Mixed-mesophytic			
Q. alba - P. strobus	*A. rubrum - F. grandifolia - Tsuga canadensis*	WV	Abrams and others 1994
Q. alba - Q. coccinea - Q. prinus	*A. rubrum - A. saccharum - F. grandifolia*	KY	Muller & McComb 1986
Q. alba - Q. velutina	*A. rubrum - N. sylvatica*	KY	Campbell 1987
Q. prinus - Q. alba	*Q. prinus - Q. alba*	TN	Schmalzer 1988
Southern evergreen			
P. palustris - P. elliotii - Q. laevis	*Q. laevis - Q. geminata - Carya floridana*	FL	Myers 1985
F. grandifolia - Q. laurifolia - Magnolia	*Liquidambar styraciflua - N. sylvatica - Fagus*	GA	Quarterman & Keever 1962

Radial growth analysis of the four oldest *Q. alba* indicated a series of releases between 1710-1740, 1800-1830 and 1900-1930, with low or decreasing growth in the interim and most recent periods (Fig. 1). In the early 1800s, releases in radial growth were associated with high *P. strobus* recruitment, while releases in the early 1900s coincided with episodic *A. rubrum* recruitment. Individual radial growth chronologies for trees of various species and age classes indicated a series of major and moderate releases every 20-30 years throughout the forest (Abrams, et al. 1995). The asynchronous nature of these releases suggest a series of small-scale disturbances with localized impacts.

We found evidence of fire scars, soil charcoal and windthrow throughout the forest, and believe that these disturbance factors significantly influenced the ecology of this old-growth forest. *Quercus* and *Pinus* perpetuated themselves during the 1600s, 1700s, and 1800s, but not in the 1900s, despite evidence of blowdown during this century. These data are consistent with the fire exclusion hypothesis, which led to a shift in tree recruitment from *Quercus* and *Pinus* to *Acer*, *Fagus* and *Tsuga*. Without intensive management in the future, including prescribed fire, we predict this forest will no longer support a significant *Quercus* and *Pinus* component.

Dendroecology of Old-growth *Quercus Prinus*

We identified an old-growth *Q. prinus* forest on a dry talus slope with canopy trees up to 326 years old at the Detweiler Run in central Pennsylvania (Ruffner and Abrams 1998). Unlike the transitional nature of eastern oak forest on mesic sites, this xeric forest may maintain stable oak populations even in the absence of fire. In 1996, *Q. prinus* at the study site represented 39% importance, while *A. rubra*, *Betula lenta*, *Nyssa sylvatica*, and *Q. rubra* represented a combined 50% importance. *Q. prinus* represented 75% of the canopy dominant trees, but less than 10% of the intermediate and overtopped trees. Continuous recruitment of *Q. prinus* in this forest occurred between 1660-1950 (Fig. 2). Peak recruitment periods for *Q. prinus* occurred during the late 1700s and early 1800s, which coincided with a release in radial growth during this period.

Ironically, this forest is located near major 18th and 19th century charcoal iron activity, where adjacent forests were logged on a 20-30 year rotation. The extreme talus slope undoubtedly protected this forest from cutting during that period. Frequent cutting and occasional burning of most forests in the region promoted oak coppicing and checked the advance of later successional species until the late 1800s when charcoal iron production ceased in central Pennsylvania. Fire suppression activities and less forest cutting during the 20th century promoted *Acer*, *Betula*, and *Nyssa* dominance in area forests. Because the extreme edaphic condition, we predict that *Q. prinus* will remain a dominant on the site with lesser amounts of more shade tolerant *Acer*, *Nyssa* and *Betula* species. Thus, oak climaxes may exist in the eastern oak region on very dry and nutrient poor sites, such as this study area.

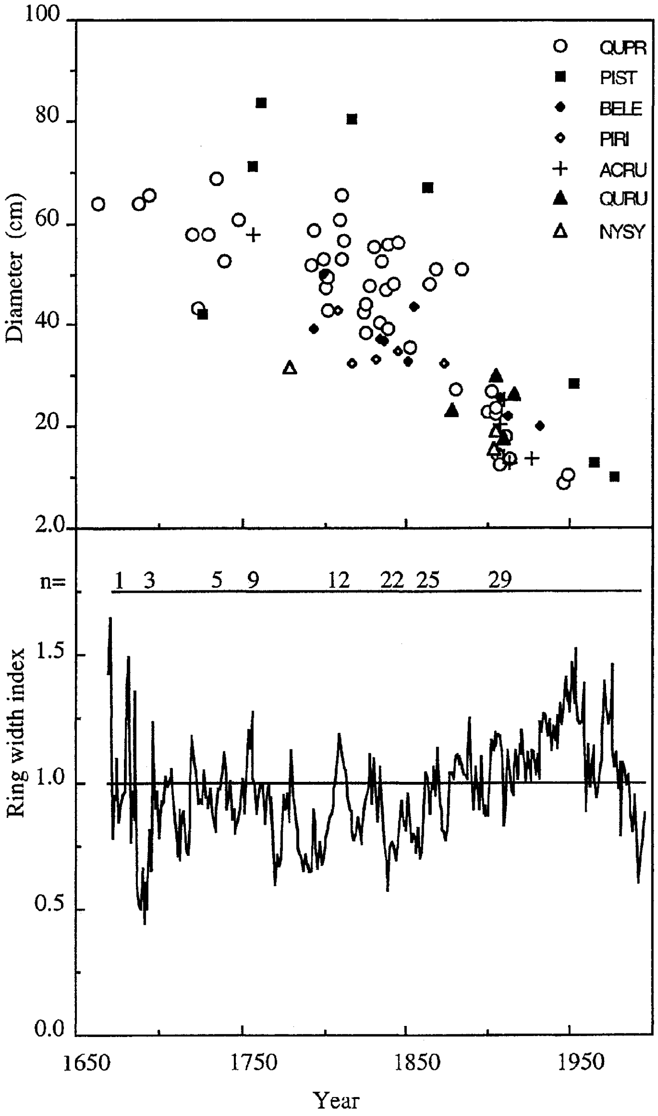

Figure 2.—Age-diameter data for all cored trees and mean ring width index for chestnut oak in an old-growth mixed-oak forest in central Pennsylvania (adapted from Ruffner and Abrams, 1998).

Successional Status of Eastern Oak Forests

Despite their importance throughout the eastern forest biome, there is a strong indication that many oak forests are transitional to later successional associations. Table 2 lists the principal overstory and understory composition in various oak forest types. *Acer rubrum* and *A. saccharum* are the two most important understory species, and they most likely represent the major replacement species of oak throughout much of the biome (cf. Lorimer 1984, 1989, Abrams 19992, 1998). However, west of the *Acer* range, *Celtis occidentalis* and *Cercis canadensis* appear to be important oak replacement species (Abrams 1986). In the mid-Atlantic region, south of *A. saccharum*'s range, *Nyssa sylvatica* may be a future overstory dominant in current oak forests (Ross and others 1982, Farrell and Ware 1991, Orwig and Abrams 1994). *Fagus grandifolia* is occasionally noted as an important understory or overstory in oak forests, and these are generally limited to the maple-beech-basswood and mixed mesophytic associations. *Q. alba* forests in southwestern Pennsylvania and northern Virginia are presently dominated by *F. grandifolia*, *A. rubrum* and *Liriodendron tulipifera* (Abrams and Downs 1990, Abrams and Copenheaver 2000), indicating that *F. grandifolia* may play an important successional role in eastern oak forests as they move from second-growth to old-growth condition.

A few studies predict long-term stability of oak species on very extreme sites. Examples of this include, xeric *Q. marilandica* and *Q. stellata* forests derived from former oak savannas in Oklahoma, *Q. marilandica* - *Q. velutina* in xeric, upland glades in Illinois, mix-oak forests on nutrient-poor barrens sites in New Jersey, dry talus slopes and Coastal Plain forests in the mid-Atlantic region (Little 1974, Adams and Anderson 1980, Dooley and Collins 1984, Orwig and Abrams 1994, Ruffner and Abrams 1998). While these forests may in fact represent edaphic climaxes they may alternatively be exhibiting slow rates of successional replacement and thus not have long-term stability in the absence of fire. Clearly, the rates of succession on mesic oak sites greatly exceeds that on xeric or nutrient poor sites (cf. Abrams 1992). We still know very little about the successional dynamics of oak forests in the southeastern U.S. In one such study, *A. rubrum* is predicted to form a dominant part of the overstory in a mature *Q. alba* forest in North Carolina (Christensen 1977, R. Peet personal communication). However, oak forests located to the south and west of the peak distribution of *A. saccharum*, *A. rubrum*, *F. grandifolia* and *P. serotina* may be experiencing less successional pressure then oak forests in the northeastern and north-central U.S. and may have stable oak populations even in the absence of fire. Nonetheless, red maple density increased by more than 300% in the subcanopy of a mature pine-oak forest in southern Arkansas between 1954 and 1992 (Cain and Shelton 1995).

Fire Adaptations

In an early opinion survey of the fire resistance of 22 northeastern tree species, oaks (*Q. prinus*, *Q. velutina*, *Q. alba* and *Q. coccinea*) where rated in four of the top six positions (Starker 1934). It was further determined that these oak species had a much greater bark thickness to trunk diameter ratio than several mixed-mesophytic species, such as *A. rubrum*, *P. serotina*, and *F. grandifolia* (Spalt and Reifsnyder 1962, Harmon 1984). Among oak species, a ranking of increasing bark thickness and fire resistance was reported as follows: *Q. macrocarpa* > *Q. velutina* > *Q. alba* > *Q. rubra* (Lorimer 1985, Hengst and Dawson 1994). Fire may also be beneficial to oaks, relative to other hardwood species, because they have relatively high resistance to rotting after scarring, deep rooting and vigorous sprouting ability, and increased germination and survival on fire-created seedbeds with reduced litter layers.

We compared the fire history and tree ring dynamics of two old-growth forest stands in Western Maryland (Shumway and others in press). The stands were located on the side-slopes of a ridge system (Savage Mountain) but differed in the amount of periglacial material present. Both stands are dominated by *Quercus rubra*, *Q. alba*, *Q. velutina*, *Q. prinus* and *Acer rubrum*. Basal cross sections were obtained from a partial timber cut in 1986, which provided evidence of 42 fires from 1615 to 1958 (fig.3). Fires occurred on average every eight years, and there were no significant differences in mean return interval among pre-European settlement (before 1750), post-settlement (1750-1900) and fire suppression (after 1900) periods. However, there were no major fire years (25% of samples with scars) after 1900. Oaks recruited consistently from the early 1600s to the early 1900s, but there was increasing amounts of *A. rubrum* and *B. lenta* after 1900. There was a significant degree of correspondence between the occurrence of major fires and recruitment of oak species in either stand. Moreover, there was an increase in radial growth in oak species in a four-year period following the vast majority of fires, and this increase was largely due to increased latewood production following fires. Tree-ring dynamics, species recruitment patterns, and long-term fire history reported in this study offer important direct support for the hypothesis that periodic fire played an important role in the historical development and perpetuation of oak forests of the mid-Atlantic region before and after European settlement, and that fire suppression during the 20th century is facilitating a replacement of canopy oak by later successional tree species (Abrams 1992, 1998).

There exists a substantial database of paleoecological studies that support the oak and fire hypothesis. For example, at Crawford Lake in southern Ontario, beech-maple forests converted to pine-oak forests in response to Iroquois cultivation and burning (Clark and Royall 1995). Repeated disturbances (including fire) at a series of sites in north-central Massachusetts led to a decrease in northern hardwoods and an increase in pine, oak and chestnut during the past 1000 years (Foster and Zebryk 1993, Fuller and others 1998). Large oak pollen percentages were associated with continuous charcoal influx during the Holocene in the Hudson Highlands of southeastern New York (Maenza-Gmelch 1997). A paleoecological investigation in eastern Tennessee revealed that the study area was dominated by oak, pine and ragweed during the last 1500 years, and that charcoal influx increased during the period with increasing human populations (Delcourt and others 1986). These studies suggest a relationship among oak dominance, fire, climate, and human activity.

Direct evidence for the role of fire in oak forests comes from additional studies using the dating of fire scars. Most of these data have been reported for oak forests in the central plains, at the western extreme of the eastern forest, or from pine-oak forests in southern Ontario (Abrams 1985; Guyette and Cutter 1991; Cutter and Guyette 1994; Guyette and others 1995; Guyette and Dey 1995). The mean fire return interval (the time between fires) in these disparate locations is surprisingly similar and ranges from 2 to 24 years. Fire frequency either increased or decreased during the initial period following European settlement depending on the stand and region. A study of fire scars in a mixed-oak forest that originated after 1850 in southeastern Ohio revealed that fire occurred every 1 to 24 years, and averaged 7.5 years (Sutherland 1997). Two black oak (Quercus velutina Lam.) woodlands in Indiana had a fire return interval of 5.2 and 11.1 years during the 1900s (Henderson and Long 1984). Six fires scars dating from 1641 to 1711, with a mean return interval of 14 years, were reported from a pre-European settlement oak in Mettler's Woods, New Jersey (Buell and others 1954).

Conclusions

The rise in oak dominance in the eastern U.S. at the beginning of the Holocene epoch (10,000 y. B.P.) was associated with warmer and drier conditions and the increased occurrence of fire (Watts 1980, Davis 1985, Webb 1988). It is well documented that American Indians actively used fire for a multitude of purposes, and they were probably responsible for increasing the incidence of forest and prairie fires above that caused by lightning strikes (Gleason 1913, Day 1953, Pyne 1985, Patterson and Sassaman 1988).

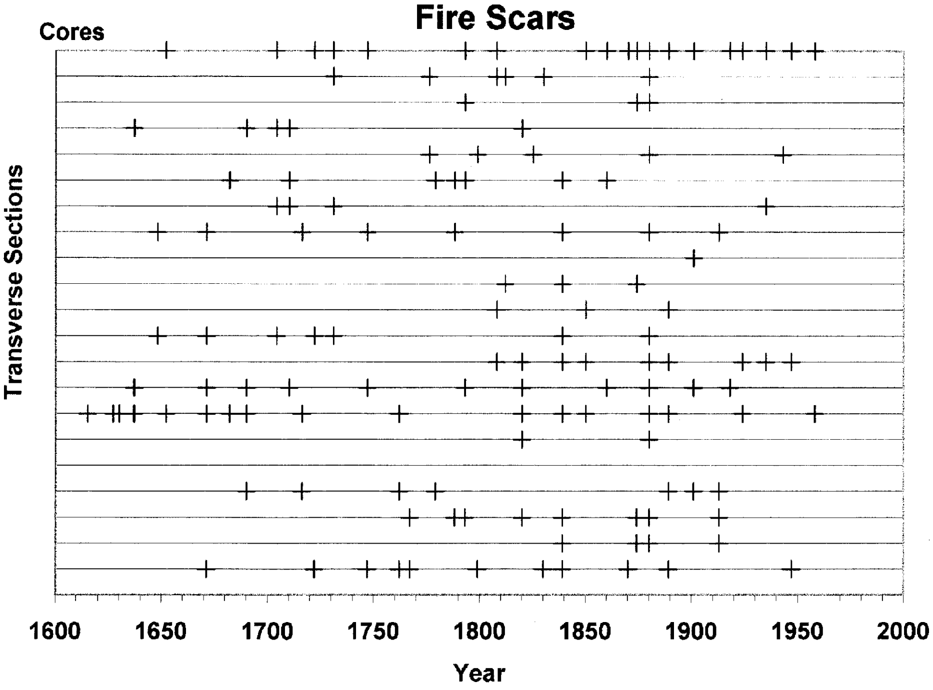

Figure 3.—Fire scar records obtained for 20 transverse sections and fire scars observed in increment cores for an old-growth oak forest on Savage Mountain in western Maryland (Shumway, Abrams, and Ruffner unpublished data).

While fires were too frequent in the tallgrass prairie region and too infrequent in the northern hardwood forests for oaks to prosper, the intermediate frequency and intensity of fire in presettlement oak-hickory, oak-chestnut and oak-pine forests were apparently necessary for long-term oak stability (Abrams 1992).

Initially following European settlement, oak populations throughout much of the eastern biome increased due to fire exclusion in tallgrass prairie and southern pine forests, widespread logging and burning of northern hardwood-conifer forests, and logging, burning and the chestnut blight in the eastern mixed-oak forest types. Moreover, oak species have typically shown a strong affinity for drought-prone sites, which are fairly common in the eastern U.S. from extreme edaphic, physiographic and/or climatic factors. Despite their low to moderate shade tolerance rating, eastern oak species maintained themselves in pre-European settlement forests in uneven-aged conditions, often growing very slowly for long intervals. Recurring fire in presettlement oak understories most likely prevented significant invasion by later successional species. This coupled with adequate light transmission through oak canopies probably facilitated oak establishment and recruitment in presettlement forests. Consistent with these ideas, oak species exhibit a suite of adaptations for fire.

Widespread invasion of most oak understories, excluding the most xeric and nutrient poor sites, by later successional tree species and a lack of oak recruitment coincides with the start of fire exclusion in the early 1900s. It has been argued that continued fire exclusion will lead to a vast reduction in oak dominance in the eastern forest (Lorimer 1985, Abrams 1992, 1998). However, white-tailed deer populations have also risen dramatically in many eastern region during the 1900s. Considering that many oak species are highly preferred browse species, large deer populations are probably exacerbating the oak regeneration problem (Hughes and Fahey 1991, Abrams 1998). Thus, the increase in *Acer* and other later successional trees in many oak forests may be a function of fire exclusion, differential deer browsing, as well as other less well documented factors. Therefore, without intensive management to reduce competition from non-oak species and the predation of seeds and seedlings, such as through the increased use of fire and controlling deer populations, a major loss of overstory oak dominance can be anticipated for the near future.

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Effects of Fire on the Ecology of the Forest Floor and Soil of Central Hardwood Forests

Ralph E.J. Boerner[1]

Abstract

This paper reviews the scanty available database on the effects of dormant season fire on ecological processes and organisms in the forest floor and soils of forests of the central hardwoods region. Data from studies in other ecosystem types are used to supplement the existing database, but extrapolation across ecosystem types should be viewed with caution. Fires in the central hardwoods are typically low in intensity and consume primarily the unconsolidated leaf litter. As long as the fire can move across the open forest floor, soil temperatures generally do not increase enough to cause significant heating-induced mortality among soil-dwelling organisms. Soils under smoldering piles of woody fuels may, however, be subject to sterilization. Direct N volatilization is probably not an important pathway for fire-related nutrient loss due to low fire temperatures. The microclimate at the forest floor surface is probably affected significantly and this may produce phenological changes in root growth and microbial activity. More research in this area is warranted. Base cations released from dissolving ash may or may not increase soil pH and cation availability, depending on the nutrient status of the soil and the amount of ash deposited. Nitrogen availability typically increases after one or a small number of fires but may decrease over the long term. Abundances of soil animals in the forest floor are reduced by fire whereas those in the mineral soil are affected little. Recolonization of the redeveloping post-fire forest floor is rapid. Microbial abundances in the forest floor are typically reduced by fire, but rapid recolonization by these groups is also likely, except under smoldering piles of woody fuels. The database for the central hardwoods region is scanty and this area is greatly in need of additional research attention. Based on this scanty database, belowground impacts of low intensity, dormant season fires in the central hardwoods is likely to be less that the impact aboveground, though considerably more research covering a greater range of ecosystem types and fire frequencies is needed to verify this.

Introduction

The history of ecological studies indicating the importance of relatively frequent fire in the maintenance of forest composition and structure in eastern North America goes back over half a century. For example, Cottam (1949) concluded that frequent fire played a key role in the prehistoric ecology of oak-hickory *(Quercus-Carya)* forests in Wisconsin, and Daubenmire (1936) emphasized the role of fire in the vegetation of the “big woods” region of Minnesota. Based on pollen and charcoal profiles, Delcourt and Delcourt (1997) have determined that fire has been common in this region for at least 4000 yr. Their analysis of the fire record indicates, however, that the importance of local fires has increased and that of regional fires has decreased over time.

The importance of local, relatively small fires in this region is clear from the analysis of fire occurrence over the last century in oak forests in Vinton County, Ohio done by Sutherland (1997). In this region, the great majority of fires occurs either during the dormant season (69%) or in early spring (25%), with large, regional summer fires being rare (Sutherland 1997). The mean fire interval for the Vinton County sites was 3.6 yr, with locally extensive fires occurring on the average of 7.5 yr (Sutherland 1997). Based on these and other similar records of fire in the central hardwoods region (e.g. Guyette and Cutter 1991), this remainder of this review will focus on the effects of relatively low intensity, dormant season fires on the ecology of the hardwood forest floor and soil.

Forest fires may affect the belowground parts of the ecosystem through direct heating and by consuming and/or changing the characteristics of the surface fuels (Figure 1). Such alterations to the ground surface may then affect belowground organisms and ecological processes through direct volatilization of N and S, microclimate alteration, and deposition, dissolution, and convection of ash. In the paragraphs that follow I attempt to review each of these impact components individually, and conclude with a review on the biological and ecological consequences of these impact components taken together. One important caveat is necessary, however, before proceeding: the direct database for the central hardwoods appears to be sparse, at best. This is particularly apparent when comparing the data from this region to what is available for western conifer forests, shrublands, and prairie (reviews by Wells et al. 1979 and Neary et al. 1999). My approach is to summarize the data from the central hardwoods and supplement where required with studies from other temperate, humid ecosystems.

Fire Behavior and Direct Heat Effects

Low intensity dormant season fires rarely produce the large flame fronts and extreme temperatures of dormant season fires in chaparral or growing season fires in conifer forests (Boerner 1982). Franklin et al. (1997) recorded surface fire temperatures during two prescribed burns in forests near the Kentucky-Tennessee border. They reported average temperatures at the surface of the forest floor of 226C in an oak-maple forest and 190C in a mixed oak site. Similarly, Boerner et al. (2000b) reported mean fire temperatures of 210C and 157C in two early spring prescribed burns in oak-hickory stands in southern Ohio, and Blankeship and Arthur (1999a) reported forest floor surface temperature ranges of 316-398C and 205-315C in two prescribed fires in oak-pine forests in Kentucky.

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[1] Professor and Chair, Department of Evolution, Ecology, and Organismal Biology, Ohio State University, Columbus, OH 43210.

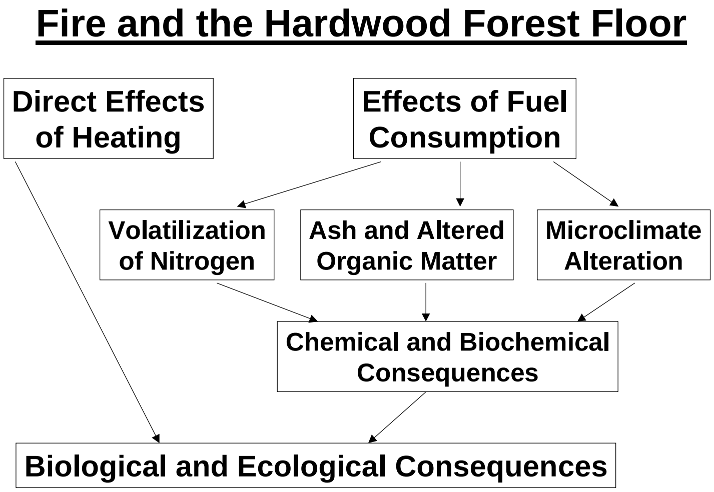


Figure 1.—Schematic approach for analyzing the direct and indirect effects of fire on organisms inhabiting the soil and forest floor.

The range of variability in forest floor surface temperatures that can occur in this region is well illustrated by the studies of prescribed fires in northern Indiana forest stands conducted by Cole et al. (1992). Low creeping fires in oak forests in their sites often burned only leaf litter and produced little surface temperature rise when burning downhill; in contrast, peak temperatures 15 cm above the surface in other stands reached 300-400C with very high spatial variability (Cole et al. 1992).

Variations in fire intensity are related to many factors, including forest floor biomass/depth, slope position, aspect, and angle, and fire weather (Franklin et al. 1997; Boerner et al. 2000b). As an illustration of this, Boerner et al. (2000b) compared the behavior of prescribed fires done in two sites over three days in which the weather changed from warm and relatively dry to cool and moist. The fire conducted during the relatively drier, warmer day attained average temperatures at 10 cm above the forest floor of 210C, with a strong variation in maximum fire temperature related to slope position and soil moisture content. In contrast, the fire conducted during cooler, moister conditions only attained an average of 157C and there was no significant correlation between maximum fire temperature and slope position or soil moisture content (Boerner et al. 2000b). In the former fire, 2/3 of sample plots attained maximum surface temperatures >200C at least transiently, whereas in the latter fire only 1/6 of sample plots reached 200C (Boerner et al. 2000b).

Fires of this intensity typically burn only unconsolidated litter and fine woody fuels, leaving the humus and upper soil layers uncombusted. For example, in the fires described above (Boerner et al. 2000b); litter loss from a single fire ranged from <30% to >80%, depending on landscape position and fire temperature, but the underlying humus layer was unaffected. These results were similar to those obtained in a series of late autumn prescribed burns in oak-pine forests in southern New Jersey in which Boerner (1983) observed losses of 50-70% of the litter mass but <5% of humus mass, as well as an analysis of the impact of a single winter fire on oak-pine forest in Kentucky which demonstrated losses of ~30% of litter mass but no humus (Blankenship and Arthur 1999b). Studies of fires in northern Indiana oak forests and Georgia piedmont forests have also indicated that the effects of direct combustion are typically limited to the unconsolicated leaf litter (Cole et al. 1992; Bender and Cooper 1968).

Although the underlying, humified portion of the forest floor and the upper mineral soil layers are typically not combusted directly in such fires, transfer of heat to the unburned forest floor and soil could produce ecological consequences if severe enough. For example, soil enzymes will denature when temperatures reach 70C even transiently, and temperatures >70C for 10 min will kill fungi, protozoa and some bacteria (Lawrence 1956).

In general, however, surface fires moving across the open forest floor do not have strong potential for severe soil heating because only approximately 5% of heat released by fire is partitioned to soil, and mineral soil is a very poor conductor of heat (Raison 1979). Although there are few studies that report mineral soil temperatures during fire in the central hardwoods, there are studies from other regions that can serve as examples of what one might actually expect in our region. For example Saa et al. (1993) recorded mineral soil temps at 5 cm depth were <50cm in fires in pine forests and gorse (*Ulex europaea*) shrublands in Spain. Heyward (1938) recorded soil temperatures during a fire with 1m flame height in a North Carolina longleaf pine (*Pinus palustris)* forest and found temperatures at 2.5cm into mineral soil of 40C just after the flame front passed and only 25C 45 minutes after the fire had passed.

Such fires can, however, have significant effects on temperatures within the forest floor layers that remain largely unconsumed by fire. Blankenship and Arthur (1999a) report temperatures as high as 315C in the upper 0.5 cm of the Oe layer during a fire in a Kentucky oak-pine forest that generated temperatures as high as 398C at the forest floor surface. Similarly, Cole et al. (1992) measured temperatures in the humus layer during prescribed fires and recorded ranges of 100-133C at 2 cm depth into the humus and 40-90C at 3 cm into the humus. Thus, low intensity fires moving across the open forest floor do not appear to have the potential to cause significant heating-induced mortality to organisms living in the lower humus layers or upper mineral soil, but may have such effects near the litter/humus interface.

The one situation in which direct, heating-induced mortality of soil organisms may become significant is when localized concentrations of woody debris continue to smolder in place for an extended period of time. Miller et al. (1955) monitored ground surface and soil temperatures during and after a fire in a New Zealand shrubland. The main fire area experienced maximum surface temperatures of about 200C, but there was no change in temperature recorded by sensors buried 5 cm and 10 cm into mineral soil. However, in areas where localized wood heaps burned for several hours, sensors recorded maximum temperatures of 100C at 5 cm and 60C at 10 cm depth in mineral soil. Thus, localized woody fuel accumulations do present a situation in which direct heating effects may have negative effects on soil biota.

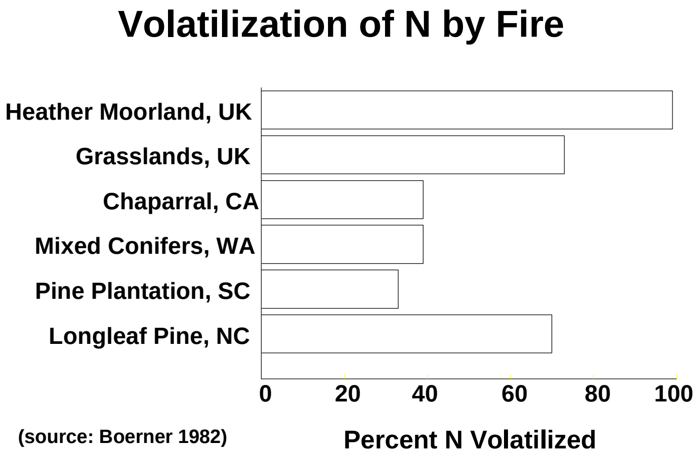
Nitrogen Volatilization

Direct volatilization of N during combustion of litter can result in large losses of N from the ecosystem. In a review of the impacts of fire on nutrient cycling, Boerner (1982) reported net losses of 30-100% of litter N during fire as the result of direct volatilization and ash convection (Figure 2). However, direct volatilization of N does not begin until fire temperatures exceed 200C and does not become a major pathway for N loss until temperatures exceed 300C (Raison 1979; Boerner 1982). Given the ranges of surface temperatures recorded in fires in the central hardwoods noted earlier, it seems unlikely that direct N volatilization will be significant in this region. For example, Boerner et al. (2000b) estimated direct N volatilization from two prescribed burns in Ohio oak-hickory forests to be <1 kg/ha, less than 10% of annual additions to the site through atmospheric N deposition.

In addition to losses due to direct volatilization, nutrients may be lost from the forest floor via convection of ash (Boerner 1982). Monitoring of precipitation and ashfall in sites adjacent to active fires in western conifer forests (Clayton 1976) and southeastern pine forests (Lewis 1974) have demonstrated that nutrient redistribution via ash convection and deposition can have significant effects on nutrient budgets of neighboring sites.

Microclimate Alteration

Removal of the unconsolidated litter and deposition of blackened, partially combusted material on the ground surface have the potential to alter microclimate at the soil surface. Although such effects have commonly been reported in grassland ecosystems (Boerner 1982), to date I've been able to locate no published microclimate data for burned sites in the central hardwoods. However, in an unpublished study from southern Ohio, early spring surface soil temperatures in sites burned annually for three years were higher during the day and lower during the night than were those in soils of unburned controls (Personal communication. 1999. Louis Iverson and Todd Hutchinson, Northeastern Research Station, Delaware, OH 43015). Viro (1974) presented air and soil temperature data for burned spruce plantation in Finland. Soil temperatures were both higher in the burned site than under the canopy in a neighboring unburned site. However, soil temperatures in a canopy gap in the unburned site were intermediate between those of the burned site and those under the intact canopy. Thus, in this site at least, both the opening of the canopy and the alteration of the ground surface conditions contribute to microclimate alteration.



(source: Boerner 1982)

Figure 2.—Proporation of fuel N content lost to direct volatilization and ash convection. Original data sources are cited in Boerner (1982).

What impacts might earlier warming and more extreme conditions have on soil biota? In grasslands, the earlier warming of soils in burned sites contributes to an increase in the rate of N mineralization and nitrification, compared to unburned sites (Boerner 1982). This may well occur in central hardwoods sites as well but has not been explicitly measured. However, Dress and Boerner (submitted) observed that spring increases and summer decreases in fine root biomass in an Ohio oak-hickory site burned three times over three years were shifted 1-2 months earlier than in unburned controls or sites burned once two years before. The potential impact of ground surface microclimate alteration on belowground ecology in the central hardwoods remains largely unexplored.

Ash Deposition, Dissolution, and Their Effects on Soil Chemistry

The material that remains after fire has consumed part or all of the litter layer is a combination of partially-combusted organic matter and inorganic ash. This inorganic ash is rich in base cations such as Ca, Mg, and K, and is easily dissolved by rainfall. Whether the dissolution of this ash has subsequent effects upon surface soil chemistry depends, to a great extent, on the nutrient status of the soil prior to the fire and the mass of ash-derived nutrients that are added as a consequence of fire.

The impact of ash deposition and dissolution can be most vividly seen in studies of sites in which fire has been both frequent and spatially concentrated. Mikan and Abrams (1995) examined soil chemical properties in an oak forest site in southeastern Pennsylvania that had been used as the charcoal hearth for an iron plantation from 1771-1884. The soils under the former hearths were 0.25 units higher in pH, and base saturation increased from 10% in nonhearth soils to 38% in hearth soils. Furthermore, exchangeable Ca was 12.1X higher and Mg was 4.8X higher in hearth soils than nonhearth soils. Thus, the effects of repeated ash deposition could still be observed even 110 years after the last hearth fire.

A less extreme example comes from a study of the effects of burning of concentrated slash piles after logging in an East Anglia, England oak forest (Jalaluddin 1969). In this situation, the slash was concentrated in 1.5-2.0m diameter piles and burned, with active flames and smoldering lasting >3hr. Samples taken several days after the fires indicated that pH had increased from 6.0 in unburned to 9.0 under ash piles. However, within six months, soil pH under the former slash piles returned to preburn levels. Thus, the effects of fire on soil chemical properties are sensitive to both the amount of ash deposited in a fire and the number of fires that occur at that site over time.

Eivasi and Bryan (1996) have studied the impact of long-term prescribed fire on southeastern Missouri oak-hickory upland flatwoods. This experiment established in 1949, with annual burns and periodic burns (every four years) done in April prior to leaf-out. After 40+ years of burning there were no significant effects of burning on soil pH, Ca, K, or Al, and only a small indication of lowered Mg in annual burns. However, available P was only 24% of control in the annual plots and 35% of control in the periodic burn plots. Thor and Nichols (1973) report similar results from an annual and periodic burning study in the Highland Rim region of Franklin County, Tennessee. After 8 annual or 2 periodic burns, there were no statistically significant differences in pH among treatments. DeSelm et al. (1991) studied the same Tennessee site 27 years after the onset of burning. Again, the pH and availability of K and P of soils in the burned plots did not differ significantly from those of the controls. Thus, in the loess/limestone-derived soils of these two sites, lengthy periods of burning had no striking effect on base cations, but did impact P availability negatively in site with the longer history of burning.

Knighton (1977) studied the effects of 0, 1, 2, or 3 years of annual burning on an oak-hickory forest in central Wisconsin. Availability of Ca and Mg were considerably higher in the soil solution from the burned sites than the control, but with no indication of a cumulative effect over time. The soil solution concentrations of NO_3^- and PO_4^{3-} were also greater in the burned sites, and the average concentrations of both appeared to increase with the number of fires. However, Knighton (1977) points out that the increasing trend was not statistically supportable due to high among-sample variability in soil leachate samples. Blankenship and Arthur (1999b) report an increase of 0.2-0.3 pH units and a transitory increase in available N following a winter prescribed fire in an oak-pine stand in eastern Kentucky. They did not, however, report increases in extractable cations to accompany the increase in pH.

In our continuing study of annual and periodic prescribed fire in southern Ohio we have observed a somewhat different pattern of pH change. Our experimental design is based on four study sites, each of which is divided into three watershed-size treatment units. One of the units within each study area has been burned annually since 1996, a second was burned in 1996 and 1999, and the third remains as an unburned control. In 1995, prior to any burning, soil pH did not vary significantly among the three watershed-scale units in three of our four study areas (Figure 3A; Boerner, unpublished data). In the fourth site (Watch Rock), one unit had significantly greater pH than the other two, although we have still not yet determined why this difference existed. Following 2-4 fires, we now see a significant and positive effect on soil pH among all four study areas pooled and within two of the four individual sites (Figure 3B; Boerner, unpublished data). One site (Young's Branch) has not responded in soil pH to the burning. At Watch Rock, the one treatment unit with the unusually high pH has remained an outlier; however, the other burned unit at Watch Rock is now significantly greater in soil pH than its unburned control. Our study sites differ from those studied by Eivasi and Bryan (1996), Thor and Nichols (1973) and DeSelm et al. (1991) in that the soils of our sites were formed on nutrient and base poor sandstones and shales, not on relatively nutrient-rich loess and limestone as in the other studies. Thus, the impact of the base cations dissolved from the ash deposits is likely to be greater and more easily resolved in a nutrient-poor site such as ours than in the richer limestone and loess sites.

In a study of site preparation burning prior to restoration of Appalachian pine-mixed hardwood forests in North Carolina, Knoepp and Swank (1993) demonstrated that soil NH_4^+ concentrations and N mineralization rates increased after fire. The additional N made available from ash dissolution and from enhanced organic matter mineralization did not, however, result in increased stream water N concentrations; thus, the increased available N remained within the ecosystem.

Despite considerable combustion of surface organic matter by fire, few studies have noted changes in soil organic matter content. Knighton (1977) observed no effect of 1-3 fires on soil organic C in a Wisconsin oak-hickory site, and we have observed only slight increases in soil organic C in our Ohio oak-hickory sites (Boerner et al. 2000b); Boerner and Brinkman, submitted): Even after 40+ yr of burning in a Missouri oak-hickory forest organic C content in the mineral soil did not differ from that of unburned controls (Eivasi and Bryan 1996).

It should be emphasized again at this point that these patterns of effects are specific to dormant season, low intensity fires. The effects of high intensity wildfire may be quite different. For example, soil organic C and N availability were reduced by as much as half following intense wildfire in *Pinus halepensis-Quercus calliprinos* forests in Israel (Kutiel and Naveh 1987)

Biological and Ecological Effects

For convenience, I will review the biological and ecological effects of fire on soil organisms in decreasing size order: from roots to soil animals to microbes. The only study of the effect of fire on root biomass and production in the central hardwoods comes from our own study sites. Dress and

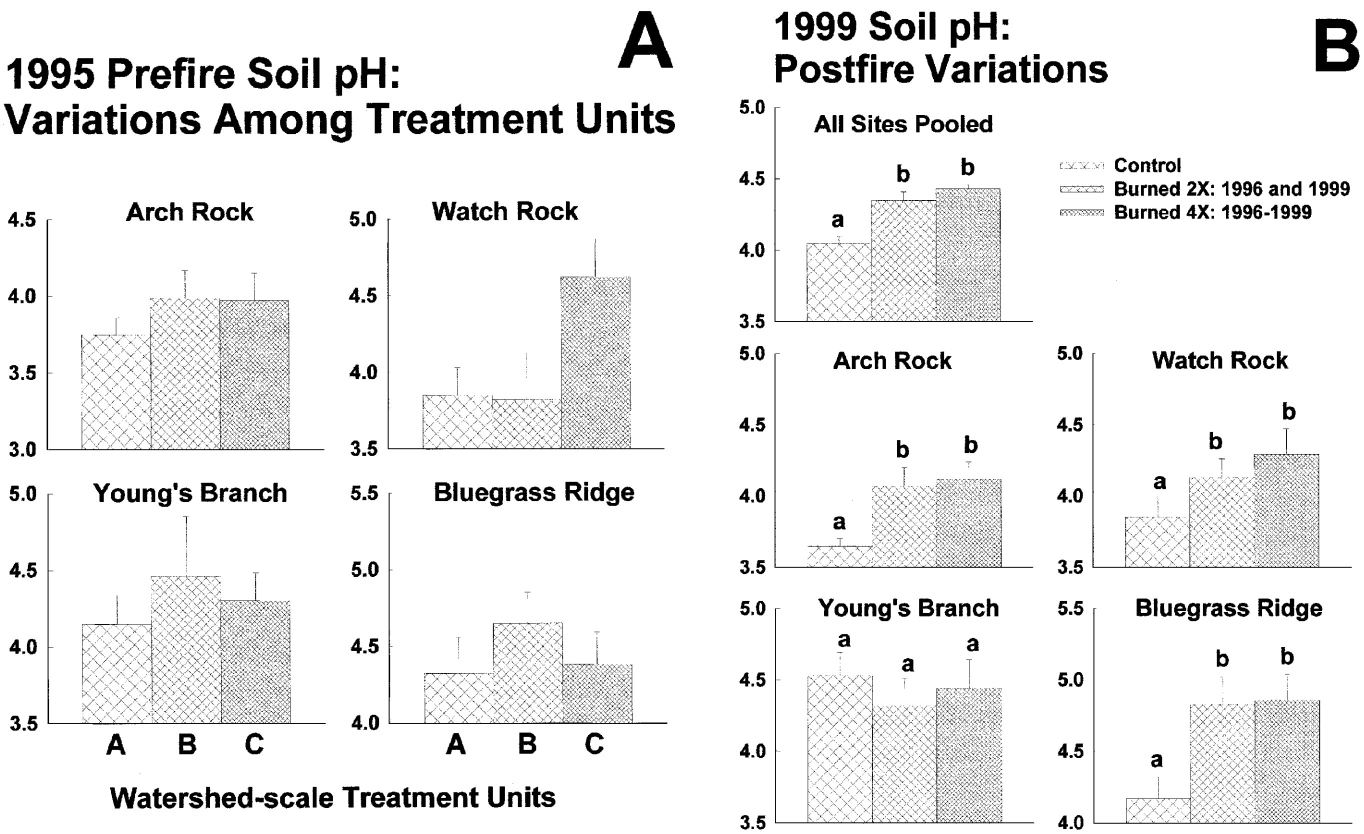


Figure 3.—A-horizon mineral soil pH in forested watersheds in southern Ohio. N=18 for each histogram bar (except for pooled data in B where N=72) with standard errors of the means shown. A. 1995 prefire variations among watershed-scale treatment units within the four study areas. B. 1999 postfire variations among treatment units. Histogram bars labelled with different lower case letters were significantly at p<0.05.

Boerner (in press) determined that both live and dead fine root biomass were significantly lower in a site burned annually for three years than in an unburned control. The variations in root biomass were inversely proportional to variations in soil N availability and N mineralization. This observation was consistent with both general models of the relationship between root biomass/production and N availability (Chapin 1980) and with studies in grasslands that also demonstrated reciprocal responses of roots and N availability to fire (Benning and Seastedt 1997).

The assemblage of small animals in the forest floor and soil of a forest ecosystems is both extremely diverse and critically important. These animals form a complex food web that is responsible for much of the processing of the detritus produced both above ground and below ground. Soil animals such as springtails (collembola), mites, and millipedes are important in breaking down coarse organic matter such as leaves into fine material that is suitable for microbial colonization and decay. Without soil animals present, leaf litter decomposition essentially ceases (Edwards and Heath 1963)

Once again, the database for the central hardwoods is sparse. In our study area, the abundance of oribatid mites in the top 15 cm of soil and forest floor decreases with increasing fire frequency (control, one burn, three annual burns), but the abundance of other mite groups was lowered only in the annual burn site (Dress, unpublished data). Metz and Farrier (1971) monitored the abundance of mites and collembola in the forest floor and soil of a North Carolina pine forest. In the forest floor layers, abundance of both groups decreased with increasing fire frequency. However, when fire was applied only every 4 years there was sufficient time for recolonization and population growth to near control levels. Abundances of collembola and mites in the mineral soil were not affected by either annual or periodic burns in these sites. Thus, on the basis of a scanty database we suggest that animals in the litter are likely to be more affected than those in the mineral soil and that effective repopulation can occur under a periodic but not annual burning regime.

The effects of fire on the soil microbial community can be evaluated by examining directly the effects on individual species or species groups or by determining the effect of fire on microbially-mediated processes such as N mineralization. Jorgensen and Hodges (1970) analyzed the impact of annual and periodic winter burns on the microbial community structure of a loblolly pine *(Pinus taeda)* forest on the South Carolina piedmont. In this study, the last periodic burn was eight years earlier and last annual burn was slightly less than one year earlier. In the mineral soil, none of the burning treatments significantly affected the abundance of either fungi or bacteria+actinomycetes, nor did burning affect the diversity of fungi in either the mineral soil or the forest floor. In the forest floor of the annual burn plots, however, the total abundance of fungi was only 78% of that in the control and the total number of bacteria+actinomycetes was only 18% of the control level. Thus, the microbial abundance in forest floor was strongly reduced by the annual burning. In contrast, microbial abundance in the periodic burn site was greater than in control, by 2.2X for fungi and by 1.2X for bacteria+actinomycetes. So recovery from burn happened in <8yrs and actually produced a forest floor environment more suited to microbial activity than the unburned control.

The potential of fungi present in the mineral soil to facilitate recolonization as the forest floor redevelops after fire is also supported by the work of Tresner et al. (1950) in hardwood forests of southern Wisconsin. They noted that although numbers of fungi decrease from forest floor down to mineral soil, and with depth in mineral soil, species composition does not change, so colonization of disturbed surface layers from deeper soil is possible without changing community structure (Tresner et al. 1950).

A more extreme example of the impact of fire on fungi comes from the work of Jalaluddin (1969) on the microbial ecology of soils under small plots on which concentrated piles of slash had burned and smoldered had continued >3hrs. At one week, three months, and six months post-fire, fungal abundance in the center of the former ash piles were only 3%, 6%, and 11% of that in areas >3m from fire piles. At the edge of the burned areas where mycelial recolonization would have supplemented colonization from new spores, one week, three month, and six month abundances were 17%, 31%, and 43% of those >3 m from the burned areas. Thus both mycelial ingrowth from areas surrounding the fire and new spore colonization occur. However, in this case, recolonization from below was prevented by the impact of the lengthy smoldering of the fire sterilizing the lower soil layers.

Mycorrhizal fungi are key symbionts for virtually all forest plants. There are two major groups of mycorrhizal fungi found in this region. The ectomycorrhizal fungi are a group of higher fungi (basidiomycetes and ascomycetes) which form symbioses with conifers, oaks, hickories, and beech. The arbuscular mycorrhizal fungi are a group of lower fungi (zygomycetes) that form symbioses with herbaceous plants and woody plants other than those that depend on ectomycorrhizae. In western conifer forests, some ectomycorrhizal fungi seem to be sensitive to fire while others are unaffected (Schoenberger and Perry 1982). Thus, in western conifer forests, how seedlings of a host tree species perform in recently burned sites may depend on which fungi that species depends upon to form ectomycorrhizae. Unfortunately, little is known about either the diversity of either group of mycorrhizal fungi in eastern forests or the response of these organisms to fire.

There may also be an interaction between effects on soil animals and effects on microbes. Lussenhop and Wicklow (1984) determined that fungal species diversity in Wisconsin prairie plots subjected to spring burning and raking in a Wisconsin prairie increased by 29% compared to the control. Moreover, the fungal propagules were also less aggregated and more evenly distributed in the burned plots. Lussenhop and Wicklow (1984) feel this change in spatial dispersion was due more to the increase in the abundance of mites and collembola on the burn plots than the actual burning and raking treatment because changes in fungal spatial distribution correlated most closely with changes in fauna.

Other than the study of Jorgenson and Hodges (1970) described above, studies of the direct impact of fire on bacterial abundances and/or community composition in the central hardwoods are uncommon. Blankenship and Arthur (1999b) did find a significant, positive effect of a single prescribed winter burn in an eastern Kentucky oak-pine stand on bacterial biomass and also reported a decrease in the fungal:bacterial biomass ratio following fire. Clearly the impact of fire on the forest floor and soil microbial assemblage continues to be neglected in fire research in this region.

Studies of functional measures of the impact of fire on microbes in the central hardwoods are more common than are direct community and density analyses. Vance and Henderson (1984) in a study of the same Missouri sites later used by Eivasi and Bryan (1996) found that N mineralization was reduced by long-term burning, but possibly not with just a single burn. As there was no change in soil organic C content due to fire, Vance and Henderson (1984) concluded that this long-term change was the result primarily of a change in the quality of the organic C in the soil. In contrast to what Vance and Henderson (1984) concluded, a progressive reduction in microbial activity or biomass is also a possible cause for this long-term reduction in N mineralization, and this is what Eivasi and Bryan (1996) reported 12 years later. Eivasi and Bryan (1996) demonstrated that microbial biomass was reduced relative to the control by 32% in the annual burn plots and by 21% in the periodic burn plots. In addition, Eivasi and Bryan (1996) demonstrated significant, burning-induced reductions in the activity rates of five key enzymes that serve as indicators of microbial activity: Acid phosphatase, a- and b-glucosidase, sulfatase, and urease. Vance and Henderson (1984) also report long-term reductions in tree growth on the burned plots, and attribute this to reduced N mineralization, thus linking the microbial effects to above ground ecological impacts.

In contrast to the Missouri case, studies done in many temperate ecosystem types have demonstrated an increase in N mineralization and N availability after a single or small number of fires (Boerner 1982). These increases are often attributed to the alteration of the organic matter by fire in such a way as to render it more susceptible to microbial decay, to increased microbial activity, and to altered microclimate. For example, Boerner et al. (2000b) demonstrated strong increases in N mineralization in plots burned thrice relative to those in those unburned plots. Plots burned once were intermediate in N mineralization rate. However, this observed increase in the activity of those fungi and bacterial involved in N mineralization probably cannot be generalized to all soil microbial groups. Both Saa et al. (1993) in pine forests and Boerner et al (2000a) in oak-hickory forests report decreased acid phosphatase activity following a single fire.

Summary and Conclusions

Fires in the central hardwoods are typically low in intensity and consume primarily the unconsolidated leaf litter. As long as the fire can move across the open forest floor, soil temperatures generally do not increase enough to cause significant heating-induced mortality of organisms dwelling in the mineral soil. Soils under smoldering piles of woody fuels may, however, be subject to sterilization. Direct N volatilization is probably not an important pathway for fire-related nutrient loss due to low fire temperatures. The microclimate at the forest floor surface is probably affected significantly and this may produce phenological changes in root growth and microbial activity. More research is in this area is warranted. Base cations released from dissolving ash may or may not increase soil pH and cation availability, depending on the nutrient status of the soil and the amount of ash deposited. Nitrogen availability typically increases after one or a small number of fires but may decrease over the long term. Abundances of soil animals in the forest floor are reduced by fire whereas those in the mineral soil are affected little. Recolonization of the redeveloping post-fire forest floor is rapid. Microbial abundances in the forest floor are typically reduced by fire, but rapid recolonization by these groups is also likely, except under smoldering piles of woody fuels. The database for the central hardwoods region is scanty and this area is greatly in need of additional research attention. Based on this scanty database, overall impacts of low intensity, dormant season fires in the central hardwoods is less that the impact on above ground organisms and ecological processes.

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Fire and Understory Vegetation: A Large-scale Study in Ohio and a Search for General Response Patterns in Central Hardwood Forests

Todd F. Hutchinson[1] *and Steve Sutherland*[2]

Abstract

Oak-dominated ecosystems throughout the central hardwood region (CHR) are experiencing a shift in species composition toward more mesic and/or fire-sensitive species such as red maple (*Acer rubrum*), sugar maple (*Acer saccharum*), and blackgum (*Nyssa sylvatica*). Prescribed fire can enhance oak regeneration but few studies have examined the effects of fire on other ecosystem components, including understory vegetation. In 1994, a multi-disciplinary project was initiated to study the use of prescribed fire as a restoration tool in mixed oak ecosystems in southern Ohio. Here, I report on the response of understory vegetation after 3 years of early spring (dormant-season) fires. Each year, from 1995 to 1998, species' frequencies were recorded in 1,728 2m^2 quadrats in four 75 to 100 ha study areas. Each study area included a control, infrequently burned (1X), and frequently burned (3X) unit. In all, 428 vascular plant species were identified. Surface fires have altered the understory environment by greatly reducing the quantity of leaf litter on the forest floor, but light availability has increased only slightly as the tree canopy has remained closed. Several groups of species have responded consistently to fire, including seed bank increasers, tree seedlings, and spring monocots. On control plots, few species have increased or decreased in frequency by more than 5 percent. On 1X plots, 18 species have increased and 5 have decreased; on the 3X plots, 22 species have increased and 18 have decreased in frequency. From 1995 to 1998, species richness (nearly all native species) increased on the burned plots relative to the control plots. This was particularly true at the quadrat level, where richness has decreased slightly on the control plots but increased by 11.4 and 10.4 percent on 1X and 3X plots, respectively. From our research in Ohio and from other studies in the CHR, we conclude that dormant-season prescribed fire may be used to promote oak and pine regeneration while maintaining or increasing the diversity of native understory species.

Introduction

Fire has played an important role in shaping the structure and composition of vegetation in the central hardwoods region (CHR). Fire use by Native Americans is widely believed to have created and maintained many of the more open plant communities in the CHR, including savannas, barrens, and prairies (Baskin and others 1999, Fralish and others 1999, Heikens 1999). Fire history studies using dendrochronology methods support the hypothesis that frequent fire maintained savanna vegetation in the CHR (Guyette and Cutter 1991, Batek and others 1999). Currently, fire is being used in the CHR by land management agencies including the Nature Conservancy, state heritage programs, USDA Forest Service, and the National Park Service to restore and maintain these plant communities, many of which contain threatened and endangered species (e.g., Bender, this volume)

It is also hypothesized that frequent fire maintained the dominance of oak (Abrams 1992, Abrams, this volume) and pine (Waldrop, this volume) in the presettlement CHR landscape. The absence of fire has been implicated in the widespread abundance of shade-tolerant and fire-sensitive tree species in the regeneration layer of oak forests in the eastern United States (Lorimer 1984, Loftis and McGee 1993, Lorimer and others 1994). Prescribed fire has been used to regenerate oaks with varying degrees of success (Wendel and Smith 1986, Barnes and Van Lear 1998, Elliot and others 1999). However, shelterwood harvests followed by high intensity prescribed fires have improved oak regeneration on productive upland sites in Virginia (Brose and others 1999, Van Lear and Brose, this volume).

In addition to understanding the effects of fire on tree regeneration, managers need to understand the effects of fire on the nontree components within the CHR. In this paper I examine the effects of fire on the understory vegetation layer (herbaceous and woody species) in forests in and around the CHR by presenting data from previous studies and an ongoing study of prescribed fire in mixed-oak forests in southern Ohio.

Effects of Fire on Understory Vegetation in the Central Hardwood Region

The understory vegetation layer in central hardwood forests is composed of several major life forms: graminoids (grasses, sedges, and rushes), forbs (nongraminoid herbaceous plants), shrubs, woody vines, and seedlings of tree species. Herbaceous species, particularly perennial forbs, comprise the majority of the species pool in CHR forests (Hutchinson and others 1999). The cover and abundance of herbaceous species varies annually due to climatic conditions (Rogers 1983) and throughout the growing season as spring ephemerals, summergreen species, and late-summer species emerge and reach maximum biomass at different times (Bratton 1976, Goebel and others 1999).

Understory composition and richness are strongly related to topographic moisture and fertility gradients (Huebner and others 1995, Olivero and Hix 1998, Hutchinson and others

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[1] Research Ecologist, USDA Forest Service, Northeastern Research Station, 359 Main Road, Delaware, OH 43015

[2] Ecologist, The Nature Conservancy, Ohio Chapter, 6375 Riverside Drive, Suite 50, Dublin, OH 43017
Current address: USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, P.O. Box 8089, Missoula, MT 59807

1999), soil parent material (Howell and Vankat 1981), and land-use history (Motzkin and others 1996), including fire history (Swan 1970). Mean species richness of 14.5 species per m^2 has been reported in mixed mesophytic forests in the southern Appalachians (Duffy and Meier 1992). In the study described here, mean richness per 2 m^2 ranged from 5.9 to 29.6 and averaged 16.7 (Hutchinson and others 1999).

Due to the limited number of fire studies in the CHR, it is useful to first examine the long-term effects of fire on understory vegetation from 2 field studies in southeastern coastal plain pine forests. After 43 years of fire treatments in loblolly pine (*Pinus taeda*) stands on the Santee Experimental Forest, which included control (no fire), and both periodic and annual winter and summer burns, the structure, composition, and diversity of understory vegetation varied primarily by fire treatment (Lewis and Harshbarger 1976, White and others 1991). There was a gradient from shrub and hardwood domination on control plots to herbaceous (forb and graminoid) dominance in the burned plots. Maximum abundance and diversity of native herbaceous species occurred in plots burned annually (White and others 1991). Brockway and Lewis (1997) reported that longleaf pine (*P. palustris*) stands burned frequently over 4 decades had a much more diverse and productive herbaceous understory than control stands, which developed a dense shrub layer of gallberry (*Ilex glabra*) that reduced diversity.

In a long-term study within the CHR, DeSelm and others (1973) and DeSelm and Clebsch (1991) documented the effects of annual and periodic fires on vegetation over a 25-year period (1965-89) in an oak barrens community in middle Tennessee by recording vegetation cover annually using the point-intercept method. The burned plots maintained high forb and graminoid cover; forb cover averaged 46.5 and 28.3 percent on annually and periodically burned plots, respectively, and graminoid cover averaged 59.6 and 48.3 percent on annual and periodic burn plots. In sharp contrast, the control plots rapidly lost herbaceous cover, which had decreased to less than 1 percent by 1976 as woody plants invaded and shaded the ground-layer vegetation.

In more closed-canopied oak and pine forests, studies on fire effects have generally consisted of 1 to several fires. The response of understory vegetation in and around the CHR is summarized in Table 1.

Species groups that have increased in frequency, abundance, or cover following fire include annual forbs such as fireweed (*Erechtites hieracifolia*) (Masters 1991, Dolan 1994, Sparks and others 1998), legumes (e.g., *Desmodium* spp., *Lespedeza* spp) (Ducey and others 1996, Sparks and others 1998), warm-season grasses, including *Panicum* spp. and little bluestem (*Schizachyrium scoparium*) (Dolan 1994, Sparks and others 1998, Elliot and others 1999), and late-summer forbs such as the asters (*Aster* spp.), goldenrods (*Solidago* spp.), and sunflowers (*Helianthus* spp.) (Swan 1970, Wilhelm and Masters 1994, Sparks and others 1998). The most consistent species group was the brambles (*Rubus* spp.), which increased via abundant germination in almost every study (e.g., Swan 1970, McGee and others1995, Nuzzo and others 1996). For all of these species, increased light availability following fire likely was the most important response factor, i.e., it increased seed germination (Pickett and Baskin 1973, Baskin and Baskin 1988) and/or growth and reproduction (Pitelka and others 1985).

Most species that decreased consistently were tree seedlings, including red maple (*Acer rubrum*) and black cherry (*Prunus serotina*) (Swan 1970, Cole and others 1990), and shrubs and vines, including Virginia creeper (*Parthenocissus quinquefolia*) and greenbriar (*Smilax rotundifolia*) (Wilhelm and Masters 1994, Nuzzo and others 1996; Arthur and others 1998). The cover of mountain laurel (*Kalmia latifolia*), an evergreen shrub, decreased in several studies, but resprouted prolifically (McGee and others 1995, Ducey and others 1996, Elliot and others 1999).

Herbaceous cover generally increased following fire (McGee and others 1995, Nuzzo and others 1996) while the cover (Nuzzo and others 1996; Elliot and others 1999) and density (Schwartz and Heim 1996) of woody vegetation decreased (Table 1). However, due to resprouting, understory woody cover also has exceeded pre-burn levels within several years (McGee and others 1995, Arthur and others 1998). When mixed oak-pine stands in eastern Oklahoma were thinned and burned, herbaceous biomass increased significantly (Masters and others 1993).

In nearly all of the fire studies in the CHR, species richness or diversity increased following fire, though the magnitude and duration of the responses were highly variable (Table 1). In a sand forest in Illinois, species richness increased from 6.2-7.0/ m^2 to 9.8-10.8/ m^2 following the first of 3 fires, due to an increase in herbaceous species, particularly annuals (Nuzzo and others 1996). However, the strong effects were short-lived as richness subsequently declined after the first fire-free year. In western Arkansas, species richness increased significantly following a single dormant-season fire in restored shortleaf pine (*Pinus echinata*) forests (Sparks and others 1998). The stands gained 14-16 species and more than 3 species were gained per m^2 (Sparks and others 1998). More than 8 years following 2 fires in a transition oak-northern hardwood forest, forb richness increased significantly over pre-burn levels in 2 stands, 26 to 44 and 23 to 27 species/4 m^2, respectively (McGee and others 1995). Other studies reported increased species richness following fire (Cole and others 1990, Wilhelm and Masters 1994, Taft and others 1995, Arthur and others 1998), though Dolan (1994) found no change in richness and Schwartz and Heim (1996) reported a decrease (Table 1).

Fire intensity and season are other important factors that influence the response of vegetation. Ducey and others (1996) and Elliot and others (1999) reported that species richness increased more following high-intensity fires than in areas that burned at lower intensities. In Arkansas, dormant-season burns in March and April increased the abundance of several *Panicum* species but a September fire reduced *Panicum* abundance (Sparks and others 1998). After an

Table 1.—Summary of fire studies examining the response of understory forest vegetation in and around the central hardwoods region

Overstory dominant	Location	Duration (pre-burn data)	Fire Treatments	Herbaceous response	Author(s)
Post oak Shortleaf pine	OK	5 years (yes)	1 or 3 fires Winter	richness +	Masters 1991 Masters and others 1993
Shortleaf pine	AR	2 years (yes)	1 or 2 fires Mar-Apr; Sep	richness + density + / -	Sparks and others 1998
Mixed: oak, pine, maple, hickory	NC	2 years (yes)	1 fire Apr	richness + cover +	Elliot and others 1999
Scarlet oak Chestnut oak	KY	4 years (no)	1 or 2 fires Mar	richness + cover +	Arthur and others 1998
Post oak	IN	7 years (yes)	1 fire Apr	Richness: No effect	Dolan 1994
Mixed oak	NY	1 year (no)	1 wildfire Apr-May	# increasers > # decreasers	Swan 1970
Red oak, white oak, white pine	CT	1 year (no)	1 fire Apr	richness + density +	Ducey and others 1996
Red oak, white oak	NY	12 years (yes)	1 or 2 fires Apr	richness + height +	McGee and others 1995
Black oak	IL	5 years (yes)	3 fires (dormant) autumn and spring	richness + cover +	Nuzzo and others 1996
White oak, red oak, black walnut	IL	4 years (yes)	1 fire Mar or May	richness - (May burn only)	Schwartz and Heim 1996
Mesic woodland	IL	5 years (no)	4 fires Spring	richness +	Wilhelm and Masters 1994

initial depression in species richness following both March (dormant season) and May (growing season) fires in Illinois, richness soon recovered following the March but not the May burns (Schwartz and Heim 1996).

Fire has not facilitated the invasion of exotic species in CHR, possibly because most of the areas that have been burned and studied quantitatively have been relatively xeric and undisturbed, and thus not highly invasible (Rejmanek 1989). In more mesic and disturbed forests where garlic mustard (*Alliaria petiolata*) has invaded, fire moderately reduced both its cover and abundance (Nuzzo and others 1996, Schwartz and Heim 1996).

Effects of Fire on Understory Vegetation in Ohio Oak Forests

Witness tree data from land surveys indicates that the unglaciated Allegheny Plateau region of southeastern Ohio was composed primarily of mixed oak forests just prior to European settlement (ca. 1800) (Gordon 1969). In the southern portion of this region, many of the oak forests were harvested repeatedly from 1850 to 1900 to supply charcoal for the numerous iron furnaces in the region (Stout 1933). In secondary forests dating to the end of the iron-producing period, oaks, primarily white (*Quercus alba*), chestnut (*Q. prinus*) and black (*Q. velutina*), continue to dominate the overstory but the midstory and understory layers are dominated by shade-tolerant species such as red maple (*Acer rubrum*), sugar maple (*Acer saccharum*), blackgum (*Nyssa sylvatica*), and beech (*Fagus grandifolia*) (Figure 1). Similar trends have been reported for numerous oak forests in the eastern United States (Abrams 1992). On the basis of these forest trends, a multidisciplinary study was established to examine prescribed fire as a tool to restore structure, composition, and function to oak forests in southern Ohio.

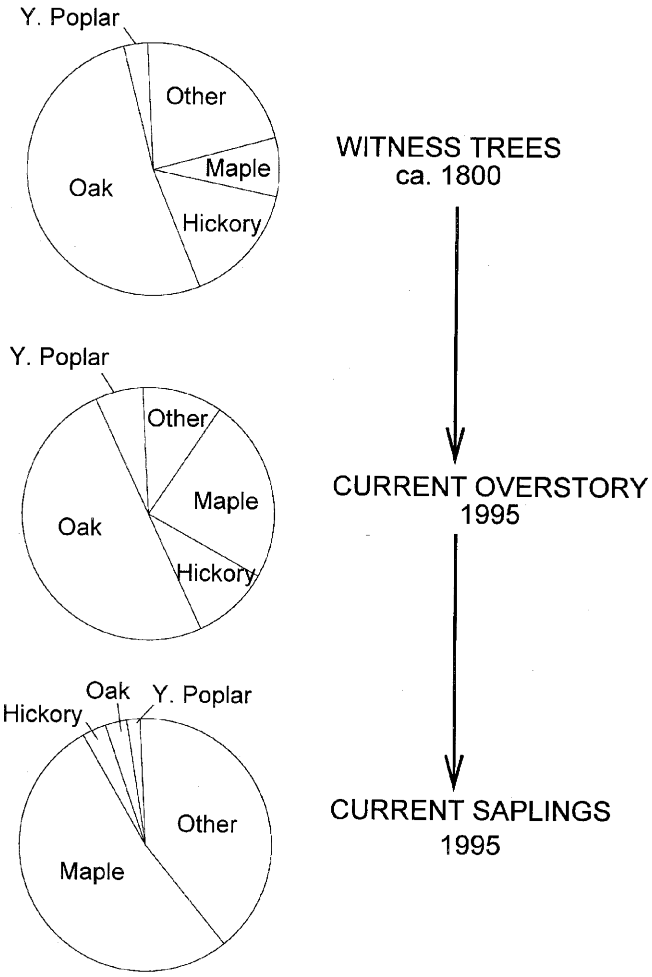

Figure 1.—Forest composition trends in southern Ohio. Witness tree data are from townships in and adjacent to the study areas. Data for the Vinton County areas are from Beatley (1959). Current overstory (trees > 10 cm dbh) and sapling (trees 1.37 m height to 9.9 cm dbh) data were collected from 108 vegetation plots The primary "other" species for the saplings are blackgum and beech.

Methods

Study Areas and Experimental Design

In 1994, four study areas (75 to 100 ha each) were established, two in Vinton County (Arch Rock, 39°12' N, 82°23' W; Watch Rock, 39°11" N, 82°22' W), and two in Lawrence County (Young's Branch (38°43' N, 82°31' W; Bluegrass Ridge (38°36' N, 82°31' W). The sites were dissected, underlain by sandstones and siltstones, and had moderately acidic siltloam alfisols with low water-holding capacity. The study areas have been relatively undisturbed since clearcutting 80 to 120 years ago, and were dominated by oak. Average tree basal area was 24 to 28 m^2/ha (Hutchinson and others 1999). Three fire treatment units were established within each area. Control units were not burned, infrequent units were burned once (1X) in March-April 1996, and frequent units were burned three times (3X) in March-April, 1996, 1997, and 1998. The surface fires generally were low intensity, with flame lengths less than 50 cm and temperatures averaging 89.6°C (193.2°F) at a height of 25 cm over the 3-year period (Hutchinson, unpublished data). In the uplands of each unit, nine 50 x 25m (0.125 ha) vegetation plots (N = 108 total plots) were stratified by a GIS-derived integrated moisture index based on a suite of landscape features (Iverson and others 1997).

Field Data Collection

The frequency of understory species was recorded in four 2 x 1 m quadrats along each of four transects spaced 5 m apart within a 25 x 25m portion of each plot (16 quadrats per plot). Each year, three quadrats per transect were located randomly and one quadrat was located permanently at the midpoint of each transect. Here, data from random and permanent quadrats were combined. Vegetation was sampled twice each year (May and September) from 1995 (pre-burn) to 1998. Herbaceous plants were recorded on a presence/absence basis in each quadrat to obtain frequency. For each species, the maximum frequency was recorded for the year. For understory individuals of tree species, we recorded the presence of each species in each of four size classes: < 30 cm height, 30 cm height to 2.99 cm diameter at breast height (dbh), 3.0 to 9.9 cm dbh, and ≥ 10 cm dbh.

Results and Discussion

Following the first fires, 37 percent of the small saplings (1.37 m height - 2.9 cm dbh), 9 percent of the medium saplings (3.0 - 5.9), and 4 percent of the large saplings (6.0 - 9.9) had been topkilled though most stems resprouted (Sutherland and others 1997). After three fires, nearly 95 percent of the small saplings had been topkilled (Sutherland and Hutchinson, unpublished). However, few trees greater than 10 cm dbh have been topkilled (D. Yaussy, USDA Forest Service, unpublished) and open sky remained below 6 percent after two fires on the 3X plots (Robison and McCarthy 1999). Leaf litter consumption ranged from 28.5 to 80.4 percent during the first fires in 1996 (Boerner and others, in press). Litter mass has continued to decrease on frequently burned plots, while increasing to pre-burn levels by 1998 on the 1X plots (Hutchinson, unpublished).

Over the 4-year period, we recorded 428 vascular plant species, composed of 254 perennial forbs, 59 graminoids, 45 trees, 30 shrubs, 24 annual forbs, 11 woody vines, and 5 monocarpic perennial forbs. While the frequency of most species did not change significantly following prescribed fire, the following groups exhibited distinctive patterns of change.

Seed Bank Increasers

Several species increased in frequency following abundant germination from the seed bank. Fireweed (*Erechtites hieracifolia*) and yellow-poplar (*Liriodendron tulipifera*) had the greatest increases in frequency on the burned plots (Figure 2). Prior to burning, fireweed had a mean frequency of less than 10 percent per plot; following the first fires in 1996, it increased to nearly 80 percent frequency Presumably, most of the germination was from the buried seed pool as fires would have consumed the previous autumn's small, wind-dispersed seeds. On the 1X plots, the frequency of fireweed returned to pre-burn levels following 2 years without fire; frequency remained high (> 55 percent) on 3X plots. Like many annuals, fireweed probably requires light for germination (Pickett and Baskin 1973, Baskin and Baskin 1988). Disturbances that remove litter and expose mineral soil are required to maintain fireweed at high frequencies in standing vegetation. Yellow-poplar, which produces seeds that are viable in the seed bank for 4 to 7 years (Burns and Honkala 1990), increased in seedling frequency from less than 25 percent prior to burning to nearly 80 percent on burned plots following the first fires. In contrast to fireweed, the frequency of yellow-poplar seedlings then decreased to approximately 50 percent on the 1X and 3X plots from 1996 to 1998. On the 1X plots, the decrease likely was caused by seedling mortality due to low light levels (Burns and Honkala 1990). On the 3X plots fire likely killed most of the seedlings that had germinated, though new germination maintained relatively high frequencies of seedlings.

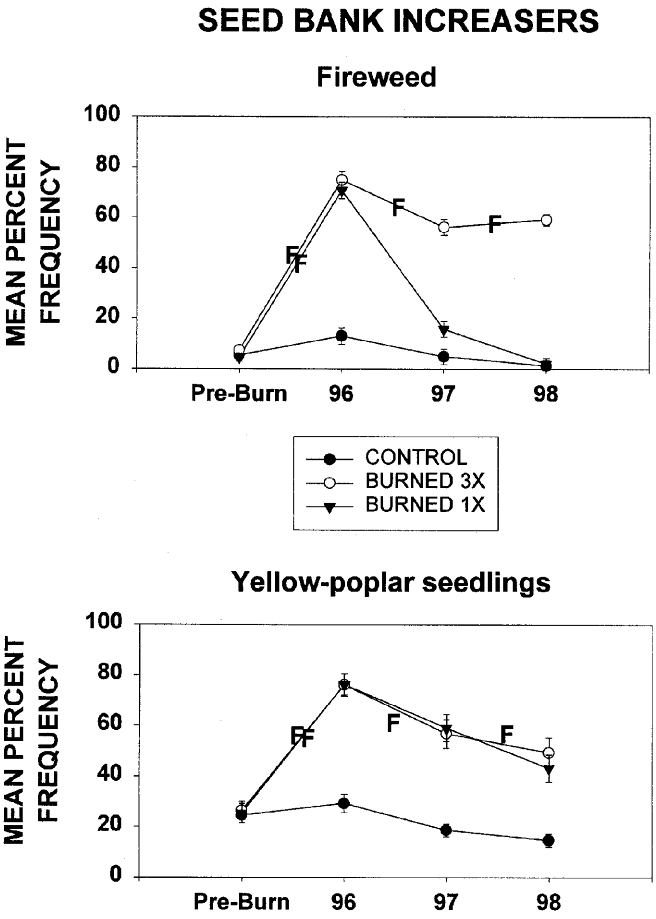

Figure 2.—Frequency data for fireweed and yellow-poplar seedlings. Mean frequencies were calculated as the mean number of quadrats per plot in which the species occurred for 36 plots per year in each of three fire treatments (fires are indicated with an F).

Other species in the pre-burn standing vegetation that increased in frequency by more than 10 percent on the burned plots included Virginia copperleaf (*Acalypha virginica*), sedges (*Carex* spp.), white snakeroot (*Eupatorium rugosum*), sweet-scented bedstraw (*Galium triflorum*), panic grasses (*Panicum boscii*, *P. latifolium*), brambles (*Rubus* spp.), sassafras (*Sassafras albidum*), violets (*Viola* spp.), and wild grapevine (*Vitis* spp.) Smooth sumac (*Rhus glabra*), which was absent from the pre-burn standing vegetation, increased in frequency to 11.3 percent on the 3X plots by 1998. Several species not recorded in the pre-burn standing vegetation germinated following fire, but remained at frequencies under 10 percent. These species included sensitive plant (*Chamaecrista nictitans*), horse-weed (*Conyza canadensis*), late eupatorium (*Eupatorium serotinum*), upland boneset (*E. sessilifolium*), cudweed (*Gnaphalium obtusifolium*), beggar's-lice (*Hackelia virginiana*), indian-tobacco (*Lobelia inflata*), pokeweed (*Phytolacca americana*), dwarf sumac (*Rhus copallina*), fire-pink (*Silene virginica*), and *Sphenopholis nitida,* a grass. Of several exotic species that appeared after the fires, only black bindweed (*Polygonum convolvulus*) was found in more than 1 percent of quadrats within burned treatments (2.4 percent in 1998).

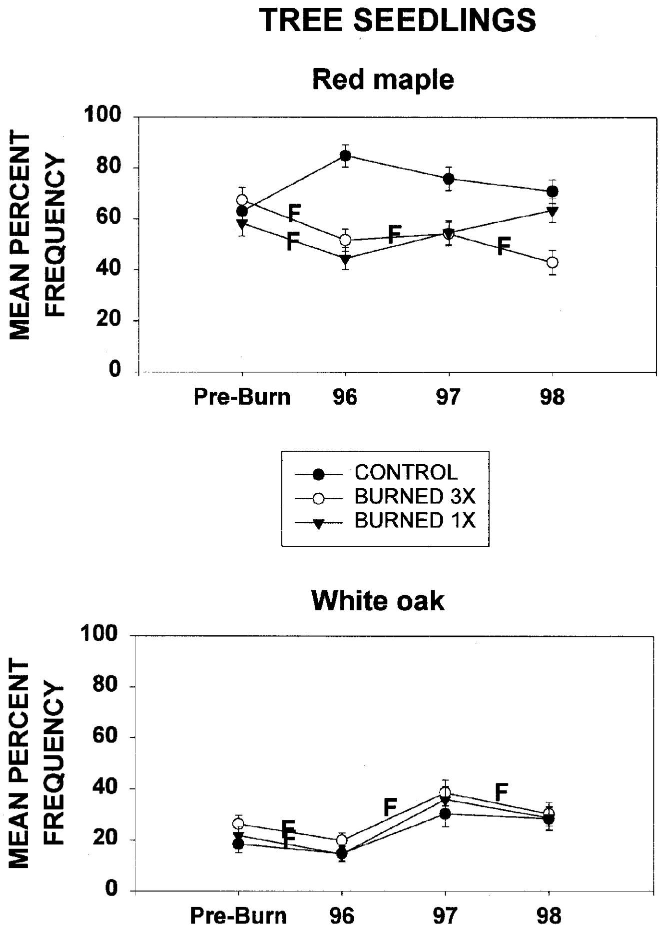

Figure 3.—Frequency data for red maple and white oak seedlings. Mean frequencies were calculated as the mean number of quadrats per plot in which the species occurred for 36 plots per year in each of three fire treatments (fires are indicated with an F).

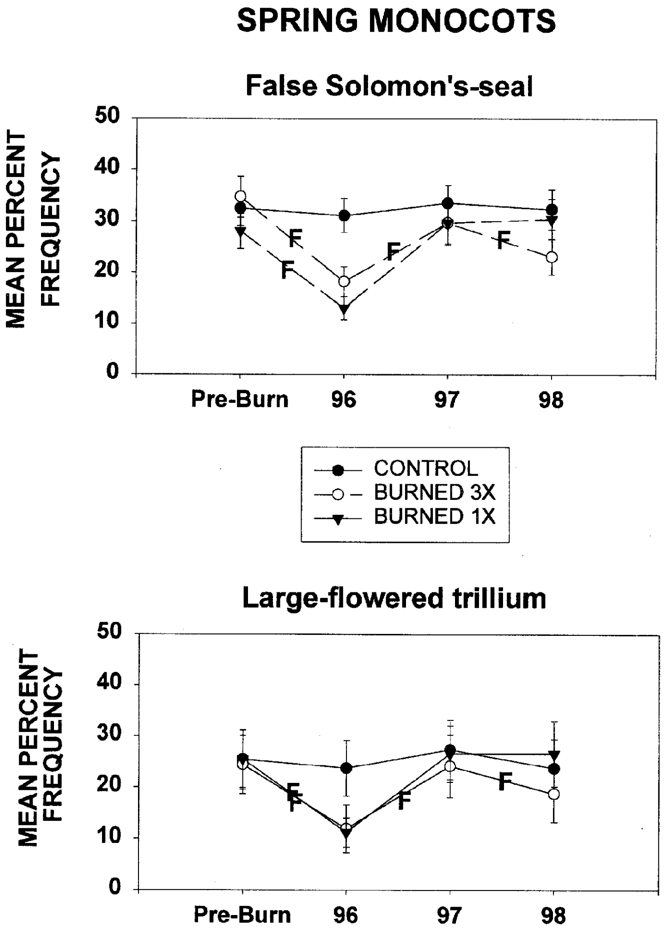

Figure 4.—Frequency data for false solomon's seal and large-flowered trillium. Mean frequencies were calculated as the mean number of quadrats per plot in which the species occurred for 36 plots per year in each of three fire treatments (fires are indicated with an F).

Tree Seedlings

The effects of fire on tree seedlings varied by species, as shown by the contrasting responses of red maple and white oak (Figure 3). From 1995 to 1998, the frequency of red maple seedlings increased from 63.0 to 70.8 percent on control plots, but decreased from 67.4 to 43.1 percent on the 3X plots. On the 1X plots, the frequency of red maple seedlings decreased from 58.4 to 44.6 percent following the fire but increased to 63.4 percent in 1998 after 2 years without fire. Although larger red maple seedlings sprout following topkill, three fires reduced frequencies substantially. By contrast, seedlings of white oak, which was the most abundant overstory species and the most frequent oak species in the seedling layer, exhibited a neutral response (< 5 percent frequency shift) to fire (Figure 3). White oak seedlings increased in frequency in all treatments following a mast year in 1996. In 1998, frequencies were nearly identical across treatments (28.3 to 30.2 percent). Other fire studies have shown similar responses of red maple and oaks (e.g., Kruger and Reich 1997).

Other tree species that decreased in seedling frequency by more than 5 percent from 1995 to 1998 on the 3X plots were flowering dogwood (*Cornus florida*, 17.0 percent), American ash (*Fraxinus americana*, 15.8), sugar maple (*Acer saccharum*, 10.4), slippery elm (*Ulmus rubra*, 8.2), hop-hornbeam (*Ostrya virginiana*, 7.5), and black cherry (*Prunus serotina* 6.9). In addition to white oak, common seedlings that showed a neutral response to fire on the burned plots included scarlet oak (*Quercus coccinea*), chestnut oak (*Q. prinus*), red oak (*Q. rubra*), black oak (*Q. velutina*), pignut hickory (*Carya glabra*), bitternut hickory *(C. cordiformis*), redbud (*Cercis canadensis*), hawthorn (*Crataegus spp.*), and blackgum (*Nyssa sylvatica*).

Spring Monocots

Several monocots that emerge early in the spring decreased in frequency on the burned plots, particularly in years when the fires were conducted later in the spring. Both false Solomon's seal (*Smilacina racemosa*) and large-flowered trillium (*Trillium grandiflorum*) decreased in frequency by

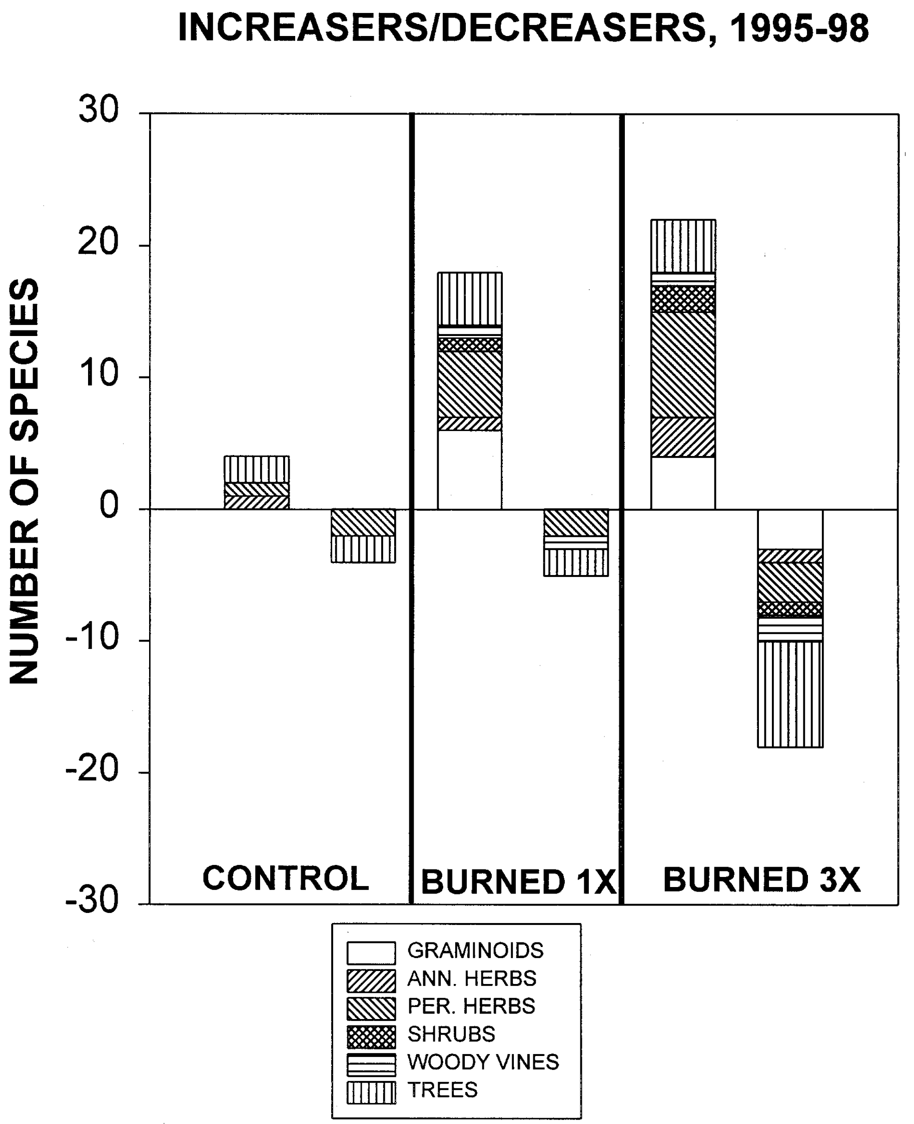

Figure 5.—Species that increased or decreased in mean frequency by more than 5 percent from 1995 to 1998 in three fire treatments.

12.5 to 16.5 percent following the first fires (Figure 4). The 1996 fires were conducted later (through April 22) than in subsequent years due to a wet spring. However, in both fire treatments, these species returned to pre-burn frequencies in 1997 following another fire on the 3X plots and no fire on the 1X plots. The 1997 burns were completed by April 3, suggesting that dormant-season fires have a negative impact on these species only if conducted when they begin to emerge. Damage to the leaf tips of several monocot species has been observed frequently in the study areas (personal observation). From 1997 to 1998 the frequency of false solomon's seal and trillium decreased by 6.6 and 5.4 percent, respectively, suggesting that the fires occurred at a phenological stage intermediate to 1996 and 1997 (Figure 4). Solomon's seal (*Polygonatum biflorum*) and bellwort (*Uvularia perfoliata*) also decreased in frequency on burned plots.

Summary of Increasers and Decreasers

To examine the effects of fire on the entire species pool, we classified species as increasers or decreasers if their frequency shifted by more than 5 percent from 1995 (pre-burn) to 1998. On the controls, few species shifted in frequency by more than 5 percent (Figure 5). On the 1X plots, there were 18 increasers, of which 11 were graminoids and perennial forbs, and 5 decreasers. On the 3X plots, there were 22 increasers but also 18 decreasers, most of which were woody species (11), including 8 tree species. These results indicate that the composition of the 3X plots has changed more than the1X plots, primarily due to the reduced frequencies of several tree species. Despite the greater change on the 3X plots, most species have not shifted more than 5 percent following dormant-season prescribed fires. This suggests that perennial herbaceous species have not been damaged directly by fire, and have

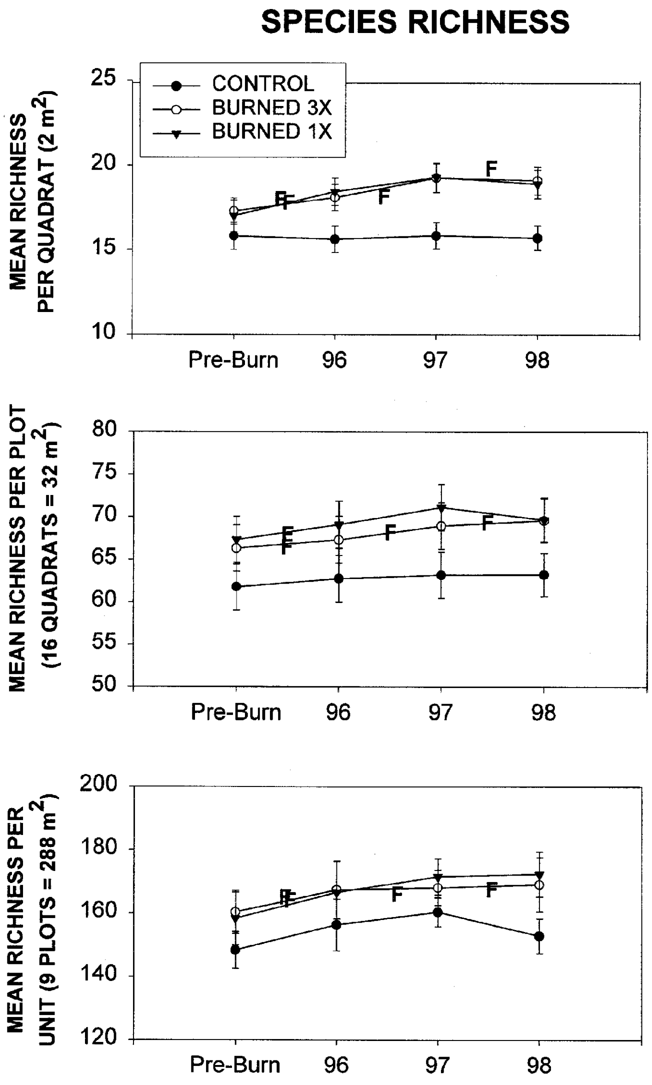

Figure 6.—Mean species richness values for the three fire treatments at three spatial scales of observation (fires are indicated with an F).

emerged in numbers similar to those on the controls. Also, most woody species that were topkilled by fire have resprouted. Although there have been several seed bank increasers, most species have not exhibited abundant germination following fires.

Species Richness

From 1995 to 1998, species richness per quadrat increased by 11.4 percent on 1X plots and 10.4 percent on 3X plots. Over the same period, richness decreased by less than 1 percent on the controls (Figure 6a). Species richness per plot (based on 16 quadrats) also increased more on the burned plots relative to the controls, but only by several species; richness increased by 2.4 percent on control plots, 3.4 percent on 1X plots and 5.0 percent on 3X plots (Figure 6b). When richness was summed for the nine plots in each unit, richness increased in all treatments from 1995-98, but to a greater degree on the burned plots. Richness per unit increased by 3.4 percent on control plots, 5.6 percent on 3X plots, and 8.9 percent on 1X plots. (Figure 6c).

The increases in species richness on the burned plots are subtle, but our results indicate that fire has not depressed richness in these oak forests over the initial 4-year period of this study. Our results also underscore the need for pre-treatment data when conducting fire studies. Our control units were less species rich prior to treatment at all three scales of observation. Had we examined only post-burn data, the presumed effects of fire on increased species richness would have been exaggerated.

Conclusions

Because trees dominate the structure of CHR forests and have high economic value, most research and discussion on fire effects in this region has focused on tree regeneration. However, plant diversity in the CHR region is concentrated in the understory layer, particularly among herbaceous species. It is these plants that also provide habitat and food for diverse assemblages of fauna. There is growing evidence that plant diversity directly affects the productivity and sustainability of ecosystems (Tilman and others 1996). From studies in the CHR it is clear that dormant season fire has no negative impact on plant diversity. In fact, most studies show small to moderate increases in diversity as fire creates an improved regeneration niche for some species (Table 1).

Our Ohio study showed that prescribed fire has both direct and indirect effects on undestory vegetation. Fire directly affects species composition by topkilling most of the smaller woody stems. The relative frequencies of woody species then shift due to variation in resprouting ability; we reported a decrease in the relative frequencies of maple, dogwood, ash, and elm seedlings to those of the oaks and hickories. Fire also directly damaged several herbaceous species (spring monocots), causing reduced frequencies.

Indirectly, fire caused a shift in species composition by greatly reducing leaf litter and exposing mineral soil, which facilitated the abundant germination of several species from the seed bank and the addition of species to the standing vegetation. Other indirect fire effects that may affect species composition include a subtle increase in light availability throughout the growing season, an increase in soil nutrients and pH from leaf litter ash (see Boerner, this volume), and a decrease in soil moisture through increased evaporation.

It is clear that fire suppression since the 1940's has resulted in significant changes in CHR forests. There is a need for long-term research on the effects of fire on the understory vegetation in these forests and for additional population-level studies of fire response by individual species. We also need to understand how the effects of fire and fire suppression on understory vegetation are linked to physical and biological patterns and processes (e.g., light availability and nutrient cycling) and to the population dynamics and behavior of animals, including herbivores, pollinators, and microbial communities.

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Fire and Birds in the Central Hardwood Landscape

R. Todd Engstrom[1]

Abstract

Fire is often viewed as destructive of wildlife and wildlife habitat, because it can cause dramatic change. The perception of catastrophic fire that burns up forests and kills wildlife has been fostered to enlist public support for suppressing fires. A message that is much more difficult to convey is that when fire is removed from the landscape important wildlife habitat can be slowly and subtly eliminated from ecosystems that depend on the frequent occurrence of fire.

Many fire-dependent communities—shortleaf pine forests, barrens, glades, prairies, balds, and oak savannas—are embedded in the extensive and diverse central hardwood landscape. Approximately 232 species of birds (119 breeding, 44 wintering, and 69 year-round residents) reside in this landscape. Some of the rarer bird species in the region are closely tied to fire-dependent habitats. The status of the Red-cockaded Woodpecker (*Picoides borealis*) is tenuous in pine forests of Kentucky and Oklahoma where fire has been suppressed for much of the century. Efforts to reintroduce fire to restore the open-structured, pine-dominated forests are hopeful signs for the species. Grassland birds are one of the most rapidly declining groups of species in North America. Habitat conversion has eliminated many of the grassland communities within the Central Hardwoods region, but plant succession in the absence of fire has also degraded habitat suitability for many grassland birds. Numerous preserves are being established to retain the rare biota of these grasslands. Many oak-dominated forests and oak woodland- savanna mosaics are slowly shifting to greater proportions of trees more associated with mesic conditions, such as maples (*Acer* sp.), and canopy closure. These shifts in structure and composition bring attendant changes in the avian community. Fire—with proper consideration of the potential effects on small populations of birds and invertebrates and the tradeoffs in bird species that will be favored—is an essential tool for habitat maintenance for some of the rare and declining birds in the Central Hardwoods region.

Introduction

A diverse array of vegetation communities that have experienced frequent fire for millennia are scattered throughout the extensive Central Hardwoods forest region (Hicks 1998; Braun 1950; Braun 1950; Hicks 1998; Anderson and others 1999). Some of these communities were fragmented and highly dispersed (e.g., balds) and some were extensive (e.g., the Kentucky barrens). The structure and plant composition of some of these communities was largely shaped edaphic conditions (Quarterman and others 1993), but fire played an important role to arrest succession and control brush and saplings as soils developed (Deselm and Murdock 1993). Lightning-started fires played a role in creating some of these habitats, but Native Americans undoubtedly used fire frequently and pervasively to create the open habitats found by early European colonists (Bonnicksen and others 1999; Askins 1999). Timber harvest, fire suppression, and plant succession have dramatically changed the composition of some of these plant communities, and, in the process, the avian comunities. Excellent descriptions of some of the grassland preserves in the Central Hardwoods region (e.g., Tennessee) can be found at: http://www.tnc.org/infield/State/Tennessee/

The purpose of this paper is to provide a brief overview of the avifauna of the region, including species that have become extinct or extirpated, to explore the bird species composition of three general vegetation communities that were shaped by frequent fires—shortleaf pine (*Pinus echinata*) forests, grassland/shrubland communities (balds, barrens, and glades), and open oak woodlands and savannas—and to discuss the effects on birds of use of prescribed fire in the management and restoration of these communities.

The Central Hardwoods Avifauna

Hicks (1998) uses four of Braun's (1950) forest types to define the Central Hardwoods region: the mixed mesophytic, western mesophytic, oak-hickory, and oak-chestnut forests. This large region extends over all or part of 35 states (Figure 1). I obtained a list of the bird species within this region by comparing distribution maps of bird species (Dunn 1987) to the map of the Central Hardwoods. Bird species whose winter or breeding season distributions overlapped with the outline of the Central Hardwoods region were included; migrant and transient birds were not.

Approximately 232 bird species winter or breed within the Central Hardwoods region (Appendix); all scientific names follow the American Ornithologists' Union (1998). Sixty-nine (30%) of these species occur year-round, 44 (19%) occur in the winter, and 119 (51%) occur in the breeding season only.

Four bird species that were found in the Central Hardwoods historically, Passenger Pigeon (*Ectopistes migratorius*), Carolina Parakeet (*Conuropsis carolinensis*), Ivory-billed Woodpecker (*Campephilus principalis*), and Bachman's Warbler (*Vermivora bachmanii*), have become extinct. The Passenger Pigeon and Bachman's Warbler may have been affected by changes in fire regimes in the Central Hardwoods region, although it seems likely that outright habitat loss was more important in their decline than habitat degradation through fire exclusion. For example burning in forests by Native Americans kept forest structure open and

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[1] Tall Timbers Research Station, 13093 Henry Beadel Road, Tallahassee, FL 32312-0918

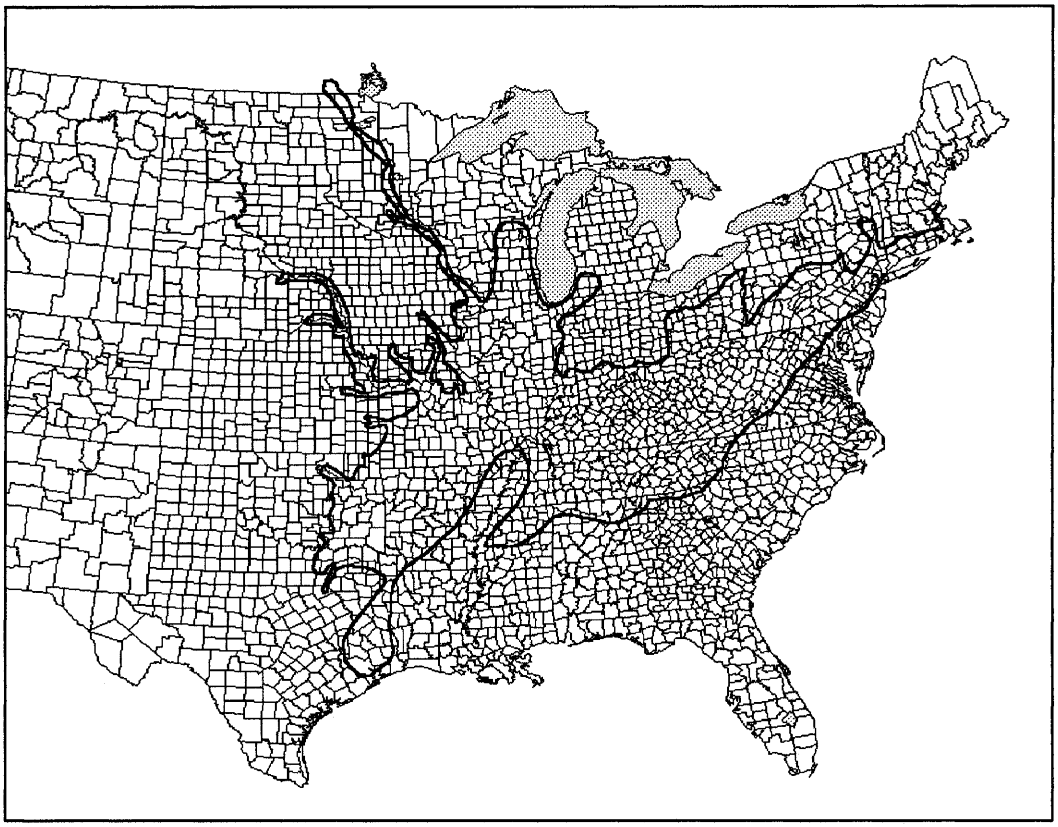

Figure 1.—Central Hardwoods region from Braun (1950).

parklike, which may have favored mast producing species and facilitated foraging by Passenger Pigeons on the ground. Bucher (1992) has hypothesized that destruction of forest breeding habitat and the lack of social facilitation in foraging were the major factors that contributed to the extinction of the species. Bachman's Warbler may have narrowly specialized on canebrakes (*Arundinaria* spp.) in bottomland hardwoods for breeding habitat (Remsen 1986). Fire played a critical role in maintenance of these large stands of cane (Hughes 1966).

Several bird species have been nearly extirpated from the Central Hardwoods landscape. The distribution within the region of Greater Prairie Chicken, Red-cockaded Woodpecker, Bewick's Wren, and Brown-headed Nuthatch have all receded within the last 200 years. The Greater Prairie Chicken formerly occurred in Ohio, Kentucky, Arkansas, and Missouri, and the eastern seaboard from Massachusetts and Virginia, but disappeared from hunting pressure and habitat destruction (Schroeder and Robb 1993). Lumsden (1966) suggested that the Greater Prairie Chicken may have colonized the eastern U.S. in response to grassland openings created when Native Americans burned woodlands.

Causes for the decline of the Bewick's Wren in the Central Hardwoods is far from clear. The most common explanation is that the Bewick's Wren expanded eastward in the early 19th century in response to agricultural expansion, but was then out-competed by the House Wren as it expanded its range (Kennedy and White 1997). Although the Bewick's Wren was not found in the region prior to 1821, C. Hunter (pers. comm.) has suggested that it might have been present in the oak-savanna mosaic of the Central Hardwoods. Loss of this habitat to agriculture and modification of the fire regime may have degraded the best habitat for the Bewick's Wren.

Pine Woodland Specialists in the Central Hardwoods Region

Some of the rarest and most threatened bird species of the Central Hardwoods landscape are associated with pine woodlands along the southern edge of the region. The Red-cockaded Woodpecker, a federally endangered species since 1970, is highly associated with open pinelands in the southeastern United States (Jackson 1994). Historically, the distribution of the woodpecker within the Central Hardwoods was primarily the Ouachita Highlands of Arkansas, Missouri, and Oklahoma, and the Cumberland Plateau of Kentucky and Tennessee (fig. 1). This species was typically associated with stands of shortleaf pine *(Pinus echinata)* on dry slopes and ridges. Interestingly, the distribution of Red-cockaded

Woodpecker clusters in the Daniel Boone National Forest has striking overlap with the distribution of Native American archeological sites (L. Perry, pers. comm.), which suggests that Native American use of fire may have been important in maintaining habitat for the woodpecker.

The Red-cockaded Woodpecker was apparently extirpated from Missouri by 1946 (Eddleman and Clawson 1987) and from Tennessee by 1994 (Nicholson 1997). Only one individual native to Kentucky remains, although 16 individuals have been translocated to Kentucky from larger populations to support the recovery effort there (L. Perry, pers. comm.). In each of these states the loss or decline of the red-cockaded woodpecker seemed to follow a similar pattern: elimination of old-growth shortleaf pine and drastic reduction of the fire frequency put extreme pressure on populations that were naturally small and fragmented by topography.

The largest remaining population of Red-cockaded Woodpeckers in the Central Hardwoods region is found in the 5701-ha McCurtain County Wilderness Area in southeastern Oklahoma, which is dominated by old-growth shortleaf pine-hardwood forest on steep ridges. Historical accounts characterize the vegetation of the Ouachita Mountains of Oklahoma "...as widely spaced trees with low basal area and stem density, open grassy understories, sparse midstories, and indicate that fire shaped forest conditions (e.g., pine-bluestem communities)" (Masters and others 1995). Substantial evidence indicates that Native Americans (the Caddo Indians before 1751 and the Choctaw in the mid-nineteenth century) used fire frequently in the area. Fire was also used frequently by early European settlers as indicated by analysis of fire scars, up until the era of modern fire suppression in 1957 (Masters and others 1995). Mean fire interval of the Wilderness Area during the presettlement period was between 3.5 and 5.6 years; this extended to over 500 years in the era of modern fire suppression.

Fire suppression and attendant extensive growth of hardwoods within the McCurtain County Wilderness Area is believed to have been partially responsible for the decline of the Red-cockaded Woodpecker (Kelly 1991; Masters and others 1995; Masters and others 1995). Wood (1977) located 29 groups (89-92 individuals) from 1974-77; Kelly (1991) found 15 active clusters (31 individuals); and Masters and others (1995) reported 9 active clusters (22 individuals): an average loss of 1.4 clusters per year. The Oklahoma Department of Wildlife Conservation is actively working to restore woodpecker habitat through hardwood control and reintroduction of fire.

Efforts to restore Red-cockaded Woodpecker habitat include removal of hardwoods that have grown in response to pine harvest and fire suppression and reintroduction of fire. In the Central Hardwoods region Red-cockaded Woodpecker habitat restoration has resulted in predictable effects on other species in the avian community. (Wilson and others 1995) reported increases in numbers of 10 pine-grassland bird species (Eastern Wood-Pewee, Northern Bobwhite, Red-headed Woodpecker, Prairie Warbler, Indigo Bunting, Bachman's Sparrow) in response to management for Red-cockaded Woodpeckers in Arkansas. (Hines 1999) found similar response of the avian community in Kentucky. In both studies some "interior" species (Black-and-white Warbler, Ovenbird) were found more frequently in control plots than in treatments.

Avian Communities of Savannas, Barrens, and Glades in the Central Hardwoods Region

Savannas, barrens, and glades were important components of the Central Hardwoods landscape before European settlement and even before human colonization. Pine barrens in New Jersey and New York (Gibson and others 1999), serpentine barrens in Pennsylvania (Tyndall and Hull 1999), rock outcrop communities (Quarterman and others 1993), cedar glades (Baskin and Baskin 1999), and the Big Barrens region of Kentucky and Tennessee (Baskin and others 1999) are some of the vegetation types that provided open, grassland/shrubland/savanna habitats for birds in the Central Hardwoods region. Some of these plant communities were extensive--the Big Barrens was approximately 2 million acres at the time of European settlement–but many were relatively small and isolated (Quarterman and others 1993) historical and prehistorical evidence suggests that grasslands in the region have long provided important habitat for many species of birds (Askins 1999).

Many of these communities share common features that inhibit tree growth: harsh edaphic conditions (i.e., shallow soils over bedrock that dry quickly) and the frequent occurrence of fire (Anderson and others 1999). Use of fire by Native Americans played an essential role in retarding succession and maintaining these communities with few or no shrubs or trees (Bonnicksen and others 1999; Askins 1999).

Populations of birds, as measured by the North American Breeding Bird Survey (BBS), that inhabit grassland, shrubland, and savanna communities are showing some of the most dramatic declines of any functional group of bird species in the eastern United States (Peterjohn and Sauer 1999). To better focus on bird population trends in the Central Hardwoods region, I selected the 10 physiographic regions within the Central Hardwoods region that are used by the BBS for trend estimates (table 1). I counted the number of physiographic regions in which each species showed statistically significant positive and negative trends over the period 1965 to 1995 (DeGraaf and Rappole 1995; Appendix). For example the Field Sparrow has had significant negative trends over a 30-year period in 9 of the 10 BBS physiographic regions; the American Robin has significantly increased in 7 of 10 regions (Appendix).

Relatively few studies have been published of the avian communities of the savannas, barrens, and glades of the Central Hardwoods region. One study of a cedar glade (Schultz 1930; Quarterman et. al 1993) and a Breeding Bird Census in a serpentine grassland in Pennsylvania (Greenewalt and others 1961) indicate that many of the birds

Table 1.—Comparison of physiographic regions used by Braun (1950) and the Breeding Bird Survey (Robbins et al. 1986).

Braun (1950)	Breeding Bird Survey
1. Mixed Mesophytic Forest	21. Cumberland Plateau
a. Cumberland Mtns.	22. Ohio Hills
b. Alleghany Mtns.	
c. Cumberland and Allelghany Plateaus	
2. Western Mesophytic Forest	
a. Bluegrass	14. Highland Rim
b. Nashville basin	15. Lexington Plain
c. Illinoian glaciation	
d. Mississippian Plateau	
e. Mississippi Embayment	
3. Oak-hickory Forest	
Southern Division	19. Ozark-Ouachita Plateau
a. Interior Highlands	31. Till Plains
b. Forest-Prairie Transition	
Northern Division	
c. Mississippi Valley	
d. Prairie Peninsula	
4. Oak-Chestnut Forest	
a. Southern Apalachians	10. Northern Piedmont
b. Northern Blue Ridge	12. Southern New England
c. Ridge and Valley	13. Ridge and Valley
d. Piedmont	23. Blue Ridge Mountains
e. Glaciated	

of these vegetation communities are showing significant regional declines (table 2).

Some of the best remaining examples of grasslands in the Central Hardwoods region have some form of conservation protection. Management objectives of these preserves frequently include maintenance of small, relatively isolated populations of rare plants, invertebrates, and birds. Burning in grassland preserves such as these requires special attention (Engstrom 1997). Consider the following general suggestions for managing grasslands. (1) Mix up management applications (e.g., season of fire) from year-to-year to provide variation in disturbance. No single management prescription will be best for all species. (2) Consider vegetation litter recovery and accumulation when establishing fire frequency. This is important for some bird species, such as Henslow's Sparrow, which prefers relatively thick grasses for nesting. (3) Consider the size of the management area in relation to the needs of the species of interest. Subdividing large tracts into smaller units and applying different management practices on different units would provide habitats in a variety of stages of recovery from disturbance.

Birds of Oak Savannas

Fire is critical for determining the structure and composition of vegetation of sand savannas (Will-Wolf and Stearns 1999), deep soil savannas (Anderson and Bowles 1999), and open oak woodlands (Fralish and others 1999) in the Central Hardwoods region. In the absence of fire, shade-tolerant tree species such as sugar maple (*Acer saccharum*), gradually replace oaks (Abrams 1992) close the canopy, and create more mesic conditions. Changes in tree species composition and forest structure strongly affect the avian community. Recently, (Brawn 1998) and (Artman and Downhower 2000) have experimentally studied the effects of fire on bird communities in oak woodlands in Illinois and Ohio respectively.

As in the cases of restoration of shortleaf pine woodlands the use of fire to manage oak woodlands and savannas favors some bird species over others (table 3). Note that many of the bird species that increase significantly after application are those that are declining within the region.

Conclusions

Many of the structurally open habitats in the Central Hardwoods region probably originated because edaphic

Table 2.—Bird species found in two grassland communities in the Central Hardwoods region. Underlined species are showing significant declines in at least three physiographic regions used by the Breeding Bird Survey (See Appendix).

Cedar glade in central Tennessee (Schultz 1930)	Serpentine Barrens in Pennsylvania (Greenewalt et al. 1961)
Northern Cardinal	Rufous-sided Towhee
Indigo Bunting	Field Sparrow
Field Sparrow	Gray Catbird
Chipping Sparrow	Brown Thrasher
Rufous-sided Towhee	
White-eyed Vireo	
Red-eyed Vireo	
Carolina Chickadee	
Common Yellowthroat	
Carolina Wren	
Blue Jay	
Yellow-breasted Chat	
American Crow	
Tufted Titmouse	
Wood Thrush	
Summer Tanager	
Blue-gray Gnatcatcher	
Northern Mockingbird	
Northern Flicker	
Brown Thrasher	
Gray Catbird	
Northern Bobwhite	
Mourning Dove	
Yellow-billed Cuckoo	
Downy Woodpecker	
Eastern Bluebird	
Orchard Oriole	

Table 3—Bird species that showed significant responses to use of prescribed fire in oak woodlands.

	Brawn (1998)	Artman and Downhower (this conference)
Northern Bobwhite	Positive	
Mourning Dove	Positive	
Red-headed Woodpecker	Positive	
Great Crested Flycatcher	Positive	
Eastern Wood-Pewee		Positive
American Robin		Positive
Wood Thrush	Negative	
Veery	Negative	
Brown Thrasher	Positive	
Ovenbird	Negative	Negative
Hooded Warbler		Negative
Worm-eating Warbler		Negative
Scarlet Tanager	Negative	
Summer Tanager	Positive	
Eastern Towhee	Positive	
Indigo Bunting	Positive	Positive
Baltimore Oriole	Positive	

conditions permitted few trees to develop and then were kept open by extensive use of fire by Native Americans. Bird species that prefer open habitat structure have either largely disappeared (Greater Prairie Chicken, Bachman's Sparrow, Red-cockaded Woodpecker) or shifted to agricultural habitats. Many of the species that were common in grasslands prior to European settlement and subsequently shifted to agricultural habitats are undergoing widespread population declines as farmland has been converted to forests or human environments. Great reduction in the number of grasslands, balds, and barrens has increased the importance of managing the remaining grasslands with care to avoid endangering small isolated populations of invertebrates and some bird species. Prescribed fire is an effective tool and sometimes the only practical tool for achieving management goals. Managers can take practical steps to minimize loss of individuals through the use of prescribed fire. Any change of fire frequency—either increasing or decreasing—over a long time period will favor some plant species over others, which will strongly affect the habitat structure and associated bird species. In the Central Hardwoods region widespread declines of bird species that are associated with open habitats indicates that fire should be used widely to maintain openings.

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Appendix

Bird species of the Central hardwoods landscape. Status: B=breeding, W=winter, and Y=year-round. Species that have shown statistically significant changes from 1965-1995 from Breeding Bird Survey results. (See text for list of BBS physiographic regions.)

			N physiographic regions showing significant trends	
Common Name	Scientific Name	Status	Positive	Negative
Pied-billed Grebe	*Podilymbus podiceps*	Y		
Eared Grebe	*Podiceps nigricollis*	W		
American White Pelican	*Pelecanus erythrorhynchos*	B		
Double-crested Cormorant	*Phalacrocorax auritus*	B		
Anhinga	*Anhinga anhinga*	B		
American Bittern	*Botaurus lentiginosus*	B		
Least Bittern	*Ixobrychus casmerodius*	B		
Great Blue Heron	*Ardea herodias*	B	7	0
Great Egret	*Casmerodius albus*	B		
Snowy Egret	*Egretta thula*	B		
Little Blue Heron	*Egretta caerulea*	B		
Tricolored Heron	*Egretta tricolor*	B		
Cattle Egret	*Bubulcus ibis*	B		
Green Heron	*Butorides striatus*	B	0	2
Black-crowned Night-Heron	*Nycticorax nycticorax*	B		
Yellow-crowned Night-Heron	*Nycticorax violaceus*	B		
Canada Goose	*Branta canadensis*	Y		
Wood Duck	*Aix sponsa*	B	4	0
Green-Winged Teal	*Anas crecca*	W		
American Black Duck	*Anas rubripes*	Y		
Mallard	*Anas platyrhynchos*	Y	2	0
Northern Pintail	*Anas acuta*	Y		
Blue-winged Teal	*Anas discors*	B		
Northern Shoveler	*Anas clypeata*	B		
Gadwall	*Anas strepera*	W		
American Wigeon	*Anas americana*	B		
Canvasback	*Aythya valisineria*	W		
Redhead	*Aythya americana*	W		
Ring-necked Duck	*Aythya collaris*	W		
Lesser Scaup	*Aythya affinis*	W		
Common Goldeneye	*Bucephala clangula*	W		
Bufflehead	*Bucephala albeola*	W		
Hooded merganser	*Lophodytes cucullatus*	B		
Common Merganser	*Mergus merganser*	W		
Ruddy Duck	*Oxyura jamaicensis*	W		
Black Vulture	*Coragyps atratus*	B	1	0
Turkey Vulture	*Cathartes aura*	Y	1	0
Osprey	*Pandion haliaetus*	B		
Mississippi Kite	*Ictinia mississippiensis*	B		
Bald Eagle	*Haliaeetus leucocephalus*	W		
Northern Harrier	*Circus cyaneus*	Y		
Sharp-shinned Hawk	*Accipiter striatus*	Y	1	0
Cooper’s Hawk	*Accipiter cooperii*	Y	1	0
Northern Goshawk	*Accipiter gentilis*	W		
Red-shouldered Hawk	*Buteo lineatus*	Y	3	0
Broad-winged Hawk	*Buteo platypterus*	B	0	1
Red-tailed Hawk	*Buteo jamaicensis*	Y	7	0
Rough-legged Hawk	*Buteo lagopus*	W		
Golden Eagle	*Aquila chrysaetos*	W		

Continued

Appendix continued

Common Name	Scientific Name	Status	N physiographic regions showing significant trends Positive	Negative
American Kestrel	*Falco sparverius*	Y	1	1
Merlin	*Falco columbarius*	W		
Gray Partridge	*Perdix perdix*	Y		
Ring-necked Pheasant	*Phasianus colchicus*	Y		
Ruffed Grouse	*Bonasa umbellus*	Y		
Sharp-tailed Grouse	*Tympanuchus phasianellus*	Y		
Greater Prairie Chicken	*Tympanuchus cupido*	Y		
Wild Turkey	*Meleagris gallopavo*	Y		
Northern Bobwhite	*Colinus virginianus*	Y		
King Rail	*Rallus elegans*	B		
Virginia Rail	*Rallus limicola*	B		
Sora	*Porzana carolina*	B		
Purple Gallinule	*Porphyrula martinica*	B		
Common Moorhen	*Gallinula chloropus*	B		
American Coot	*Fulica americana*	Y		
Piping Plover	*Charadrius melodus*	B		
Killdeer	*Charadrius vociferus*	Y	4	1
Spotted Sandpiper	*Actitis macularia*	B		
Upland Sandpiper	*Bartramia longicauda*	B		
Common Snipe	*Gallinago gallinago*	W		
American Woodcock	*Scolopax minor*	Y		
Wilson's Phalarope	*Phalaropus tricolor*	B		
Ring-billed Gull	*Larus delawarensis*	W		
Herring Gull	*Larus argentatus*	W		
Common Tern	*Sterna hirundo*	B		
Forster's Tern	*Sterna forsteri*	B		
Least Tern	*Sterna antillarum*	B		
lack Tern	*Chlidonias niger*	B		
Black Skimmer	*Rynchops niger*	B		
Rock Dove	*Columba livia*	Y		
Mourning Dove	*Zenaida macroura*	Y	4	0
Black-billed Cuckoo	*Coccyzus erythropthalmus*	B	0	1
Yellow-billed Cuckoo	*Coccyzus americanus*	B	0	6
Common Barn-Owl	*Tyto alba*	Y		
Eastern Screech-Owl	*Otus asio*	Y		
Great Horned Owl	*Bubo virginianus*	Y		
Barred Owl	*Strix varia*	Y		
Long-eared Owl	*Asio otus*	Y		
Short-eared Owl	*Asio flammeus*	Y		
Northern Saw-whet Owl	*Aegolius acadicus*	Y		
Common Nighthawk	*Chordeiles minor*	Y	0	2
Chuck-will's-widow	*Caprimulgus carolinensis*	B		
Whip-poor-will	*Caprimulgus vociferus*	B	0	2
Chimney Swift	*Chaetura pelagica*	B	1	3
Ruby-throated Hummingbird	*Archilochus colubris*	B	0	1
Belted Kingfisher	*Ceryle alcyon*	Y	1	3
Red-headed Woodpecker	*Melanerpes erythrocephalus*	Y		
Red-bellied Woodpecker	*Melanerpes carolinus*	Y		
Yellow-bellied Sapsucker	*Sphyrapicus varius*	W		
Downy Woodpecker	*Picoides pubescens*	Y		
Hairy Woodpecker	*Picoides villosus*	Y		
Red-cockaded Woodpecker	*Picoides borealis*	Y		

Continued

Appendix continued

Common Name	Scientific Name	Status	N physiographic regions showing significant trends Positive	 Negative
Northern Flicker	*Colaptes auratus*	Y		
Pileated Woodpecker	*Dryocopus pileatus*	Y		
Olive-sided Flycatcher	*Contopus borealis*	B		
Eastern Wood-Pewee	*Contopus virens*	B	0	5
Acadian Flycatcher	*Empidonax virescens*	B	0	2
Alder Flycatcher	*Empidonax alnorum*	B		
Willow Flycatcher	*Empidonax traillii*	B		
Least Flycatcher	*Empidonax minimus*	B	0	1
Eastern Phoebe	*Sayornis phoebe*	B	3	0
Great Crested Flycatcher	*Myiarchus crinitus*	B	0	5
Eastern Kingbird	*Tyrannus tyrannus*	B	0	2
Scissor-tailed Flycatcher	*Tyrannus forficatus*	B		
Horned Lark	*Eremophila alpestris*	Y	0	4
Purple Martin	*Progne subis*	B	1	2
Tree Swallow	*Tachycineta bicolor*	B	1	0
Northern Rough-winged	Swallow*Stelgidopteryx serripennis*	B	2	1
Bank Swallow	*Riparia riparia*	B		
Cliff Swallow	*Hirundo pyrrhonota*	B	1	0
Barn Swallow	*Hirundo rustica*	B	1	2
Blue Jay	*Cyanocitta cristata*	Y		
American Crow	*Corvus brachyrhynchos*	Y		
Fish Crow	*Corvus ossifragus*	B		
Common Raven	*Corvus corax*	Y		
Black-capped Chickadee	*Parus atricapillus*	Y		
Carolina Chickadee	*Parus carolinensis*	Y		
Tufted Titmouse	*Parus bicolor*	Y		
Red-breasted Nuthatch	*Sitta canadensis*	W		
White-breasted Nuthatch	*Sitta carolinensis*	Y		
Brown-headed Nuthatch	*Sitta pusilla*	Y		
Brown Creeper	*Certhie americana*	W		
Carolina Wren	*Thryothorus ludovicianus*	Y		
Bewick's Wren	*Thryomanes bewickii*	B	0	1
House Wren	*Troglodytes aedon*	B	4	2
Winter Wren	*Troglodytes troglodytes*	W		
Marsh Wren	*Cistothorus palustris*	B		
Golden-crowned Kinglet	*Regulus satrapa*	W		
Ruby-crowned Kinglet	*Regulus calendula*	W		
Blue-gray Gnatcatcher	*Polioptila caerulea*	B	1	0
Eastern Bluebird	*Sialia sialis*	Y	4	1
Veery	*Catharus fuscescens*	B		
Swainson's Thrush	*Catharus ustulatus*	B		
Hermit Thrush	*Catharus guttatus*	W		
Wood Thrush	*Hylocichla mustelina*	B	1	4
American Robin	*Turdus migratorius*	Y	7	0
Gray Catbird	*Dumetella carolinensis*	B	2	4
Northern Mockingbird	*Mimus polyglottos*	Y		
Brown Thrasher	*Toxostoma rufum*	B		
Cedar Waxwing	*Bombycilla cedrorum*	Y	6	0
Northern Shrike	*Lanius excubitor*	W		
Loggerhead Shrike	*Lanius ludovicianus*	Y	0	3
European Starling	*Sturnus vulgaris*	Y		
White-eyed Vireo	*Vireo griseus*	B	1	3

Continued

Appendix continued

			N physiographic regions showing significant trends	
Common Name	Scientific Name	Status	Positive	Negative
Solitary Vireo	*Vireo solitarius*	B		
Bell's Vireo	*Vireo bellii*	B	0	1
Yellow-throated Vireo	*Vireo flavifrons*	B	0	1
Warbling Vireo	*Vireo gilvus*	B	1	1
Red-eyed Vireo	*Vireo olivaceus*	B	1	1
Blue-winged Warbler	*Vermivora pinus*	B	0	2
Golden-winged Warbler	*Vermivora chrysoptera*	B	0	1
Nashville Warbler	*Vermivora ruficapilla*	B		
Northern Parula	*Parula americana*	B	1	0
Yellow Warbler	*Dendroica petechia*	B	2	0
Chestnut-sided Warbler	*Dendroica pensylvanica*	B		
Magnolia Warbler	*Dendroica magnolia*	B		
Black-throated Blue Warbler	*Dendroica caerulescens*	B		
Yellow-rumped Warbler	*Dendroica coronata*	W	1	0
Black-throated Green Warbler	*Dendroica virens*	B	1	0
Blackburnian Warbler	*Dendroica fusca*	B		
Yellow-throated Warbler	*Dendroica dominica*	B	2	1
Pine Warbler	*Dendroica pinus*	B	2	0
Prairie Warbler	*Dendroica discolor*	B	0	6
Cerulean Warbler	*Dendroica cerulea*	B	0	3
Black-and-white Warbler	*Mniotilta varia*	B	0	4
American Redstart	*Setophaga ruticilla*	B	0	4
Prothonotary Warbler	*Protonotaria citrea*	B	0	1
Worm-eating Warbler	*Helmitheros vermivorus*	B		
Swainson's Warbler	*Limnothlypis swainsonii*	B		
Ovenbird	*Seiurus aurocapillus*	B		
Louisiana Waterthrush	*Seiurus motacilla*	B		
Common Yellowthroat	*Geothlypis trichas*	B	0	2
Hooded Warbler	*Wilsonia citrina*	B		
Kentucky Warbler	*Oporornis formosus*	B	0	1
Canada Warbler	*Wilsonia canadensis*	B		
Yellow-breasted Chat	*Icteria virens*	B	0	7
Summer Tanager	*Piranga rubra*	B	0	2
Scarlet Tanager	*Piranga olivacea*	B	3	1
Northern Cardinal	*Cardinalis cardinalis*	Y		
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	B	2	1
Blue Grosbeak	*Guiraca caerulea*	B		
Indigo Bunting	*Passerina cyanea*	B	0	7
Painted Bunting	*Passerina ciris*	B		
Dickcissel	*Spiza americana*	B	0	2
Rufous-sided Towhee	*Pipilo erythrophthalmus*	Y	0	9
Bachman's Sparrow	*Aimophila aestivalis*	B		
American Tree Sparrow	*Spizella arborea*	W		
Chipping Sparrow	*Spizella passerina*	B	2	2
Field Sparrow	*Spizella pusilla*	Y	0	10
Vesper Sparrow	*Pooecetes gramineus*	B	0	3
Lark Sparrow	*Chondestes grammacus*	B	0	1
Savannah Sparrow	*Passerculus sandwichensis*	B		
Grasshopper Sparrow	*Ammodramus savannarum*	B	0	6
Henslow's Sparrow	*Ammodramus henslowii*	B		
Le Conte's Sparrow	*Ammodramus leconteii*	W		
Fox Sparrow	*Passerella iliaca*	W		

Continued

Appendix continued

Common Name	Scientific Name	Status	N physiographic regions showing significant trends Positive	 Negative
Song Sparrow	*Melospiza melodia*	Y		
Lincoln's Sparrow	*Melospiza lincolnii*	W	1	0
Swamp Sparrow	*Melospiza georgiana*	Y		
White-throated Sparrow	*Zonotrichia albicollis*	W		
White-crowned Sparrow	*Zonotrichia leucophrys*	W		
Dark-eyed Junco	*Junco hyemalis*	W		
Lapland Longspur	*Calcarius lapponicus*	W		
Snow Bunting	*Plectrophenax nivalis*	W		
Bobolink	*Dolichonyx oryzivorus*	B	1	1
Red-winged Blackbird	*Agelaius phoeniceus*	Y	0	6
Eastern Meadowlark	*Sturnella magna*	Y	0	9
Western Meadowlark	*Sturnella neglecta*	B		
Yellow-headed Blackbird	*Xanthocephalus xanthocephalus*	B		
Rusty Blackbird	*Euphagus carolinus*	W		
Brewer's Blackbird	*Euphagus cyanocephalus*	B		
Common Grackle	*Quiscalus quiscula*	Y		
Brown-headed Cowbird	*Molothrus ater*	Y	0	3
Orchard Oriole	*Icterus spurius*	B	1	3
Northern Oriole	*Icterus galbula*	B	1	3
Purple Finch	*Carpodacus purpureus*	W		
House Finch	*Carpodacus mexicanus*	Y		
Pine Siskin	*Carduelis pinus*	W		
American Goldfinch	*Carduelis tristis*	Y	0	1
Evening Grosbeak	*Coccothraustes vespertinus*	W		
House Sparrow	*Passer domesticus*	Y		
Eurasian Tree Sparrow	*Passer montanus*	Y		

Stream Mayflies, Stoneflies, and Caddisflies: Do They Care About Controlled Burning in the Forest?

Jack W. Feminella[1]

Despite the widespread use of fire as a management tool in forested landscapes, we know surprisingly little about its effects on streams and their biota. Forest management of this type may affect light, temperature, nutrient, sediment, and hydrologic regimes within stream channels, which in turn may affect the abundance and diversity of aquatic microbes, plants, and animals. In cooperation with the USDA Forest Service / National Forests in Alabama, in 1994 my graduate students and I began a long-term experimental study of the effects of forest thinning and prescribed burning on stream water quality within the Talladega National Forest (Shoal Creek District), Alabama. Streams in six small watersheds have been monitored seasonally; three of the streams are being exposed to understory thinning and infrequent, low-intensity burning (i.e., once every 2 years), and the three remaining streams drain watersheds that have been left intact as reference sites. Study watersheds lie in close proximity to each other and have similar drainage areas, geomorphology, parent geology, and soils. Streams are small (1st-order channels, < 2 m wide), heavily shaded (> 85% riparian cover), and are well oxygenated (> 7 ppm). Particular emphasis has been placed on the response of benthic invertebrates (mostly aquatic insects) to management, although we also have studied physical-chemical water quality attributes and stream periphyton (attached algae).

Our analyses to date revealed no major differences in water chemistry between managed and reference streams, with both groups of streams showing uniform pH (slightly acidic), and low alkalinity, conductivity, total suspended solids, and dissolved nutrients. Periphyton abundance was higher in spring and summer (vs. winter), and also was generally higher in managed streams. All streams showed high benthic macroinvertebrate species richness (> 125 species per stream). Preliminary analysis from spring and summer 1995 samples suggests that habitat-specific richness was higher in reference than in managed streams, especially in spring. However, the two groups of streams showed similar total invertebrate total density, species diversity, EPT richness (i.e., number of mayfly, stonefly, & caddisfly species), and species dominance. Although these data are preliminary, they are compelling in that they suggest thinning and burning and associated environmental changes (e.g., increases in light levels reaching streams, short-term but biologically significant nutrient releases, etc.) may have positive effects on the abundance and species composition of stream periphyton, but only minor effects on stream invertebrate diversity.

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[1] Associate Professor, Department of Biological Sciences, Auburn University, 331 Funchess Hall, Auburn, AL 36849-5407, email: feminjw@mail.auburn.edu

Fire Bugs

Jim Bess[1]

In 1990, I initiated insect surveys for the Kentucky Field Office of The Nature Conservancy (KYFO) on several of the highest quality oak/pine barrens and sandstone/limestone glades left in the state. These plant communities are known to contain many state and globally rare plant species found only on these remnant natural areas. Therefore, it was thought that they might also contain remnant-requiring insects. Given that these plant communities are dominated by grasses and fire-dependent plant species, fire was an important part of their history and is used widely in their current management and restoration. The KYFO was primarily interested in identifying those species restricted in occurrence to barrens and glade remnants which might also be sensitive to the short-term effects of prescribed burning. Our studies have focused on the three major herbivorous, grassland insect orders: the Homoptera (cicadas, leafhoppers, aphids and their relatives), the Lepidoptera (butterflies and moths) and Orthoptera (grasshoppers, katydids, crickets, mantids and stick insects). The results of my survey work have identified a large number of potentially remnant-requiring insects (roughly 25 percent of the total), several of which are new to the state. On going survey work is continuing to identify new species and we are focusing our attention on an even broader assortment of insect taxa.

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[1] 13501 South 750, West Wanatah, IN 46390, ph.: 219-733-2947, email: jabess@mail.netnitco.net

Black Bear Ecology and the Use of Prescribed Fire to Enhance Bear Habitat

Keith M. Weaver[1]

Abstract

The black bear (*Ursus americanus*) is a species of interest for wildlife and forest managers, landowners, hunters, and the public. Because of their overall adaptability to changing habitat conditions, wide-ranging nature, and omnivorous habits, black bears are readily accommodated in a forest management program. A well-managed forest that provides adequate opportunities for movements, foraging, denning, and cover has better potential to satisfy the needs of bears than an unmanaged forest. Food supplies should be abundant, stable, and diverse. Habitats that harbor natural dens should be conserved, known den sites should be protected, and creation of potential den habitat and den sites should be considered in forest habitat treatments. In addition, natural escape cover should be maintained, and opportunities should be undertaken to create or enhance cover for bedding, escape, and dispersal. Although bear habitat management has traditionally been associated with timber management, the use of prescribed fire holds promise as an effective tool for enhancement of black bear habitat in the central hardwood landscape. In this paper, various aspects of black bear habitat ecology are reviewed, and the use of prescribed fire alone or in conjunction with other silvicultural techniques to maintain, enhance, and restore habitat conditions for black bears is explored. Evaluation of prescribed fire behavior and activity, habitat responses to fire under various conditions, and associated implications for bears could be useful in developing "best fire management practices" in bear habitat. Prescribed fire should be used in an adaptive management approach. This entails not only the application of fire, but also the diligent monitoring of habitat and wildlife responses, and fine-tuning future actions. Topics of investigation concerning bears and fire are listed. Public education programs conveying the ecological effects and benefits of prescribed fire for habitat, wildlife, and humans, and the development of public/private partnerships for habitat management are recommended.

Background

The black bear is a species of management interest in most areas of its range. Humans hold widely divergent viewpoints regarding bears. Black bears are characterized as big game animals, links to cultural heritage, food sources, dangerous animals, tourist attractions, photographic subjects, spiritual beings, indicators of healthy ecosystems, symbols of wilderness, pawns for environmental activism, agricultural pests, and as a threat to resource extraction and economic development programs. Because of the importance of bears in the lives of many humans, and the need to satisfy legislative, regulatory, or other obligations to properly manage wildlife and their habitats, wildlife and land managers seek to enact management practices beneficial to bears and humans. Responsible bear management includes the recognition that carrying capacity has biological and socioeconomic aspects. Both must be considered for their relevance to providing adequate resources for bears, reducing human/bear conflicts, and fostering public acceptance of bears. Bear damage complaints in agriculture and garbage usually increase when shortages of natural foods occur, which provides additional justification for establishing and maintaining suitable forest conditions for bears. This is particularly necessary in fragmented habitat where bears are surrounded by more agriculture and development than forest. Bear habitat management should focus on providing suitable habitat and habitat linkages, abundant natural food supplies, denning sites, escape cover, and human access management. The purpose of this paper is to highlight some aspects of black bear habitat ecology and to explore the use of prescribed fire to enhance bear habitat in the central hardwood landscape. The concept of prescribed fire as used in this paper is the application of human ignited and controlled fire to a site, according to approved techniques, within specified limits of the physical conditions of air, soil, and vegetation, in order to effect a desired change in site conditions, for the purpose of achieving previously defined goals and objectives.

Habitat

Black bears are adaptable, opportunistic, and highly mobile animals. Bears can move great distances in response to changing weather, food supplies, habitat conditions, disturbance, and internal population pressures. Movements and home range size vary greatly depending on age, sex, reproductive status, population density, and habitat quality. Males typically range farther than females. Sizes of bear home ranges in the Cherokee National Forest in east Tennessee, as determined through analysis of radiotelemetry data, varied from 20 to 192 square kilometers for males and 4 to 23 square kilometers for females (Villarubia 1982, Garris 1983, Clevenger 1986). Such mobility simplifies habitat management for bears in some respects, because management actions can be directed beyond the microsite to the forest or landscape scale. Conservation or creation of habitat linkages between disjunct tracts of forest can help provide corridors for bear movements in search of food, dens, and mates; juvenile dispersal; and also facilitate bear population expansion.

Although black bears are creatures of the forest, they do not require wilderness or old growth habitats. However, both of these habitat characteristics can be useful components of the overall bear landscape. Black bears can survive and coexist with human inhabitation nearby if the bears are afforded a forest that satisfies their life requisites. The use of

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[1] Refuge Manager, U.S. Fish and Wildlife Service, Nulhegan Basin Division, Silvio O. Conte NFWR, P.O. Box 427, Island Pond, VT 05846.

prescribed fire can aid in establishment of desired regeneration. It also can be used for site preparation in reforestation programs that will expand forested habitat and provide an additional buffer from human encroachment into bear habitat. The greater the quantity, but perhaps more important, the quality of bear habitat, the less chance for conflicts between humans and bears.

The key to improving the quality of bear habitat is to provide habitat diversity. Within the forest, a mosaic of habitats or habitat conditions (e.g., species composition, age class, structure, successional stage, openings, understory) holds the best potential for the satisfaction of bear life requirements. The value of managing a mixed pine-oak-hickory component within the central hardwood landscape should not be overlooked for its benefit for bears. A diverse forest may be better adapted to withstand the ravages of disease, insect invasions, or catastrophic events. Therefore, a properly managed forest likely will provide better bear habitat potential than an unmanaged forest. Indeed, the outcome of modeling timber harvest effects on black bear population dynamics in the southern Appalachian Mountains indicated that forests with carefully designed timber management regimes would yield greater food abundance and stability than unmanaged forests (Brody and Stone 1987). It is important to recognize that a decision not to perform habitat manipulation has its own consequences that also must be evaluated for their impact on bears.

The credibility and compatibility of forest management in providing for bear habitat needs was demonstrated by the U.S. Fish and Wildlife Service through promulgation of a special rule in the Threatened listing for the Louisiana black bear (*U. a. luteolus*), which states, in part, that the provisions of the Federal listing "... shall not prohibit effects incidental to normal forest management activities within the historic range of the Louisiana black bear except for activities causing damage to or loss of den trees, den tree sites or candidate trees" (Federal Register 7 January 1992, 57(4) sub-sect. 17.40, i(2), pp. 594-595). Normal forest management activities were defined in the rule as "... activities that support a sustained yield of timber products and wildlife habitats, thereby maintaining forestland conditions in occupied habitat." Although the range of the Louisiana bear is distant from the central hardwood landscape, the premise is identical; responsible forest management is good bear habitat management.

Traditionally, forest management for black bears has been a function of timber management. Treatments such as clearcut, patch clearcut, shelterwood and irregular shelterwood, and thinning are useful methods for creating, improving, and maintaining desirable bear habitat. However, the ability to manage forests with conventional silvicultural techniques has been hindered because of increasing public resentment to timber harvest, increased environmental regulation, and decreasing budgets. Therefore, the use of prescribed fire may provide a complementary habitat management technique for bears. Prescribed fire also may be a useful tool to enhance bear habitat in areas where timber harvest is impractical or prohibited. Ideally, the integration of prescribed fire with other silvicultural treatments affords the greatest potential for maintenance and establishment of suitable bear habitat conditions. A comprehensive prescribed fire program also should include hazard fuel reduction burns that serve to reduce the occurrence or severity of wildfires that could destroy bear habitat and negate beneficial habitat management programs.

The practice of managing forests, landscapes, or wildlife populations implies making choices that entail potential risks and benefits, and advantages and disadvantages for target and nontarget sites, species, and humans. Therefore, it is imperative that clear management goals and objectives are defined and approved, appropriate methods employed, outcomes evaluated for success or failure, and modifications for future achievement enacted. Prescribed fire use must not be relegated to the "meet the target" syndrome. Additionally, a forest management program should not be expected to provide for every wildlife or human use, on every acre, in every management compartment, or in every decade. Some sites are better suited to a certain management purpose than others, and there are practical limitations to the degree of manipulation that can be achieved or the extent of attention that can be devoted on a site during any given period. Tradeoffs will occur and management choices must be made that will accommodate the greatest suite of needs and interests within applicable laws, policy, funding, technology, and practicality. These choices should be based on the best available technical information; however, when developing new practices, guiding methodologies are often limited, and sometimes the intuition and experience of managers must be relied on to determine the prudent course of action.

Foods

Nutrition is the foundation for the survival and welfare of individual bears and the dynamics of bear populations (e.g., Rogers 1976, 1987, 1993). Therefore, it is critical that forest management practices for black bears ensure a variety of stable and abundant food supplies throughout the year. This can be accomplished by providing habitat diversity; a mosaic of habitat types, stand ages, and conditions; high yields of soft and hard mast; herbaceous foods; and decomposing wood that harbors insect and other invertebrate life. Forested habitat, forest openings, edge habitats, riparian borders, and crop fields within and adjacent to forested habitat provide feeding opportunities for bears. Black bears are excellent tree climbers and are able to forage in all vertical strata of the forest; therefore, the impacts of habitat management actions on food production in the overstory, midstory, and understory should be considered. Black bears are wide-ranging, omnivorous, and opportunistic in their feeding habits. Thus they are able to capitalize on a wide range of conditions in their search for food. Food "preferences" (Bacon and Burghardt 1983) are difficult to determine, but bear foraging behavior and food consumption are influenced by: season, food abundance and accessibility, proximity of other foraging bears, bear densities, human activity in bear habitat, physiological condition (e.g., pre and post denning), and reproductive status.

Hard Mast

The use of the term hard mast in this paper refers collectively to the fruit of oaks (*Quercus* spp.), beechnuts (*Fagus grandifolia*), and hickories and pecans (*Carya* spp.). In the fall and early winter prior to denning, bears in the eastern deciduous forest feed heavily on hard mast, particularly acorns. They largely depend on these foods to build sufficient fat reserves to survive the period of winter dormancy (Beeman and Pelton 1980, Eagle and Pelton 1983, Clark and others 1987, Rogers and Allen 1987). Bears forage for hard mast in tree crowns prior to nut drop, and on the ground post-drop. The abundance and availability of hard mast affect black bears at the individual and population levels. The availability of hard mast, particularly where there are few other foods, has been documented to affect population growth (mortality, natality, interbirth interval), bear numbers, fall feeding movements, and denning chronology (Pelton 1985, 1989; McLaughlin and others 1994; McLaughlin 1998; Schooley and others 1994). Such effects of hard mast on black bear survival and population dynamics in the Southern Appalachians have been modeled (Brody and Stone 1987, McLean and Pelton 1994).

Hard mast production should be an important consideration in bear habitat management in the central hardwood landscape. Where feasible, hard mast producing species, particularly oaks, should be encouraged as a major bear forest habitat component in silvicultural and prescribed fire treatments. Long timber rotations (75 to 100 years) will favor hard mast production, and are compatible with peak acorn yield potential (Pelton 1985, Hillman and Yow 1986, Brody and Stone 1987). Treatments should be directed toward maintaining diversity of hard mast species, high mast yields, and perpetuating desired species in future stands. A notable problem in upland hardwood management has been the difficulty in regenerating oak stands on high quality sites following harvest. Thus, many oak-dominated stands have been converted to stands of less desirable species in terms of hard mast such as yellow poplar (*Liriodendron tulipifera*), red maple (*Acer rubrum*), and sweetgum (*Liquidambar styraciflua*) (O'Hara 1986, Kays and others 1988). If this conversion occurs on a wide scale it could adversely affect bear populations. However, upland oaks exhibit morhpological and physiological characters that suggest greater fire adaptation than some of their competing species including: thick bark, vigorous sprouting ability, deep roots, rot resistance following scarring, xeromorphic leaves, and greater hardiness to drought and nutrient-poor sites (Abrams 1992). Historically, oak domination on central hardwood sites has been in response to a fire regime (Abrams 1992). Advance regeneration in the understory is a prerequisite for reestablishment of an oak-dominated stand subsequent to harvest. Van Lear and Waldrop (1989) suggested that repeated understory burns in summer and winter during an extended period (e.g., 5 to 20 years) prior to harvest could be successful in establishing adequate oak advance regeneration. Recently, compelling evidence has been presented that the proper timing, intensity, and periodicity of prescribed fire applied 3 to 5 years following shelterwood cutting in stands with advance oak regeneration will favor oak reproduction and establishment on productive upland sites. The concept, development, and applications of this technique as well as instructions for its implementation are presented by Brose and Van Lear (1998, 1999) and Brose and others (1999a,b). The benefits and drawbacks of prescribed burning in oak-dominated shelterwood stands in different seasons also are discussed by Brose and Van Lear (1998, 1999). The effects of prescribed burning on hard mast production were discussed by Harlow and Van Lear (1989). They commented that hot surface fires in the fall could readily consume acorns in the duff layer, but that low intensity surface fires could serve to expose recently fallen acorns without substantial loss of mast crops. Consumption of heavy leaf litter by fire could aid in the successful location and caching of acorns by blue jays (*Cyanocitta cristata*) and squirrels (*Sciurus* spp.), which could promote oak regeneration (Galford and others 1988). They also indicated winter burning might reduce acorn predation by weevils, thus increasing their availability for wildlife.

Squawroot (*Conopholis americana*), a saprophyte associated with oaks, is an important component of spring diets of bears following den emergence. Squawroot provides nutrition for bears at a period when soft mast production is not yet available. The perpetuation of an oak component in the forest would hopefully enable continued availability of squawroot. The ecology of oaks, squawroot, and fire warrants investigation.

Soft mast

Although hard mast crops are important for black bears, they only are seasonally present, and abundant yields are sporadic. Therefore, other foods also must be available throughout the year. In addition, during hard mast failures, alternate food sources are of great importance. Soft mast produced by woody and herbaceous species is a vital food source that also should be managed as a viable component of bear habitat. Use of the term soft mast in this paper refers to fruits with fleshy exteriors such as berries, drupes, and pomes (Harlow and Van Lear 1989). In the central hardwood landscape, bears consume the fruits of a wide variety of plants such as: blackberries (*Rubus* spp.), blueberries (*Vaccinium* spp.), huckleberries (*Gaylussacia* spp.), serviceberry (*Amelanchier* spp.), pokeberry (*Phytolacca americana*), elderberry (*Sambucus canadensis*), devil's walking stick (*Aralia spinosa*), flowering dogwood (*Cornus florida*), paw-paw (*Asimina triloba*), sassafras (*Sassafras albidum*), *Ilex* spp., hawthorn (*Crataegus* spp.), black cherry (*Prunus serotina*), black gum (*Nyssa sylvatica*), and persimmon (*Diospyros virginiana*). Growth of most of these plants are stimulated by increased light penetration to the forest floor from canopy openings that result from timber harvest, construction of wildlife openings, creation of interior forest trails, and wind or ice damage. Silvicultural and prescribed fire treatments that promote establishment, diversification, proliferation, and perpetuation of soft mast as a forest component in conjunction with hard mast are highly encouraged. Soft mast production peaked 3 to 5 years after clearcuts, shelterwood cuts, and group selection treatments in mixed pine-hardwood stands in the Oauchita Mountains of

Arkansas and Oklahoma (Perry and others 1988). However, it was noted that without additional treatment, such as burning or thinning, soft mast production was expected to decline significantly as canopies in the harvested areas closed. Fruit producing vines such as grape (*Vitis* spp.) and greenbrier (*Smilax* spp.) are also readily eaten by bears; their growth should be encouraged. Heavy canopies of vines (e.g., grape slicks) generally are not compatible with oak regeneration, but if these growths are managed for fruit production (versus mainly vegetative growth), they add a valuable component of diversity for bear foods, and can be important to bears during shortages of hard mast.

The effects of prescribed fire on soft mast production in hardwood stands has received little attention in the literature. A literature review of the effects of prescribed fire on soft mast production in various pine forest types in the Southeast was presented by Harlow and Van Lear (1989). Many soft mast producing species considered in this paper decreased fruit production in the year following burning; however, fruiting in flowering dogwood was increased in the next year after burning. Studies reviewed indicated that fruiting increased 2 to 5 years following burning in species that occur in the central hardwoods such as blueberries and huckleberries, blackberries, persimmon, chokeberry (*Sorbus arbutifolius*), muscadine grape (*V. rotundifolia*), *Viburnum* spp., yaupon (*I. vomitoria*) and plum (*Prunus umbellata*). Harlow and Van Lear (1989) suggested staggered burning cycles of 3 to 5 years. Juxtaposition of burned and unburned areas to promote stable and abundant soft mast crops also has been recommended (Landers 1987, Harlow and Van Lear 1989). Burning rotations of 3 to 10 years were recommended for soft mast production for bears in pine-hardwood stands in coastal North Carolina (Hamilton 1981) and pine dominated habitat in the panhandle of Florida by (Stratman 1998).

Herbaceous Foods

Bears forage in the forest understory, forest openings, and edge habitats for herbaceous plants. The succulent sprouts of various plants stimulated by prescribed fire can provide feeding opportunities in the understory. Pack and others (1988) documented a 31 percent increase in herbaceous forbs, grasses, and sedges following late winter-early spring burns in oak-hickory stands in West Virginia that had previously been thinned. Treatments involving thinning alone or thinning following burning did not produce similar results. To prevent risk of fire injury to the residual stand, they recommended burning only after logging slash had decomposed to the point where it would not constitute a major fuel component. Masters and others (1993), studying timber harvest and fire effects in pine-oak-hickory stands in the Ouachita Mountains of Oklahoma, reported that fire could reduce accumulations of litter and standing dead herbaceous vegetation that impeded new herbaceous growth. The combination of timber harvest and burning increased light penetration to the forest floor, resulted in increased soil temperatures and nitrogen availability, which, in turn, stimulated earlier and greater herbaceous growth and production.

Establishment of a system of widely dispersed forest openings, perhaps <1 ha in size, maintained in various age and species compositions, would provide additional diversification of bear foraging opportunities. Abandoned log loading sites and old fields can be converted to wildlife openings and managed to produce herbaceous vegetation. Old trails or roads not used for transportation routes within the forest can be converted and maintained as linear wildlife openings (Hillman and Yow 1986). Openings could even be created in marginal forest stands using a hot fire that would kill woody vegetation. Annual burning likely will be necessary to stimulate high seed production and maintain herbaceous conditions. Disking or mowing on a 1 to 3 year rotation can be used to maintain openings in early successional stages. However, prescribed fire could provide an alternate method for rejuvenation of herbaceous growth and to setback woody succession. Use of prescribed fire can eliminate the need of accessing remote openings with machinery, and reduce costs of maintaining and operating tractors and implements. Openings can be enhanced with periodic sowing of food plants such as clovers (*Trifolium* spp.), wheat, rye, chufa (*Cyperus esculentus*), brown-top millet (*Panicum ramosum*), or bahia grass (*Paspalum notatum*). Prescribed fire can aid site preparation for initial or subsequent seedings. Edge plants such as blackberries, pokeweed, elderberry, and devil's walking stick that proliferate in and around forest openings further increase habitat diversity and feeding opportunities. These openings also provide habitat for a host of other wildlife species. Prescribed fire management is also indicated for vegetation management in rights-of-way, and for the perpetuation and management of grassy balds in the Southern Appalachians. Resultant vegetation and soft mast within and along the edges of these treatment areas serve to increase bear feeding opportunities.

Invertebrates

Invertebrates such as beetles, ants, termites, and their eggs and larva are protein-rich foods for bears. Such food items often are found inside or underneath decaying trees and logs. Decayed logs that are rolled over or torn apart in search of insects are a common sight in most areas of bear activity. These logs typically are remnants of logging slash or fallen dead trees. Black bears consume insects during most of their non-denning period. Bears also feed on wild honey bees and their honey that are found in standing hollow dead trees, and in cavities in living trees. Retention of standing and fallen snags, and logging slash is encouraged to provide invertebrate food sources. When planning a prescribed fire program with bear habitat enhancement in mind (versus strictly hazard fuel reduction, for example), such food shelters for bears should be favored. Some areas typically targeted for slash reduction burns should be left unburned, or at a minimum, some slash should be protected from fire near the forest edge. Slash left solely for bear foraging opportunities should not be piled. When using prescribed fire in the understory, some fallen logs and standing snags should be protected from fire when and where feasible.

Denning Sites

In the central hardwood landscape, bears may occupy a variety of winter den types including elevated tree cavities, cavities in the base of a hollow tree or under boulders; excavated cavities, hollow logs, brushpiles, or ground nests. Typical bear den trees are: living (although dead trees also are used), large (dbh >90 cm), and old (>150 years). Entrances to elevated cavities may range from 5 to 30 meters above ground. Entrances to den cavities are either through a broken top or a side hole caused by a limb break. Cavities in the base of hollow trees are often initially caused by bark injury (including fire injury) that allowed entry of fungi or other agents that eventually resulted in cavity creation. Potential den tree species in the central hardwoods would include: northern red oak (*Q. Rubra*), chestnut oak (*Q. Prinus*), white oak (*Q. Alba*), yellow poplar, red maple, yellow birch (*B. alleghaniensis*), white pine (*Pinus strobus*), and eastern hemlock (*Tsuga canadensis*). Excavations usually occur on side hills under a blowdown, root cavity, hollow tree base, log, or stump. Brushpile dens typically are found in logging slash or sites of extensive wind throw. Ground nests are usually located in dense thickets of woody vegetation such as rhododendron (*Rhododendron maximum*), mountain laurel (*Kalmia latifolia*), or in regenerating clearcuts.

Recommendations for timber management for black bears in the southern Appalachian Mountains suggest that potential den trees should be reserved from logging, and that a minimum of 5 to10 percent of the forest be maintained in age classes of 250 years or older to ensure adequate supplies of den trees (Pelton 1985, Hillman and Yow 1986). Retention of individual large trees that contain elevated cavities, and large trees with heart rot or wind, lightning, or fire injuries that could develop cavities for dens, is also encouraged during harvest operations. Preferably, such trees should be left in small reserve clumps to help prevent wind throw, particularly in clearcuts or shelterwood cuts. Ideally, pre-burn scouting should be performed to identify potential bear den trees, and these should be protected from fire by clearing woody debris and other fuels from around the base (particularly in the case of standing dead snags). Similarly, downed hollow logs large enough to accommodate a denning bear (approximately 50 cm in diameter) should be protected from fire consumption. When silvicultural treatments are to be followed by prescribed burns, the logging operators should be instructed to clear tops and other slash away from reserved trees to protect them from flareup during the burn.

Ground denning (excavations, brushpiles, ground nests) opportunities are increased following most silvicultural treatments. Practical methods to provide ground denning sites include promoting thick regeneration and scattered piling of felled tops to create potential brushpile den sites. Like old growth habitat, early successional habitat also should be managed as a viable bear habitat component. In forests where few opportunities exist for tree denning (either due to age class structure, tree species composition, or past management practices), regeneration resulting from timber harvest could be crucial habitat for denning bears. Under such circumstances, a forest managed with consideration for tree and ground denning sites for bears will provide more varied denning potential than an unmanaged forest. However, such logic notwithstanding, the conservation of dense rhododendron and laurel thickets, which is contrary to promoting timber regeneration, also will provide ground denning habitat for bears. An integrated approach of providing early successional habitat (regeneration) and maintaining dense shrub thickets would diversify and expand availability of ground denning habitat.

Although no studies in the central hardwood region have dealt with the issue of prescribed fires and denning bears, a study in coastal North Carolina (Lombardo 1993) and two studies in Florida (Land 1994, Stratman 1998) documented abandonment of ground nests by bears, including a female with cubs, during prescribed burns. The bears in the Lombardo study subsequently redenned; such information was not reported in the Florida studies. Conversely, on the Appalachicola National Forest in Florida, denning bears were not disturbed from ground nests during prescribed burn operations (Seibert 1993). These reports serve to remind the manager that prescribed burns must be undertaken with a clear recognition and understanding of the potential risks and benefits to the targeted habitat or wildlife component, both on the site-specific versus forest scale, and on the individual animal versus population scale. Burn units must be properly laid out and of manageable size to allow for responsible protection of special resources and to minimize potential adverse effects to the physical environment, habitat, wildlife, or humans.

Escape Cover

Large expanses of contiguous forest or mountainous, inaccessible forested terrain serve to insulate bears from human disturbance. However, as forests become smaller and more fragmented, and as human encroachment and disturbance in bear habitat increases, escape cover will become even more vital to insulate bears from human activities. Escape cover is recognized as an important component of bear habitat (Pelton 1985, Rogers and Allen 1987, McLaughlin and others 1988). Black bears can thrive in close proximity to humans if afforded areas of retreat that ensure little chance of close contact or visual encounters with humans (Pelton 1982). Areas of dense or impenetrable vegetation that limits visibility and hinders travel by human and dogs, such as rhododendron thickets and regeneration following clearcuts and shelterwood cuts, can provide high quality shelter for daybeds and escape cover for bears. Because dense stands of rhododendron and mountain laurel are not conducive to oak regeneration, this is another situation of tradeoffs, one that pits the need for regeneration versus escape cover.

The utility of large trees as bear escape cover should not be overlooked. Black bears evolved behaviorally and morphologically as tree climbers, and are well-adapted for arboreal escape (Herrero 1972). The largest tree in the area is often chosen as a resting or bedding spot by females with cubs. The female can tree the cubs when danger is perceived, and may accompany them up the tree, protect the base of the tree, or flee the area to distract intruders. Bears also will use large trees for resting. Pockets of old growth forest can ensure the availability of large trees for escape and resting, but ideally, scattered large trees should be available throughout the forest to provide the greatest opportunity for tree escapes. Just as den trees and candidate den trees can be reserved from timber harvest and protected from damage during prescribed burns, so can escape trees. Such trees actually may serve multiple purposes as denning, escape, and resting sites.

Roads

Considerable attention has been given to the potential effects of roads on bears in the Southern Appalachians (Carr and Pelton 1984, Hillman and Yow 1986, Brody and Pelton 1989, Beringer and others 1990). The impacts of roads on bears are determined primarily by road structure, location, traffic characteristics, and timing of road use. Roads no longer in service and without management purpose should be closed and reforested, or seeded to serve as linear wildlife openings (Hillman and Yow 1986, Reed and others 1996). However, existing roads can have a useful function in a comprehensive fire management program. Forest roads can serve as firebreaks, primary ignition lines, and escape routes during prescribed burn operations; avenues into treated habitat for monitoring and research of the effects of prescribed fire programs on habitat and wildlife; and as access for wildfire suppression operations for the protection of bear habitat. Roads can be gated to reduce vehicular access into bear habitat, maintained as linear wildlife openings, and prescribed burned to encourage bear food plants within and adjacent to the roadbed, and still allow access for management or emergency purposes.

Summary

Black bears are not obligatory inhabitants of old growth forest or wilderness areas. Forests managed according to the basic concepts discussed in this paper have the potential for providing black bear habitat that is superior to unmanaged forests. Forest and wildlife managers should not be restricted in the methods and tools available to accomplish bear habitat management. Rarely is there only one “right” way to achieve a desired future forest condition. Management actions must be driven by overall goals and objectives. Prescriptive criteria should be formulated to effect burn conditions that will achieve management objectives for the burn unit, and such criteria must be strictly adhered to in “go-no go” decision making. Firing techniques must be appropriately selected and applied to meet burn objectives. Burns must be adequately monitored from ignition to completion to record fire behavior and intensity, and to ensure prescriptive criteria are being satisfied. Burn units must be of manageable size to minimize chance of fire escape, and to have reasonable opportunity to suppress the fire should prescriptive limits be exceeded, or if adverse conditions or circumstances are encountered. Burn plans must define methods to measure whether burns achieved specified burn objectives or not and why. Fire management plans must include provisions for monitoring of both first order fire effects and longer term impacts on habitat and associated wildlife responses. Results of fire management actions must be evaluated to ensure goals and objectives indeed are being accomplished, and future actions modified as necessary in an adaptive management fashion. Through such efforts, “best fire management practices” can be developed that will guide future management actions.

Black bear habitat and landscape management in the central hardwoods, including the use of prescribed fire and wildfire suppression, should be directed toward: (1) forest conservation; (2) creating and maintaining habitat diversity; (3) providing stable, diverse, and widely available food supplies throughout the year; (4) maintaining and increasing suitable denning sites and habitat; (5) providing escape cover; (6) conservation, enhancement, and establishment of habitat linkages; (7) reforestation programs; and (8) human access management. Responsible forest management can accommodate the needs of consumers, recreationists, residents, visitors, and bears. Bears and humans can coexist if habitat is available that satisfies the life requirements of bears, and if bears do not unduly interfere with human objectives for use of the same landscape.

As with any wildlife population, the objectives and attitudes of land owners, land managers, resource users, legislators, and the general public will determine if bears are considered a positive or negative benefit, and ultimately, if bears can survive. Expanding public awareness about the ecology and management of bears, and the utility and benefits of prescribed fire and other silvicultural techniques should encourage acceptance of agency and private programs designed to conserve, manage, and restore bear populations. Public education should include the concept of maintaining functional ecosystems, watersheds, and landscapes as the basis for conserving a unique wildlife heritage. A considerable amount of the wildlife habitat in the eastern United States is in private ownership. Therefore, it is crucial that private landowners also develop a vested interest in black bear conservation and management to ensure long-term welfare of the species. Technical advice and assistance with prescribed fire management of wildlife habitat should be provided to private landowners by federal and state agencies. Effective partnerships among agencies, universities, non-governmental organizations, private enterprise, and the public will aid in information transfer, cooperative programs, and increased funding, with the end result being increased fire management success for the benefit of bears, other wildlife, and humans in the central hardwood landscape.

Future topics for study concerning the relationship of bears, fire, and people in the central hardwood landscape include: (1) determining the most effective integration of prescribed fire and other silvicultural techniques to produce a productive and diverse forest; (2) effects of prescribed fire on hard mast, soft mast, squawroot, and herbaceous food production; (3) utility of prescribed fire in perpetuating a hard mast species component in the forest; (4) creation or perpetuation of escape cover through the use of prescribed fire; (5) use of burned areas by bears; (6) behavior of bears during prescribed burns; and (7) effective public education programs to convey the benefits of prescribed fire for wildlife habitat enhancement, forest resource production, hazard fuel reduction, and public use.

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Using Prescribed Fire to Regenerate Oaks

D.H. Van Lear[1], P.H. Brose[2], and P.D. Keyser[3]

Abstract

Fire is widely recognized as having been a major force shaping the character of eastern hardwood forests. Exclusion of periodic surface fires from mixed hardwood stands for over 70 years has contributed to the gradual succession of shade-tolerant species into the understory and midstory of these stands. Following major disturbance to the overstory, these shade-tolerant species or pioneer shade-intolerant species are able to out-compete oak regeneration and dominate the next stand. Because of fire exclusion for many decades, regeneration of oak-dominated stands has been a major silvicultural problem. Recent research indicates that a shelterwood-burn technique, which mimics the natural disturbance regime that historically favored oaks, can be used on productive upland sites to enhance the competitive position of oaks in the advance regeneration pool. This paper presents silvicultural guidelines for applying the technique and discusses several options to meet the management objectives of different landowners.

Introduction

Regenerating oaks on good quality sites has been a difficult problem throughout the eastern United States for many decades. Oak regeneration failures have generally been attributed to either poor initial establishment of oak seedlings or the slow juvenile growth of oak advance regeneration when it is present (Abrams 1992, Lorimer 1993, Loftis and McGee 1993). In the dense shade of mature mixed-hardwood stands, oak seedlings and seedling sprouts do not develop into competitive stems. Overstory removal by either partial or complete cuttings releases well-established shade tolerant regeneration and facilitates establishment of fast-growing shade intolerant seedlings. Subsequent stand development is to a mixed mesophytic forest with oak as a minor component or altogether absent (McGee 1979; Abrams 1992; Lorimer et al. 1994). This successional trend is a relatively recent phenomena, developing in the past 75 years, and is tied to the exclusion of fire from eastern hardwood forests (Little 1974; Van Lear and Johnson 1983; Lorimer 1993).

Fire research in hardwoods has lagged far behind that in pines, although several studies suggested that oaks were ecologically adapted to frequent burning (Swan 1970, Niering et al. 1970, Thor and Nichols 1974, Waldrop et al. 1987, Augspurger et al. 1987). All of these studies noted that oaks resisted root kill by fire to a greater extent than their competitors. Numerous authors, based on literature reviews, accounts of early explorers and settlers, observed vegetative patterns and responses, and other evidence, have concluded that fire played an important role in the development and maintenance of oak forests in the eastern United States (Little 1974; Van Lear and Johnson 1983; Crow 1988; Van Lear and Waldrop 1989; Williams 1989; Abrams 1992).

In this paper, we will discuss the use of fire as a silvicultural tool to regenerate oaks. Particular emphasis is placed on a new regeneration method that utilizes partial harvesting followed by prescribed fire.

Interactions Between Fire and Oaks

With the arrival of Indians in the eastern United States as early as 12,000 years ago, fire became a more frequent disturbance that shaped forest composition and structure (Pyne 1982; Williams 1989). Indians used fire for many reasons, including hunting, facilitating travel, stimulating berry production, clearing land for agricultural crops, and as a defense against enemies. The frequent, widespread use of fire by Indians and the European settlers that followed them created an environment favorable for the establishment and maintenance of oaks. In the early 1900s, fire-suppression efforts of the U.S. Forest Service and state forestry commissions began to be successful in reducing the frequency, extent, and influence of this powerful environmental force. The forest changed dramatically as fire was largely removed from the Eastern forest ecosystem. Shade-tolerant and fire-intolerant species began to dominate forest understories, overstory densities increased, and fire-sensitive species moved upslope from moist coves to xeric slopes.

Frequent burning creates environments that favor oaks on better-quality sites. Surface fires remove much of the mid- and understory strata in mature mixed hardwood stands, reducing shading. Spring fires are especially effective (Barnes and Van Lear 1998). Fire reduces the thickness of the forest floor, preparing a favorable seedbed for caching of acorns by squirrels and jays by (Darley-Hill and Johnson 1981, Galford et al. 1989). Fire reduces surface soil moisture which discourages establishment of mesophytic species (Barnes and Van Lear 1998) and may control insect predators of acorns and new seedlings (Galford et al. 1989).

The presence of oaks encourage surface fires because of the nature of their litter. An oak stand adds about 4.5 $\mathrm{Mgha^{-1}yr^{-1}}$ of leaf litter to the forest floor (Loomis 1975). This litter remains curly, creating a porous fuelbed for surface fires. Unlike leaf litter of mesophytic species which forms a flat mat upon compaction and decays rapidly, oak leaf litter undergoes little decay during the winter. In regions where snowpacks are heavy, oak litter recurls after snowmelt, once

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[1] Bowen Professor of Forestry, Department of Forest Resources, Clemson University, Clemson, SC 29634-1003

[2] Research Forester, USDA Forest Service, Warren, PA 16365

[3] Wildlife Biologist, Westvaco Corporation, Rupert, WV 25984

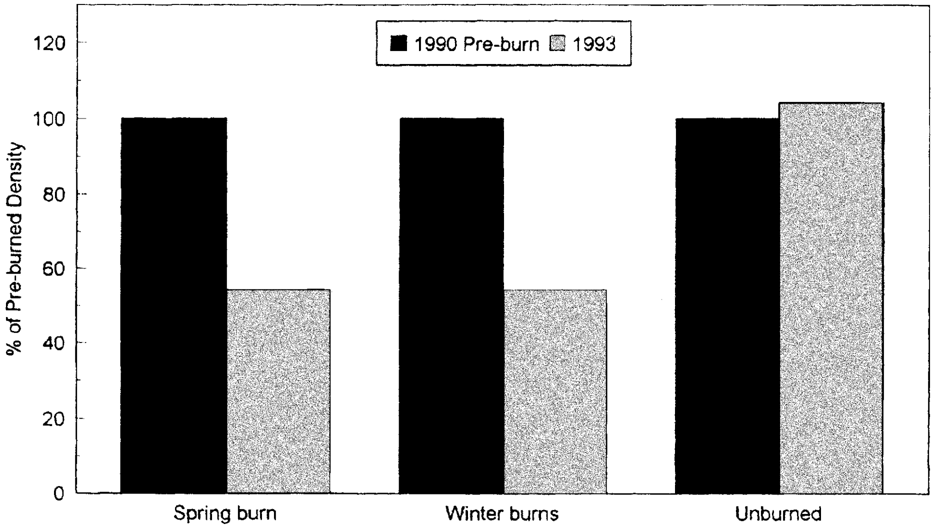


Figure 1.—Reduction of midstory/understory densities following three winter burns and one spring burn in mixed hardwood stands in the South Carolina Piedmont.

again creating a porous fuelbed capable of carrying a surface fire during the spring fire season (Lorimer 1989).

Because of the complexity of forest ecosystems, effects of fire in hardwood stands vary. Fires in stands of mixed composition have occasionally created oak-dominated stands (Roth and Hepting 1943; Carvell and Maxey 1969; Ward and Stephens 1989), probably because intense fires controlled competition and stimulated rapid growth of oak reproduction (Johnson et al. 1989; Lorimer 1989). In other cases, species composition in young stands has been altered very little by fire (Johnson 1974; McGee 1979; Waldrop et al. 1985; Augspurger et al. 1987).

Effects of fire vary because of differences in season of burning and fire intensity. Season of burning affects physiological condition of the plant and the ability of species to resprout. Hardwoods have the greatest ability to sprout when carbohydrate storage in their roots is high, i.e., in the dormant season. In the growing season, root reserves are lower and sprouting vigor is less. Fire intensity is critical because certain species, such as the oaks, can survive higher intensity fires than their competitors (Brose and Van Lear 1998) because their sprouts originate deeper in the soil than those of their competitors (Hane 1999).

Using Fire to Encourage Oak Regeneration

Understory Burning

Fire exclusion for most of the past century has altered stand structure and composition of eastern hardwood stands. Shade-tolerant and fire-intolerant species have gradually encroached into the understory of oak-dominated stands. Now, in the absence of periodic fire, there is no growing space for oak reproduction, which may or may not be present in the advanced regeneration pool. Van Lear and Watt (1993) developed a theoretical silvicultural prescription to encourage oak regeneration by burning the understory of mature mixed hardwood stands near the end of the rotation. Barnes and Van Lear (1998) tested this hypothesis in the Piedmont of South Carolina and found that one burn early in the growing season when leaves were expanding was as effective as three winter fires in reducing density of understory and midstory stems (fig. 1). The number of oak rootstocks in the regeneration layer was increased by burning, root/shoot ratios of oaks were enhanced, and competitive woody species decreased. There was little visible damage to boles of overstory oaks from the fires, especially in the larger size classes.

Although oak regeneration was increased by understory burning, it remained small and generally ephemeral. It appeared that burning would have to be continued for a relatively long period, i.e., perhaps 10 years or so, before sufficient oaks of competitive size would be present in the advance regeneration. Such an approach is handicapped by the expense and risks of multiple prescribed fires, making it a rather unattractive option for landowners and managers. Nevertheless, if no advance regeneration exists in a stand, periodic understory burning may provide a means to establish oak seedlings and seedling sprouts, which could then be encouraged to begin vigorous growth by the following technique.

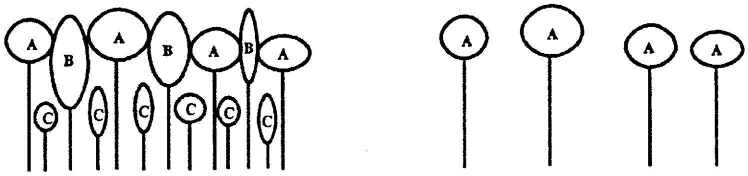
Shelterwood Cutting Followed by Prescribed Fire

The Shelterwood-Burn Method

Oak-dominated stands on better quality sites in the southern Piedmont and mountains often have abundant but small and non-competitive oak reproduction in the regeneration layer. When such stands are harvested by either clearcutting or the shelterwood method, oak reproduction cannot compete with rapidly growing shade-intolerant species and/or well-established shade-tolerant species (Loftis 1983, Abrams 1992, Schuler and Miller 1995).

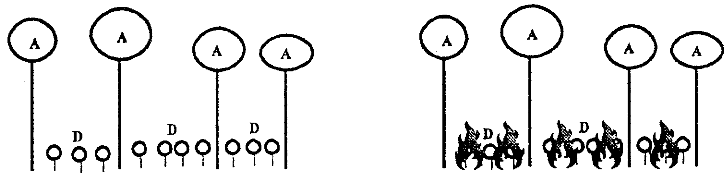
A shelterwood-burn method (fig. 2) has recently been developed in the Piedmont of the southeastern United States to enhance the competitive position of oak regeneration in such stands (Keyser et al. 1996; Brose and Van Lear 1998; Brose et al. 1999a). This method is based on the silvics and fire ecology of oak and yellow-poplar regeneration and involves an initial shelterwood harvest which removes roughly half of the overstory basal area. All yellow-poplars are removed in this first cut of the two-cut shelterwood method, leaving the best dominant and codominant oaks. Logging slash must be kept away from bases of residual oaks by directional felling. This partial harvest is followed by a 3- to 5-year waiting period, after which a relatively hot growing-season prescribed fire is run through the advance regeneration.

Oak reproduction must be relatively vigorous and free-to-grow to be competitive. We defined such oaks as straight stems at least 1.3 m tall with no major competitors within 3 m (Nix 1989). Oaks resist root kill by fire better than yellow-poplar and other competitors, especially as fire intensity increases (fig. 3) (Brose and Van Lear 1998). Density of free-to-grow oaks exceeded 800 stems/ha and yellow-poplar density was greatly reduced (up to 90%) in areas burned in the spring with high intensity flames. In contrast, winter burns provided little control of yellow-poplar and, even with a high intensity fire, oaks density did not reach 300 stems/ha. Summer fire resulted in substantial numbers of free-to-grow oaks in all fire intensity levels, but especially in the two medium intensity levels. High intensity summer fires killed many of the smaller oak seedlings while low intensity summer fires failed to control competition.



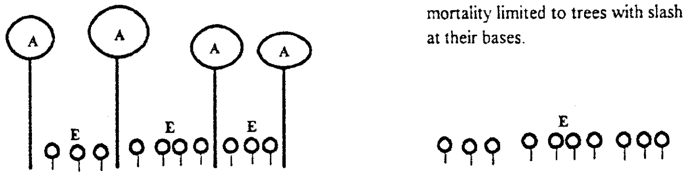
1. Typical upland mixed-hardwood stand.

2. Initial cut to a shelterwood (40 to 60% basal area reduction).



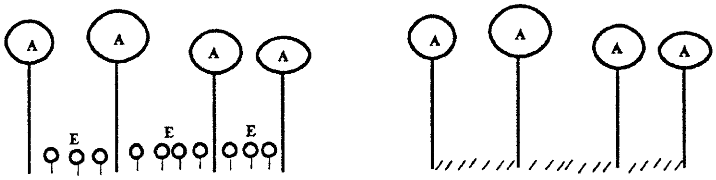
3. After 3-5 years, yellow-poplar dominates the advance regeneration pool.

4. Prescribed fire topkills the advance regeneration, forcing rootstocks to sprout. Overstory damage and mortality limited to trees with slash at their bases.



5. Oak now dominates the advance regeneration pool. Three management options available.

6a. Overstory harvested and additional fires withheld creates a new oak forest.



6b. Overstory retained and additional fires withheld creates a two-age stand.

6c. Repeat burning either stockpiles oak sprouts or creates an oak savanna.

Figure 2.—Schematic diagram of the shelterwood-burn technique. A = high quality oaks; B = hickories, poor quality oaks, and yellow-poplars; C = American beech, flowering dogwood, and red maple; D = mixed hardwood regeneration dominated by yellow-poplar; and E = mixed hardwood regeneration dominated by oaks.

Oak reproduction will not be uniform over the entire burned area. If free-to-grow oaks exceed 370/ha and 60% or more of the stocking plots have at least one free-to-grow oak, the stand can be considered regenerated with the likelihood that oaks will be a dominant component of the next stand. One burn may not be enough if oak dominance is desired in the new stand. Decades of fire exclusion have allowed oak competitors to become firmly established. If more oaks are desired, additional fires may be prescribed as dictated by leaf litter accumulation. We believe that oak dominance of the advance regeneration will continually increase with repetitive spring burning at about 2-year intervals.

We believe that a shelterwood cut, rather than a clearcut, is the essential first step in this technique. The shelterwood continues to produce oak litter which creates the continuous fine fuel bed necessary to carry the subsequent fire. If a clearcut had been used, foliage from yellow-poplar, a poor medium for carrying surface fires, would dominate the forest floor. In addition, shading from the shelterwood prevents yellow-poplar regeneration from growing so large during the interval before burning that it could not be killed by fire (Hane 1999).

In the 3 to 5- year period following the initial cut, logging slash settles and loses it’s foliage, minimizing the risk of bole damage to residual trees caused by flareups in heavy fuels (Brose and Van Lear 1998; Brose et al. 1999a). Although yellow-poplar was removed in the initial shelterwood cut, its seed remains viable in the duff for up to 10 years (McCarthy 1933). Most of this stored seed germinates during the waiting period and those seedlings are killed in the subsequent fire. The waiting period also allows residual overstory trees to recover from the shock of the initial cut before they will be stressed again by burning.

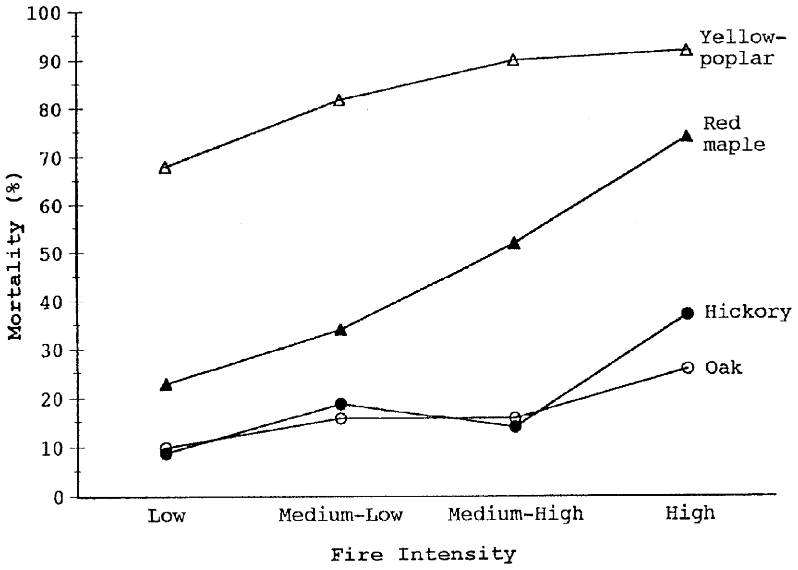


Figure 3.—Mortality (%) of hickory, oak, red maple, and yellow-poplar advance regeneration as fire intensity increases within spring prescribed burns conducted in shelterwood stands.

Advantages of the Shelterwood-burn Technique

The shelterwood-burn method is attractive to landowners because the initial cut of the shelterwood method produces immediate income. A small portion of the profit is then used to pay for the prescribed burn. Careful planning, e.g., consider using skid trails for fire breaks, before timber harvest can reduce later prescribed fire costs. Removal of the shelterwood after burning is at least as profitable as the initial cut because the best oaks were retained and some probably increased in value during the intervening years before final harvest.

If the landowner's goal is to manage for wildlife, the shelterwood-burn method can be used to sustain hard mast production while providing a source of palatable browse during the regeneration period. Landowners can maintain indefinitely the classic structure of the shelterwood while stockpiling oak regeneration with periodic burns (Brose et al. 1999b). Many upland game and non-game species utilize the mast, browse, and cover in a regenerating shelterwood (Brose et al. 1999b; Lanham et al. 2000). A note of caution – in areas where deer density is extremely high and early successional habitat is relatively rare, the shelterwood-burn method probably will not work because of over-browsing of oak regeneration.

The shelterwood-burn system could be used to restore fire-maintained ecosystems (Brose et al. 1999b). Frequent growing season burns after the initial shelterwood cut would gradually reduce woody regeneration (even the oaks) and create a hardwood woodland or savannah, two habitats that have become increasingly rare after decades of fire exclusion (Buckner 1983; Pyne 1982; Van Lear and Waldrop 1989; Abrams 1992).

Research suggests that the shelterwood-burn technique may be successful in other physiographic regions and on different sites. Repeated burning in oak-pine communities on xeric sites in the Cumberland Plateau reduced regeneration of red maple and other non-oak species and promoted chestnut oak regeneration (Arthur et al. 1998). On mesic sites in Wisconsin, two spring burns reduced densities of sugar maple and hophornbeam by 80%, while density of northern red oak increased (Kruger 1992). Ward and Gluck (1999), in Connecticut, observed that burning several years after a shelterwood harvest favored oaks and reduced competition from birch and shrubs. Hot fires in mountain laurel thickets in the Northeastern United States that opened canopies, i.e., a disturbance similar to a shelterwood harvest, allowed oak reproduction to grow past the dense shrub layer (Moser et al. 1996).

Conclusions

Oak forests on good quality sites have been in decline in the eastern United States for decades as more shade-tolerant species gradually succeed oaks or shade-intolerant species outgrew them after a major disturbance. Fire historically played a major role in maintaining oak-dominated forests but periodic surface fires have been excluded from these sites for over 70 years.

Practical silvicultural prescriptions using fire for oak regeneration have been lacking. Understory burning in mature mixed hardwood stands creates environmental conditions which should favor oak regeneration but developing oak reproduction does not reach sufficient size to be competitive when the overstory is removed.

A shelterwood-burn technique has recently been developed which overcomes this major disadvantage. This technique reduces the vigor of oaks' competitors, especially if growing season burns of relatively high intensity are used, and provides conditions (increased light and reduced growth of competitors) which allow oak regeneration to vigorously resprout following the subsequent burn. It mimics a disturbance pattern (i.e., partial overstory disturbance followed by fire) that has shaped the composition of eastern forests for millennia. The shelterwood-burn method is also economically attractive because the initial shelterwood cut yields an immediate income and the prescribed burn is low cost compared to alternative treatments.

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Current Research on Restoring Ridgetop Pine Communities With Stand Replacement Fire

Thomas A. Waldrop[1], *Nicole Turrill Welch*[2], *Patrick H. Brose*[3], *Katherine J. Elliott*[4], *Helen H. Mohr*[1], *Ellen A. Gray*[5], *Frank H. Tainter*[6], and *Lisa E. Ellis*[6]

Abstract

Ridgetop pine communities of the Southern Appalachian Mountains historically have been maintained by lightning- and human-caused fires. With fire suppression for several decades, characteristic stands are entering later seral stages. They typically have an overstory of Table Mountain (*Pinus pungens*) and/or pitch pine (*P. rigida*), a midstory of chestnut oak (*Quercus prinus*), scarlet oak (*Q. coccinea*) and blackgum *(Nyssa sylvatica*), and a shrub layer of dense mountain laurel (*Kalmia latifolia*). Previous research suggests that restoration of these communities can be accomplished with high-intensity fires that open the forest canopy and expose mineral soil. Three recent studies examined plant-community response to high-intensity prescribed fires. A series of corollary studies help to explain some of the results of these field studies. High and medium-high intensity fires provided adequate sunlight for pine seedlings, whereas medium-low and low intensity fires did not. Post-burn duff was deep and did not vary by fire intensity. We observed sufficient seedling densities to restore pine-dominated stands after all but the highest intensity fires. Many seedlings survived the first growing season as their roots penetrated duff to reach mineral soil. Hardwood rootstocks resprouted on sites treated with all fire intensities and may out-compete pine seedlings for available resources. High-intensity fires may have reduced mycorrhizal abundance and moisture availability for new germinants. Fires of lower intensity than previously recommended or multiple fires of very low-intensity may best provide conditions for pine regeneration, but additional research is needed.

Introduction

Table Mountain pine (*Pinus pungens*) and pitch pine (*P. rigida*) are ridgetop Appalachian endemics that historically were maintained by lightning- and human-ignited fires. Stands of Table Mountain and pitch pines present on the landscape today were established by fires, logging, insects, and diseases that occurred in the early twentieth century, the most recent landscape-scale disturbances. Stand-replacing fires have been nearly absent in the region since then (Williams 1998). Sutherland and others (1995) found a Table Mountain pine community in Virginia that was maintained by major fires occurring approximately every 10 years until the 1940's. Fire suppression and lack of other disturbance have allowed ridgetop pine communities to succeed to hardwood dominance and closed understories (Williams and Johnson 1992, Sutherland and others 1995, Turrill 1998, Williams 1998). As a result of these changes in dominance and structure, Table Mountain/pitch pine woodlands are recognized by the Southern Appalachian Assessment as one of thirty-one rare communities in the Southern Appalachian Mountains (SAMAB 1996).

Most research addressing the role of fire in Table Mountain pine stands has been limited to post-wildfire studies, which suggest that high-intensity prescribed fires that remove the forest canopy and expose mineral soil will help ensure successful regeneration. Zobel (1969) found that serotinous cones opened in lightly burned areas, but that seedlings survived only where fires killed overstory trees and erosion exposed mineral soil. Likewise, Sanders (1992) observed the greatest proportion of Table Mountain pine seedlings in high- and moderate-intensity burn areas, where the canopy was open and mineral soil exposed. Williams and Johnson (1992) found that seeds were abundant on the ground in lightly disturbed stands where no fire occurred. However, seedlings were successful only on microsites with thin litter layers (<1.5 in.) and where the canopy was more open than in surrounding stands. Such microsites usually were created by ice storms (Williams 1998).

Many ridgetop pine stands are located on National Forest System land and National Parks, where prescribed burning is possible and encouraged. Although many National Forest and National Park land management plans prescribe high-intensity fire for ridgetop pine communities, carrying out such burns is difficult (Turrill 1998). Such prescriptions provide a narrow window of opportunity and raise questions about worker safety and smoke management (Waldrop and Brose 1999). In addition, some land managers avoid using high-intensity fires because of the perceived risk of damaging marketable hardwoods and the inability to control such fires on steep slopes (Van Lear and Waldrop 1989). As a result, high-intensity prescribed burning has had limited application in the Southern Appalachian Mountains.

Williams (1998) suggested that Table Mountain/pitch pine stands are in decline as a result of fire suppression policies and inadequate understanding of the species regeneration biology. To date, only three studies have conducted prescribed burns to better understand the conditions necessary for pine regeneration. Together, these three studies examine community response to varying degrees of fire intensity, as well as seedling establishment in varying types of microhabitat. This paper examines the results of

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[1] USDA Forest Service, Southern Research Station, Clemson, SC

[2] School of Public and Environmental Affairs, Indiana University, Bloomington, IN

[3] USDA Forest Service, Northeastern Research Station, Warren, PA

[4] USDA Forest Service, Southern Research Station, Otto, NC

[5] Department of Forestry, Wildlife and Fisheries, University of Tennessee, Knoxville, TN

[6] Clemson University, Department of Forest Resources, Clemson, SC

those studies to evaluate the fully open canopy and exposed mineral soil conditions generally accepted as necessary to regenerate ridgetop pines.

Current Research on Stand-replacement Prescribed Burning

Effects of Fire Intensity on Table Mountain/ Pitch Pine Regeneration

Three studies have examined the response of Table Mountain pine and pitch pine stands to stand-replacement prescribed fire: Turrill (1998), Waldrop and Brose (1999), and Elliott and others (1999). The burns conducted for these studies varied in their effects on opening the forest canopy and removing litter and duff. Comparisons of these field studies allow an evaluation of the amount of pine regeneration under natural conditions.

Several corollary studies provide insight to disturbance history and methods of evaluating stands for their potential of regeneration success. Waldrop and others (1999) conducted a greenhouse study to evaluate the effects of shade and duff on seedling establishment. Ongoing studies include the dendrochronology of ridgetop pine stands across the Southern Appalachians, seed biology of Table Mountain Pine, and mycorrhizal associations in burned Table Mountain pine stands. We will discuss the preliminary results of each.

The prescribed burn observed by Waldrop and Brose (1999) was on the War Woman Wildlife Management Area of the Tallulah Ranger District of the Chattahoochee National Forest. Prior to burning, mean total basal area in study stands was 123.0 ft^2 per ac. Hardwoods made up 98.8 ft^2 of this total and pines the remaining 38.8 ft^2. Chestnut oak was the predominant hardwood and almost all pines were Table Mountain pine. USDA Forest Service personnel conducted a stand-replacement prescribed fire on a 850-ac unit in April 1997. The burn area included sharp ridgetops and steep slopes with northeastern or southwestern aspects. The fire was ignited by hand and by helicopter to create a ring fire that reached greatest intensity within ridgetop Table Mountain pine stands. The fire was large enough and its intensity varied enough to allow comparisons of regeneration success among areas burned at different intensities. This ongoing study will include four additional burns: in Georgia, South Carolina, and Tennessee during late winter 2000 or the winter of 2000-2001.

Turrill (1998) studied a 7.5-ac prescribed fire on the Grandfather Ranger District of the Pisgah National Forest. The stand's mean total basal area was 140.7^2 ft per ac prior to the burn. Hardwoods comprised 37.9 ft^2 per ac, and pines 102.8 ft^2 per ac. Blackgum was the predominant hardwood. The pine component was 51 percent Table Mountain pine, 39 percent pitch pine, and 10 percent Virginia pine (*P. virginiana*). USDA Forest Service crews used a combined ring and head fire technique to burn the stand in May 1996.

Elliott and others (1999) studied a roughly 750-ac burn on Wine Springs Creek on the Nantahala National Forest in western North Carolina. The ridgetop community was dominated by pitch pine, chestnut oak, scarlet oak, and red maple, which comprised 49, 19, 8, and 8 percent of the total basal area, respectively. This south-facing burn unit was ignited by helicopter in April 1995 in a pattern to create a mosaic of fire intensities. Low intensity fire occurred on the lower slopes and flames reached into the crowns of trees along the ridgetop.

Stands burned for Turrill (1998), Waldrop and Brose (1999), and Elliott and others (1999) contained dense mountain laurel shrub layers. Post-burn observations in both study areas were completed in the first growing season after burning.

The prescriptions applied in these studies produced four fire intensities defined by Waldrop and Brose (1999): low, medium-low, medium-high, and high. Waldrop and Brose (1999) observed all four fire intensities, while Turrill (1998) and Elliott and others (1999) observed only medium-low intensity. Waldrop and Brose (1999) give a detailed description of the discriminant functions used to classify fire intensity, which generally can be described as follows: Flames of low intensity fires never reached into the crown of trees and uniformly burned the area. Medium-low-intensity fires had flames slightly taller than those of low-intensity fire; they burned less uniformly and produced hot spots where flames reached into crowns and killed large trees. Flames of medium-high intensity fires typically reached into the crowns of all overstory trees. Flames of high-intensity fires generally exceeded the crowns of overstory trees and carried from crown to crown.

High- and medium-high-intensity fires reduced canopy cover, leaving only 4.4 ft^2 and 7.0 ft^2 per acre of basal area, respectively (table 1). Mortality was high in all diameter size classes following both high- and medium-high-intensity fires. Sunlight reaching the forest floor may have been adequate for seedling survival following fires of both intensities.

Medium-low- and low-intensity fires reduced canopy cover (table 1), but residual basal area may be too high in all three studies to allow stand replacement. Medium-low-intensity fires reduced basal area to 48.3 ft^2 per acre in Waldrop and Brose (1999), 112.8 ft^2 per acre in Turrill (1998), and to 82.1 ft^2 per acre in Elliott and others (1999). Low-intensity fires had little effect on basal area, leaving 98.9 ft^2 per acre. Mortality was greatest in lower d.b.h. classes (< 6 in. d.b.h.) following fires of medium-low and low-intensity. Shade from surviving trees may have prevented pine seedling survival.

Prolific hardwood sprouting was observed following fires of all intensities (table 1). Waldrop and Brose (1999) reported that all intensities top-killed the mountain laurel shrub layer; Turrill (1998) and Elliott and others (1999) reported that medium-low intensity fires did so. Generally, under all fire intensities there were over 10,000 stems per acre, and they were growing rapidly. Competition from these sprouts may eliminate any pine regeneration after a fire of any fire intensity. This result suggests that multiple, low-intensity fires

Table 1.—Characteristics of Table Mountain and pitch pine stands one year following stand replacement prescribed burning

	Fire Intensity Level				
Variable	Low	Medium Low	Medium High	High	Source
Pine basal area (ft^2/ac)	25.7	26.1 94.1 42.1	4.8	0.0	Waldrop and Brose (1999) Turrill (1998) Elliott and others (1999)
Hardwood basal area (ft^2/ac)	73.2	22.2 18.7 40.0	2.2	4.4	Waldrop and Brose (1999) Turrill (1998) Elliott and others (1999)
Total basal area (ft^2/ac) 82.1	98.9	48.3 112.8	7.0	4.4	Waldrop and Brose (1999) Turrill (1998) Elliott and others (1999)
Hardwood sprouts (num/ac)	13,016	15,130 929 12,354	10,765	12,768	Waldrop and Brose (1999) Turrill (1998) Elliott and others (1999)
Pine seedlings (num/ac)	5,608	9,130 3,117 ~300	3,650	1,396	Waldrop and Brose (1999) Turrill (1998) Elliott and others (1999)

may be necessary to reduce hardwood abundance while maintaining a seed source among large pines.

Post-burn counts of Table Mountain pine seedlings reported by Waldrop and Brose (1999) and Turrill (1998) suggest that fires were of sufficient intensity to open serotinous cones throughout the burn unit, even in areas of low-intensity burns. Post-burn pine density ranged from 1,400 to more than 9,000 stems per acre (table 1). An unexpected result was that the lowest pine densities were in areas burned at the highest intensity. This suggests that cones were consumed or seeds killed by intense heat, or that the seedbed became less suitable.

Although plots in high-intensity burn areas had fewer seedlings, if they are well dispersed, the 1,396 seedlings per acre present in those areas should create pine-dominated stands. However, Table Mountain pine seedlings were found at only 51 percent of the sampling points, indicating that portions of burned areas had no pine regeneration. Hardwoods may dominate such areas. Plots in areas burned at medium-high intensity also indicated low pine stocking (64 percent). If seedlings receive enough sunlight, pine density and stocking levels in those areas burned at low and medium-low intensities should be sufficient to create pine-dominated stands.

Pitch pine does not have serotinous cones in the region studied by Elliott and others (1999). Therefore, managers were concerned that a winter or spring burn would consume seeds already on the ground. In this study, over 1,100 pitch pine seedlings per acre were present the year after burning, indicating that the seed source was still viable. Among those seedlings, however, fewer than 300 per acre survived. The authors suggested that mortality was due to shading by the surviving overstory, competition from sprouts, and a thick duff layer that prevented seedling roots from reaching mineral soil.

Pine seedlings regenerated on relatively thick duff following all fire intensities in the Waldrop and Brose (1999) study. Total litter and duff depth remaining after fires was 2.1, 1.5, 2.5, and 2.6 in. for the low-, medium-low-, medium-high-, and high-intensity fires, respectively. The percentage of seedlings with roots penetrating mineral soil was 71.1, 94.6, 63.0, and 56.1 for the same order of fire intensities (Waldrop and Brose 1999). Turrill (1998) observed pine regeneration on approximately 3.6 in. of combined litter and duff. Waldrop and Brose (1999) found that root systems of over 80 percent of the sampled seedlings were able to penetrate duff up to 3.0 in., indicating that duff removal may not be as critical as once thought. However, seedling survival was not tracked beyond the first growing season in either study.

Corollary Studies

Dendrochronology

Little is known about the disturbance history of Table Mountain/pitch pine stands. The species may have been maintained by frequent low- to medium-intensity fires, infrequent high-intensity stand-replacing fires, or a combination of both. The dendrochronology study of Sutherland and others (1995) provides valuable insight to fire frequency and stand dynamics for one Virginia site. A similar study is being done by Brose, Tainter, and Waldrop

Table 2.—Age distribution of Table Mountain pines sampled on two north Georgia sites

Age Class (yrs)	n	Pct of Sampled Trees
1-50	0	0
51-75	9	33
76-100	9	33
101-125	3	11
125-158	6	22

(unpublished study plan) in conjunction with the study sites reported by Waldrop and Brose (1999). Cores have been extracted from overstory trees and understory trees and shrubs in two burn units in Georgia, two in South Carolina, and one in Tennessee.

A preliminary analysis of stand dynamics in the two Georgia units suggests a history of frequent disturbance that lasted until the 1950's (table 2). Pines in the dominant canopy position are between 100 and 158 years old. However, numerous smaller pines are between 50 and 100 years old. Shrubs, particularly mountain laurel, are less than 50 years old, and there are no pines younger than 50 years. The frequency pattern of pine age classes (table 2) indicates that pines were regenerating from the 1850's through the 1950's, and that these stands were relatively open. Well-established fire suppression policies in the 1950's allowed the shrub layer to become dominant and prevent continuing pine regeneration. Restoration of these stands will likely require some means to remove shrubs and competing hardwoods.

Seed biology

In the past, studies of prescribed burning assumed an adequate seed source that did not vary among stands or stand conditions. Any regeneration failures could have been caused by an inadequate seed source. An ongoing study by Gray, Rennie, and Waldrop (unpublished study plan) will help identify stands that have an adequate seed source for regeneration. Such studies will help managers determine the abundance and viability of seed from a range of tree ages, as well as from cones of different ages. Preliminary results for seed viability are given in table 3. Seed viability was moderate, generally between 20 and 50 percent, from cones of all ages and from trees older than 10 years. Viability did not appear to vary by age after trees reached 10 years. However, viability seemed to increase as cones matured to 4 or 5 years old. These results indicate that, if cone numbers are adequate, stands over a wide range of ages may be considered as candidates for burning. A surprising result is the presence of cones with viable seed on young trees. Trees within the 5- to 10-year age class had 3-year-old cones with 23 percent seed viability. This result suggests that Table Mountain pines are adapted to regenerating under regimes of low-intensity fires, which may occur every 5 to 10 years.

Mycorrhizae

The need for mycorrhizae is generally accepted for southern pine seedlings grown in nurseries, but it has not been studied for nontimber species such as Table Mountain pine. Both ectomycorrhizae and vesicular-arbuscular mycorrhizae may be necessary for survival of Table Mountain pine seedlings, but their respective roles in Table Mountain pine regeneration and their responses to high-intensity fires have not been considered. Neary and others (2000) suggested that fire intensity strongly affects the degree and duration of reduced soil microbial activity. Neary and others (2000) also suggested that after a period of low activity, microbial populations increase in areas burned at high intensity to a level much higher than in unburned areas, or in areas burned at low intensity. This pattern may suggest that Table Mountain pine seedlings will not develop for some time after high-intensity burning; but that they may eventually benefit from increased microbial populations. If so, prescribed burns should be conducted far enough in advance of spring germination to allow these populations to recover.

An ongoing study by Ellis, Tainter, and Waldrop (unpublished study plan) is examining the relationship of fire intensity to mycorrhizal development on Table Mountain pine roots. First- and second-year seedlings were collected on 22 plots established for the Waldrop and Brose (1999) study. Seedling root biomass was quantified, ectomycorrhizal root tips were characterized, and their presence compared by seedling age, slope aspect, and microsite. Sites burned at low and medium-low intensities had twice as many mycorrhizal root tips (40 percent) than sites burned with medium high and high intensities (22 percent). Two-year-old seedlings had twice as many mycorrhizal root tips than one-year-old seedlings. Seedlings on slopes of western aspect had twice as many mycorrhizal root tips than those on

Table 3.—Percent viability of Table Mountain pine seed by tree age and cone age within a tree

	Cone Age				
Tree age class (years)	2 years	3 years	4 years	5 years	All Cone Ages
5 to 10	8	23	1	-	-
11 to 25	20	32	41	23	27
26 to 50	33	11	24	56	31
51 to 75	29	20	34	36	30
75+	29	13	54	39	33
All tree age classes	24	21	34	36	

eastern aspects. Based on morphological and histological examinations of mycorrhizal root tips, *Pisolithus tinctorius*, *Suillus granulatus*, and *Cenococcum* spp. were identified as the predominant symbionts. Although these results are preliminary, they may indicate that the very high-intensity fires had adverse effects on mycorrhizal development, thus reducing pine regeneration success.

Seedbed habitat

In order to assess seedling establishment, Waldrop and others (1999) conducted a greenhouse study that used shade and duff treatment combinations similar to those observed in the field. Duff categories included depths of 0, 2, and 4 in.; and shade levels included 0, 30, 63, and 85 percent shade. Table Mountain pine seeds were collected on the Chattahoochee National Forest in Georgia. Soil and duff were collected from a recently burned Table Mountain pine stand on the Sumter National Forest, South Carolina. Seeds were germinated and allowed to grow under these conditions for 3 months. We compared survival of seedlings grown in the greenhouse to field seedling survival in the burn described by Waldrop and Brose (1999).

Figure 1 shows the total number of seedlings per plot in all combinations of duff and shade at the end of the 90-day greenhouse study. Stem density typically was greater in 2-in. duff than in bare soil or 4-in. duff. This pattern remained constant for all except the 0-shade category. In 0 shade, stem densities in pots with 2 in. of duff were equal to stem densities in pots without duff. Without shade, the mulching effect of a 2-in. duff layer may not have been adequate to prevent moisture deficit and seedling death.

Lack of shade reduced seed germination and the survival of germinants, while heavy shade reduced survival; more seedlings become established under 30-percent shade than under full light or the higher shade levels. This pattern was constant among pots with 2 and 4 in. of duff but differed among pots with no duff (fig. 1). With no duff, fewer seedlings per pot occurred under 30-percent shade than under no shade, although this difference was not significant. Without the mulching effect of duff, 30-percent shade may not be adequate to prevent moisture deficit.

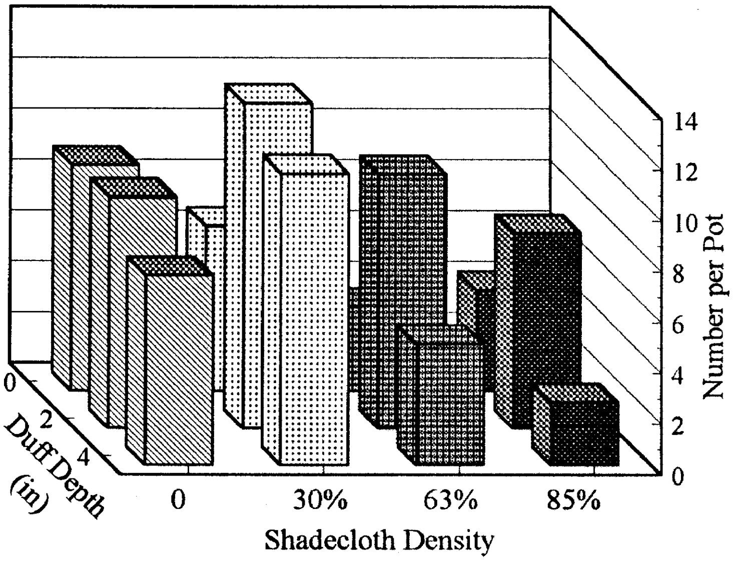

Figure 1.—Seedling density per pot after the 90-day greenhouse study for all combinations of shade level and duff depth.

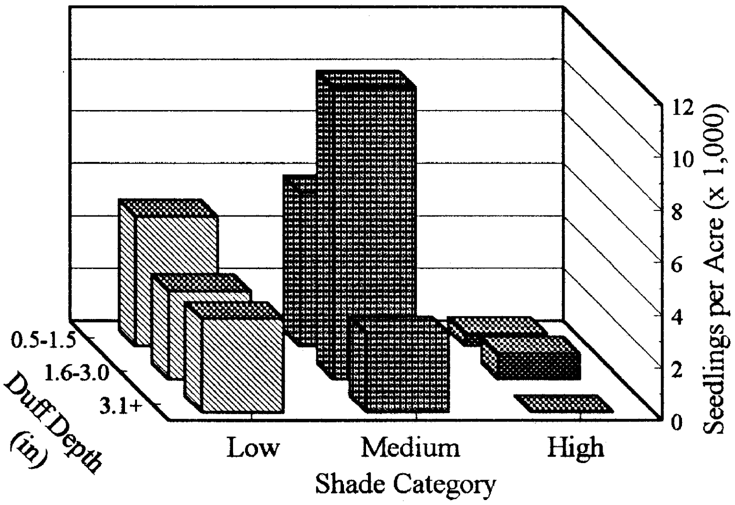

Figure 2.—Seedling density in Georgia and South Carolina burn units at the end of one growing season for all combinations of shade and duff depth categories.

If germination and survival in the field follow the same patterns as in the greenhouse, these data provide a partial description of seedbed conditions necessary to establish Table Mountain pine. Because of differences in study designs, field results shown here do not provide a direct comparison to greenhouse results. However, results of the two studies are similar. In the field, stem numbers did not vary significantly at different duff depths within a shade category (fig. 2). Seedling numbers were not significantly different between low- and medium-shade categories, but both had significantly more stems than did the high-shade category. Under high shade, stem density was less than 1,000 seedlings per acre at all duff depths, probably too few to adequately regenerate a stand. Stem numbers in medium and low shade ranged from 3,024 per acre for medium shade with over 3 inches of duff to over 11,000 stems per acre under medium shade and 1.6 to 3.0 inches of duff. Each of these stem densities probably exceeds the minimum needed to regenerate the stand.

The moderate levels of shade and duff that this study suggests are optimum seedbed habitat differ somewhat from previous recommendations. Although the exact fire regimes necessary to create this type of habitat are unknown, these results do not suggest that a single high-intensity fire is mandatory. Multiple lower-intensity fires could maintain an overstory and seed source while reducing the duff without exposing mineral soil.

Conclusions

The three studies described here represent the first attempts to restore ridgetop pine communities in the Southern Appalachians with prescribed stand replacement fires. Such fires have been attractive for a number of reasons: they provide a means of killing overstory trees and opening the forest floor to direct sunlight; they provide the heat needed to open serotinous cones; and they reduce thick duff layers or expose mineral soil. However, none of the fires observed in these studies should be considered successful for replacing older stands of mixed pines and hardwoods with newly regenerated stands of pines. Low-intensity fires observed by Waldrop and Brose (1999) and medium-low intensity fires observed in all three studies failed to kill more than a few overstory trees. High intensity fires killed most overstory trees but had few pine seedlings. Medium-high fires provided abundant overstory mortality and pine regeneration. However, fires of all intensities failed to control competition from hardwood and shrub sprouts.

Competition and shading from hardwoods and shrubs that sprout after burning may inhibit the development of a pine-dominated stand. Post-fire sprouting occurred more frequently in hardwood tree species (red maple, chestnut oak, and scarlet oak) than in shrub species (mountain laurel). The ability of Table Mountain and pitch pine seedlings to compete with the regeneration of other species is unknown. Frequent burning may be necessary to reduce hardwood sprout vigor.

The corollary studies presented here provide indirect evidence that ridgetop pine communities may be restored by frequent burning. The dendrochronology study shows that pines in study stands were uneven-aged and had regenerated frequently until the time of fire exclusion. The seed biology study suggests that a viable seed source is present over a wide range of tree ages and in cones that have been on trees for up to 5 years. This study also shows that very young trees produce viable seed, suggesting an adaptation to frequent burning. Studies of seedbed habitat and mycorrhizal populations provide evidence that the severe conditions produced by high-intensity burning are not necessary and may be detrimental to regeneration. Moisture may be limited due to lack of mycorrhizal tips on roots, loss of a mulching effect from the duff, and direct sunlight reaching the forest floor. Moderate levels of shade with some duff present were optimum for seedling survival, both in the greenhouse and in the field. These conditions may have been common in pre-1950’s stands that burned often. Thick duff was a barrier to seedling development in all field studies but seedling roots can penetrate duff up to 3 inches thick.

The results presented here suggest that ridgetop pine stands were created by lower-intensity fires than once were thought necessary, and that such fires would aid in stand restoration. Low-intensity prescribed fires, which can be used when the lower layers of the forest floor are moist, are less dangerous and present a larger window of opportunity than high-intensity fires. Low-intensity fires also decrease erosion potential on steep slopes and loss of site productivity.

There is still much to learn about restoring ridgetop pine stands. If seedlings continue to survive among sprout competition, a single medium-high intensity fire may prove sufficient. However, these results were drawn from studies which tracked seedling survival and overstory mortality for only one growing season. More research is necessary before definitive fire plans can be developed for ridgetop pine communities. These studies should apply prescriptions to achieve medium-high intensity burns and observe post-burn canopy cover, seedling density, seedling rooting depth, and seedling survival over several growing seasons. Additional research also is needed to test fires in other seasons and multiple low-intensity burns. Such fires should be conducted so as to control hardwood and shrub sprouting while maintaining healthy overstory pines as a seed source. Serotinous cones can open with the low-intensity fires (Zobel 1969, Waldrop and Brose 1999), indicating that the reduced fuel loads after multiple burns would not reduce fire intensity too much to allow cones to open.

Many questions remain about the ecology of Table Mountain and pitch pines. In particular, the competitive ability of these species is unknown. Physical, chemical, and biological properties of soils in ridgetop stands are likely to be affected by regeneration burns. These properties may affect seedbed conditions but they have not been studied. Finally, natural disturbances, other than fire, may have played an historical role in perpetuating the species; so alternative management strategies may also improve regeneration.

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Barrens Management and Restoration: A Kentucky Example

Joyce Bender[1]

The Kentucky State Nature Preserves Commission is charged with the responsibility to inventory the state for rare species and unique ecological communities and to protect them in perpetuity in a system of nature preserves. Presently, there are thirty-nine state nature preserves and nine of them protect barrens communities. Barrens can be described as sparsely wooded ecological communities with thin rocky soils underlain by sandstone or limestone. Barrens are characterized by tree cover ranging from 25 to 60 percent and a ground cover dominated by perennial grasses and prairie forbs, some of which are listed as rare species in Kentucky. Protecting these preserves in perpetuity requires active management to maintain open habitat which sustains the grasses and forbs. Years of fire suppression have resulted in canopy closure and the disappearance of some species. Efforts in Kentucky are focused on restoration of overgrown barrens. Prescribed fire is one tool to reduce canopy and midstory cover. Fire also removes litter that can inhibit seedling establishment and limit plant growth. Examples of barrens communities in Kentucky will be presented along with agency goals for restoration and management.

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[1] Kentucky State Nature Preserves Commission, 801 Schenkel Lane, Frankfort, KY 40601, ph.: 502-573-2886, email: Joyce.Bender@mail.state.ky.us

Resistance is not futile: The response of hardwoods to fire-caused wounding

Elaine Kennedy Sutherland[1] *and Kevin T. Smith*[2]

Abstract

Fires wound trees; but not all of them, and not always. Specific fire behavior and differences among tree species and individual trees produce variable patterns of wounding and wound response. Our work focuses on the relationships between fire behavior and tree biology to better understand how hardwood trees resist injury to the lower stem and either survive or succumb to low-intensity fire. Our objectives here were to 1) define and describe the wounding process, 2) to describe compartmentalization and wound closure and 3) to discuss species-specific differences among several common hardwood trees in the resistance to injury and resilience after wounding. Characteristics of fire scars are summarized.

Introduction

Prescribed fire is a frequently discussed and under-utilized tool to restore upland oak (*Quercus*) communities in Central Hardwoods forests. One barrier to the use of prescribed fire is a lack of understanding of how fire, especially low-intensity prescribed fire, affects individual trees and tree species. Fires wound tree stems (Gustafson 1946, Kaufert 1933, Nelson and others 1933, Wendel and Smith 1986), but not all of them, and not always (Smith and Sutherland 1999). Patchiness of fire behavior and different susceptibilities among individual trees produce variable patterns of wounding. Additionally, tree species differ in their effectiveness to resist injury and the spread of infection after wounding. Knowledge about potential wounding from fire may be an important consideration in forest managers' decision-making processes. For example, some managers may be concerned about loss of economic value through damage to potential high-quality veneer logs. Alternatively, those burning to restore the structure and composition of oak communities undergoing succession to mesic forest types might want to maximize injury to undesirable species (such as maple (*Acer*)). Here, we define and describe the processes of fire-caused wounding, the generalized response of hardwood trees to wounding, and species-specific differences in protection from fire injury and effectiveness in the tree wound response.

How Fires Wound Trees

In hardwood forests, fires typically burn in light fuels (hardwood litter) and are low in intensity (Komarek 1974). Not all trees are wounded by any one low-intensity fire (Smith and Sutherland 1999) because within a fire perimeter there are usually unburned patches; obviously, trees in these patches are unlikely to be injured. The probability that a hardwood tree will be wounded during a fire depends on many factors— primarily weather, fire behavior and bark characteristics. Fire wounds the vascular cambium by lethal heating, which depends on flame length, fire temperature, and duration of heating (Ryan 1998). These fire behaviors are in turn affected by fuel quantity and moisture content, ambient temperature, wind, slope, and in prescribed fires, by ignition pattern (Ryan 1998). Wildfires differ from prescribed fires. By definition and by regulation, prescribed fires should only be ignited when they are judged controllable, under specific parameters of weather and fuel conditions. Thus, the fire manager igniting a prescribed fire has some control over fire behavior and by extension, the probability of tree injury and damage.

By itself, scorch is not an indicator of injury. Fire injury is the result of fire causing a wound, a disruption of living tissues including the vascular cambium that results in an impairment or loss of function. These wounds are typically referred to as fire scars. However, not all scorched trees are injured; scorch is simply a sign or indicator of a fire occurrence. Further, a scorched or even an injured tree is not necessarily damaged: damage involves a loss of desirable wood characteristics, value, or usefulness. The determination of damage is relative to management goals. Damage due to a fire scar may be essentially zero to a tree in a conservation/preservation area, be worth thousands of dollars to an identical tree managed to produce high-quality veneer, or actually be desirable for wildlife management goals (e.g., to promote habitat for animals that dwell or feed under dead bark or in cavities). For timber management, methods to estimate cull as a result of fire wounding were developed by Hepting (1941) and Loomis (1973,1974). Notably, estimated cull is virtually negligible when externally visible wounds are less than 2 inches inches width at 1 foot aboveground (Hepting 1941). Loomis (1974) suggested that oak trees with wounds less than 6 in wide are unlikely to lose quality and no more than 3 board feet in volume, and that pole-sized trees were unlikely to lose any quality. Unpublished data of the authors indicates that most prescribed fire-caused wounds were less than 1 inch in width.

Most fire-caused injuries from low-intensity fires result from heating without combustion. Smith and Sutherland (1999) found that most injury to stems in *Quercus* sp. from two prescribed fires resulted from heat conducted through bark, and not flaming combustion. Heat transfer properties govern resistance to fire-caused injury (Ryan 1998). The single most important biological factor determining resistance to fire injury to stems is bark thickness (Hare 1965, Jackson and others 1999, Ryan 1998, Ryan and Reinhardt 1988, Vines 1968). Bark serves as an insulator that slows heat

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[1] Project Leader, USDA Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory, PO Box 8089, Missoula, MT 59807

[2] Plant Physiologist, USDA Forest Service, Northeastern Research Station, Forestry Sciences Laboratory, 271 Mast Rd., Durham, NH 03824.

conduction to the vascular cambium (Spalt and Reifsynder 1962). The thicker the bark, the longer the heating period required to injure underlying cambium (Uhl and Kaufman 1990, Vines 1968).

In general, bark thickens as trees grow in diameter, and resistance to fire-caused wounding increases (Harmon 1984, Hengst and Dawson 1994, Pausas 1997, Spalt and Reifsynder 1962). Bark thickness is dependent on diameter, not age (Hengst and Dawson 1994); suppressed trees the same age as dominant trees have relatively thinner bark and hence greater susceptibility to fire injury (Harmon 1984). Consequently, ability to resist fire injury has been attributed to stem diameter and to variability among tree species (Hare 1965, Hengst and Dawson 1994, Pausas 1997, Spalt and Reifsynder 1962).

Some tree species fundamentally have thicker bark than others, and as such have greater resistance to fire injury. Because texture is so variable thickness is difficult to quantify, but general patterns among species have been noted (e.g., Harmon 1984, Hengst and Dawson 1994). For example, upland species generally have thicker bark than bottomland species, perhaps because fire is typically more frequent at upland sites compared to bottomland sites (Hengst and Dawson 1994, Jackson and others 1999). Comparisons of data presented by Harmon (1984), Hengst and Dawson (1994), Jackson and others (1999), and Spalt and Reifsynder (1962) reveal certain patterns. As a group, white oaks (*Quercus* subgenus *Lepidobalanus*) have the thickest barks of the central hardwoods species, followed by the red oaks (*Quercus* subgenus *Erythrobalanus*). Examples of thinner barked species include American beech (*Fagus grandifolia*), maple species (*Acer*), hickory species (*Carya*), flowering dogwood (*Cornus florida*), and black cherry (*Prunus serotina*).

Additionally, the rate of bark thickening on the lower stem varies among species and primarily determines resistance to heat-caused injury; the sooner bark thickens at the base, the earlier in life the stem gains resistance (Hare 1965, Harmon 1984, Hengst and Dawson 1994). For example Hengst and Dawson (1994) showed that tulip poplar (*Liriodendron tulipifera*) has very thin bark when small but has a very rapid rate of bark thickening. Thus, seedling and sapling-sized tulip poplar trees are susceptible to topkilling by fire (Barnes and Van Lear 1998, Brose and Van Lear 1998), but tree-sized tulip poplar has been described as particularly resistant to fire-caused wounding (Gustafson 1946, McCarthy 1933, Nelson and others 1933). By contrast, silver maple (*Acer saccharinum)* also has very thin bark when small but a slow rate of bark thickening, which results in thin bark throughout its life (Hengst and Dawson 1994); silver maple is highly susceptible to fire injury (Auclair and others 1973).

The distribution of bark along and around the stem is also variable among species. Bark texture can vary from smooth to fissured and can vary significantly within as well as among species (Howard 1977). For trees with deeply fissured bark, such as chestnut oak (*Q. prinus*), the cambium directly beneath the fissures is more susceptible to injury, and multiple, discontinuous fire scars may occur around the stem (Smith and Sutherland 1999, Stickel and Marco 1936). Factors that contribute to variability in bark texture include tree vigor and growth rate, age, and height on the tree (Howard 1977). In regimes characterized by low-intensity surface fires, thick bark at tree bases provides protection from heating and a competitive advantage over trees with thin bark at the base, but since fires typically have low flames heights, bark rapidly thins with increasing height (Harmon 1984, Jackson and others 1999). For example, bur oak (*Quercus macrocarpa*) has relatively thick bark near the ground (but thinning with height); it grows in oak savannas that are maintained by frequent fire. By contrast, water oak (*Quercus nigra*) has relatively thin bark near the ground, and typically grows in bottomland forests where fire rarely occurs (Jackson and others 1999).

Heat conduction properties of bark tissue play a role in resistance to heating and injury of the vascular cambium during fires, and these properties vary among species (Hare 1965, Harmon 1984, Hengst and Dawson 1994). For example, for a given bark thickness, red maple (*Acer rubrum*) conducts heat relatively quickly (Hare 1965). Heat conduction properties have been expressed as thermal conductivity (the ability of a material to transfer heat), specific heat (the ability to absorb heat), and thermal diffusivity (ratio of thermal conductivity to the product of specific heat and bark density) (Martin 1963, Harmon 1984). The ratio of outer bark (rhytidome) to inner bark (phloem) increases with tree diameter in many species, and this factor affects heat conduction properties: rhytidome conducts heat more slowly (Hare 1965). Thus, trees with substantial rhytidome such as chestnut oak (*Quercus prinus*) are more heat-resistant than trees with little rhytidome such as American beech (*Fagus grandifolia*) (Stickel 1936). However, bark thickness alone remains the simplest, best predictor of heat conduction (Vines 1968).

How Trees Respond to Fire Injury

Tree survival after mechanical injury, including wounding caused by fire, depends on the internal boundary-setting process known as compartmentalization (Shigo 1984). Compartmentalization in the wood of living trees resists the spread of decay and loss of normal wood function, and minimizes the extent of injury. Tree species vary in ability to compartmentalize injury (Shigo 1984, Shortle and others 1996) and hence in resilience to fire-caused injury. For example, trees in the white oak group (*Quercus* subgenus *Lepidobalanus)* are unusually effective at compartmentalizing decay, which places them at a competitive advantage over other injured trees (Abrams 1996). Understanding the role of compartmentalization effectiveness following fire may be critical to understanding patterns of tree survival and changes in wood quality following fire.

Compartmentalization is a set of processes that integrate tree anatomy, physics, and inducible changes in tree physiology. These processes may be divided into those that occur in wood present at the time of wounding and those that occur in wood formed after wounding. Tree survival depends on maintaining a healthy vascular cambium, the thin layer of generative cells located beneath the bark. As the cells of the vascular cambium divide, wood (xylem) is formed to the inside of the cambium and inner bark (phloem) is formed to the outside. When part of the vascular cambium and associated tissues are overheated and killed by fire, the first changes are physical and immediate: water columns (normally held under tension in functional wood vessels) snap which is dangerous for the tree because drying kills living sapwood cells and provides a favorable environment for the spread of microbial infection. Because of cellular architecture and inducible changes in wood physiology, these abrupt changes in water tension induce the immediate formation of plugs and the eventual formation of chemical boundaries that resist both wood drying and the spread of microorganisms that lead to the breakdown of wood. Some of the physiological changes involve the oxidation and polymerization of phenolic compounds that then discolor or stain the wood (Smith 1997).

After wounding, probably in response to changes in plant growth regulators (Smith and Shortle 1990), the surviving vascular cambium produces an anatomically distinct barrier zone (Shigo 1984) that can appear as a “false ring” in the wood. This barrier is most effective in resisting the outward loss of wood function and the spread of microorganisms. Effective barrier zones limit wood discoloration and decay to wood present at the time of injury. Compartmentalization allows for the continued survival of the vascular cambium and the generation of new wood. These boundaries, although frequently effective at resisting the spread of infection, can and do fail. The effectiveness at compartmentalization depends on the severity of the wound, the tree species involved, and on the individual tree.

For years following the injury, conspicuously wide growth rings may be produced at the wound margins, apparently to both speed closure of the wound and to provide additional mechanical support for the wounded stem. Wound closure, although apparently not essential for tree survival, aids tree functioning by restoring the continuity of the vascular cambium and wood formation around the tree perimeter. Closure also seals off the wound and infected wood from the outside atmosphere, reducing the partial pressure of oxygen and increasing the partial pressure of carbon dioxide. These changes in the internal atmosphere tend to reduce the rate of the anaerobic wood decay process. Research on northern hardwoods indicates compartmentalization is more effective in sugar and red maple and red oak than in paper and yellow birch, following equivalent levels of storm injury (Smith, unpublished data).

Characteristics of Fire Scars

In earlier work (Smith and Sutherland 1999), we characterized fire scars in small, dissected oaks in southern Ohio that had been exposed twice to prescribed fire. We learned that: 1) the prescribed low-intensity fires wounded some trees, but because heat was unevenly distributed, not all trees were scorched, and not all scorched trees were wounded; 2) because flame heights were low, most wounds were small and low to the ground (about 1 foot in height above groundline); 3) wounds resulted in typical defense processes against infection, including compartmentalization and closure; and 4) nearly all wounds were caused by heating through the bark, which often persisted over the heat-killed tissue after wounding, sometimes for years, thus, small wounds that rapidly compartmentalize may only be apparent when the tree is felled and dissected.

Implications

The resistance of any hardwood tree stem to fire-caused wounding is primarily determined by bark thickness. Generally, as stems increase in diameter, bark thickness concomitantly increases along with resistance to fire-caused wounding. Additionally, there are species-specific differences in bark thickness. Thus, there is a gradient of susceptibility: small, thin-barked trees are most susceptible to wounding and large, thick-barked trees are least susceptible.

Many researchers have suggested that fire is an important process in promoting oak dominance in upland oak types, for example Abrams (1992) and Lorimer (1985, 1993). Differences among species in resistance to wounding by fire and the effectiveness of wound response (resilience to wounding) is probably an important determinant of species composition and dominance in forests exposed to fire. Jackson and others (1999) provided evidence for this idea. They pointed out that for any tree, the greatest improvement in survival probability occurs at sapling sizes, and then demonstrated that the rate of overall growth and bark thickening is more rapid in oak species typical of fire-prone habitats than those associated with moist habitats (e.g., bottomlands) that burn infrequently. They showed that resistance is higher in species typical of fire-prone habitats, and suggested that other defenses would probably be quantifiable. We believe that resilience to injury (effective compartmentalization) is one of those defenses, and would likely follow similar patterns of greater resilience in species typical to fire-prone habitats compared to those where fire is rare.

Conclusions

Any fire, whether wild or prescribed, can injure a tree, but not all trees are wounded by fire. Fires in hardwood forests are typically low in intensity and patchy in distribution. Even when scorched, not all trees are wounded. Bark is an insulator from heat and if scorched, the underlying cambium may not be damaged. Most wounds, when they occur, are near the ground, small in size, and rapidly compartmentalized: few are externally visible. In addition, not all wounds are caused by fire: other events also mechanically wound trees, including impacts from treefalls (e.g., windthrows) and management activities, including logging. Because damage is relative to management objectives, not all injuries result in damage. In the development of fire prescriptions and in implementing fires, managers have a degree of control over potential for fire-caused injury based on their desired outcome.

Better understanding of species-specific resistance and resilience to fire injury are essential to improving predictions about the effect of prescribed fire on surviving trees. To this end, the authors are currently conducting research on the characteristics of prescribed fire-caused wounds, resistance to injury, and comparative effectiveness of compartmentalization in central hardwood species.

Acknowledgments

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Managing Rare Plant Populations With Fire in Great Smoky Mountains National Park

Janet Rock[1]

Abstract

Previous to 1996, wildland fires were effectively excluded from Great Smoky Mountains National Park (GSMNP) for over 60 years. The National Park Service now recognizes the importance of fire as a natural disturbance and considers fire a critical disturbance factor in GSMNP. A Management Ignited Prescribed Fire Program, part of the Park's Fire Management Plan, was created to address some resource management objectives. Prescribed fire is now being used as a tool to enhance natural resources such as rare plant populations. Four rare plant species have been managed by prescribed fire since 1997 and all populations have responded with increased plant numbers and vigor. Based upon our success, two additional species are scheduled for management by prescribed fire in 2000.

Introduction

Fire has been excluded from Great Smoky Mountains National Park (GSMNP) since the Park's establishment (1934). An approved Fire Management Plan (1996), and the institution of a Fire Management Program, has allowed GSMNP to restore an important natural disturbance to various natural communities as well as individual plant populations. A Rare Plant Program was initiated in 1989 to monitor rare plant populations and, if these species were observed to be declining, to manage these populations in order to stabilize or reverse the trend. Currently, the following four rare plant species are being managed with prescribed fire:

- Daisy-leaved moonwort (*Botrychium matricariifolium* A. Braun)
- Bittercress (*Cardamine flagellifera* O. E. Schulz)
- Goldenseal (*Hydrastis canandensis* L.)
- Indian grass (*Sorgastrum nutans* (L.) Nash)

Southern Appalachian cove forests are typically affected by small-scale, mild disturbances such as tree falls, and many rare plant species require some level of disturbance in order to survive (Denslow 1987). Through our monitoring efforts, we determined that two species – Daisy-leaved moonwort and Bittercress – appeared to require some leaf litter disturbance to stimulate germination and thrive. We chose prescribed fire to create disturbance to the leaf litter although fires were relatively infrequent within these mountains in general, and north-facing lower slopes have the lowest incidence of fire (Harmon 1982). Three populations (Daisy-leaved moonwort, Bittercress, and Goldenseal) are located on north-facing slopes (elevation range 381-686 meters) in mixed mesophytic hardwoods. In addition, GSMNP's single population of Goldenseal was infected with a native fungus, *Streptobotrys streptothrix,* and fire was used to reduce the sclerotia of the fungus which overwinter in leaf litter. The fourth species, Indian grass, has been negatively affected by annual mowing and competition from non-native Fescue (*Festuca pratensis*). Prescribed fire was used to stimulate flowering in the native grass.

Methods

Daisy-leaved moonwort

Two populations of Daisy-leaved moonwort (*Botrychium matricariifolium* A. Braun), a small, primitive fern (3-19 cm in height), are known to occur in GSMNP. This species is listed as significantly rare in North Carolina and as a species of special concern in Tennessee. One population is comprised of 42 plants and the other approximately 114 plants, or fewer, depending on the year.

Their habitat is second-growth tulip poplar (*Lirodendron tulipifera*), red maple (*Acer rubrum*), and Eastern hemlock (*Tsuga canadensis*). We chose the larger population for management. A trail bisects the population creating a light gap. Disturbance to the leaf litter layer periodically occurs from episodes of heavy rainfall, wild boar rooting (usually in the winter), and hikers. The population is north-facing with a slope of 5-50 percent (elevation 686 meters).

Two permanent transects (21 meters and 10 meters) were established in the study population in 1991, and plants occurring within three meters of each transect were mapped. Since 1991, plants were most abundant in locations devoid of leaf litter and duff. Due to a severe decline in plant numbers in 1997 (67 percent), fire and raking were selected as management tools. Mapped plants were used as a guide to establish three 7 meter x 10 meter treatment plots to which we applied three different management techniques: (1) fire to decrease leaf litter and duff, (2) raking, and (3) no treatment in March 1998.

Bittercress

Bittercress (*Cardamine flagellifera* O. E. Schulz), a stoloniferous perennial, is listed as state threatened in Tennessee. The Park has three known populations and two are actively monitored due to potential threats (e.g. wild boar rooting and deer browsing). Baseline data were collected at one population in 1989 and censused in 1991.

The study population is on a north-facing slope in second-growth mesic hardwoods comprised of tulip poplar, red oak (*Quercus rubra*) and white oak (*Quercus alba*) (elevation 610 meters). Due to a decrease in rosettes observed in April 1997 (16 percent), a portion of the population was burned in September 1997. The study population was re-censused in April 1998.

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[1] Botanist, Great Smoky Mountains National Park, 107 Park Headquarters Road, Gatlinburg, TN 37738

Goldenseal

Goldenseal (*Hydrastis canadensis* L.) is a highly valued perennial herb with thick yellow rhizomes. This plant is listed in the Convention on International Trade of Endangered Species (CITES) Appendix II (USFWS/OSA 1997), and is state-listed as endangered in North Carolina and threatened in Tennessee due to over-collection and habitat loss. The Park has one population comprised of two colonies (subpopulations 1 and 2). Goldenseal prefers rich moist woods, and in GSMNP it is found growing in mesic second-growth woods comprised of sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), red oak (*Quercus rubra*), and Eastern hemlock on a north-facing slope of 50 percent (elevation is 381 meters).

Data has been collected each year since 1989. In 1991 a native fungal disease, identified as *Streptobotrys streptothrix*, was observed on the stems and leaves throughout the population (Alan Windham, University of Tennessee Agricultural Extension Agency). Subpopulation 1 showed the highest incidence of infection, and by June 1991 it was observed that the total number of stems had decreased by 30 percent in just one year.

Based on Windham's recommendation, in September 1991 leaves and stems were clipped, bagged, and removed from the site in both subpopulations. Leaf litter directly under the stems was bagged and removed from the site as well. In addition, in March 1998, a low-intensity fire was set at subpopulation 1 in order to consume sclerotia of the fungus overwintering in the leaf litter layer.

Indian grass

Indian grass (*Sorgastrum nutans* (L.) Nash) is a warm-season grass. Increase in Indian grass stems after burning is well-documented (Dix and Butler 1954; Robocker and Miller 1955; Kucera and Ehrenreich 1962; Hadley and Kieckhefer 1963; Anderson and others 1970). Scattered populations occur on the western end of the Park, particularly in open fields in Cades Cove. It has been suppressed as a result of haying operations since the 1940s, and is out-competed by a non-native cool-season grass, Fescue (*Festuca pratensis*). To stimulate flowering (Kucera and Ehrenreich 1962) and encourage the spread of Indian grass into larger areas, we experimented with a late-fall burn in 1997. Three 50 meter x 3 meter belt transects were established to facilitate counting fertile stems before and after the treatment.

Results and Discussion

Daisy-leaved moonwort

The population responded favorably to both treatments, prescribed fire and raking (23 plants in 1997 to 114 plants in 1999) (Figs. 1-4). The fire treatment showed a 35 percent increase in plants and the raking treatment showed a 33 percent increase. The control has shown no increase.

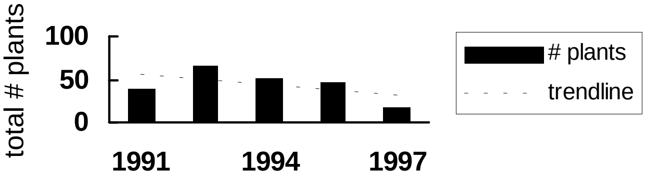

Figure 1.—Number of Daisy-leaved moonworts (*Botrychium matricariifolium*) before the 1998 spring fire.

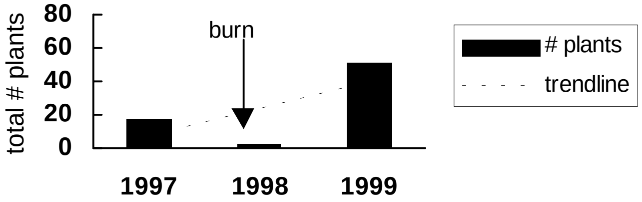

Figure 2.—Number of Daisy-leaved moonworts (*Botrychium matricariifolium*) after the 1998 spring fire.

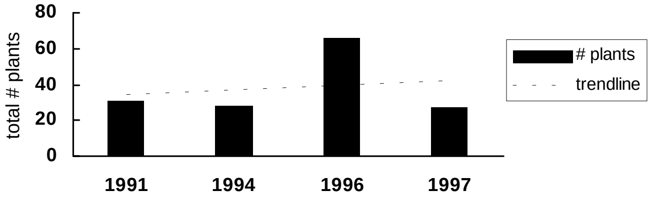

Figure 3.—Number of Daisy-leaved moonworts (*Botrychium matricariifolium*) before 1998 spring raking.

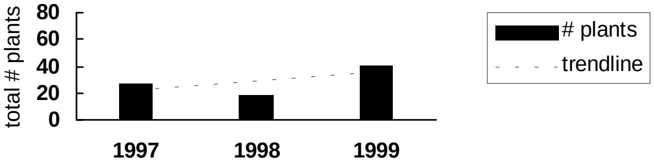

Figure 4.—Number of Daisy-leaved moonworts (*Botrychium matricariifolium*) after 1998 spring raking.

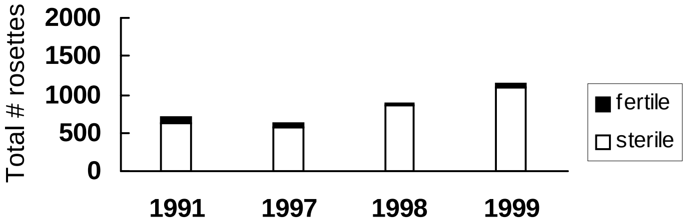

Figure 5.—Number of fertile and stertile Bittercress (*Cardamine flagellifera*) rosettes 1991-1999 in burned portions of the population.

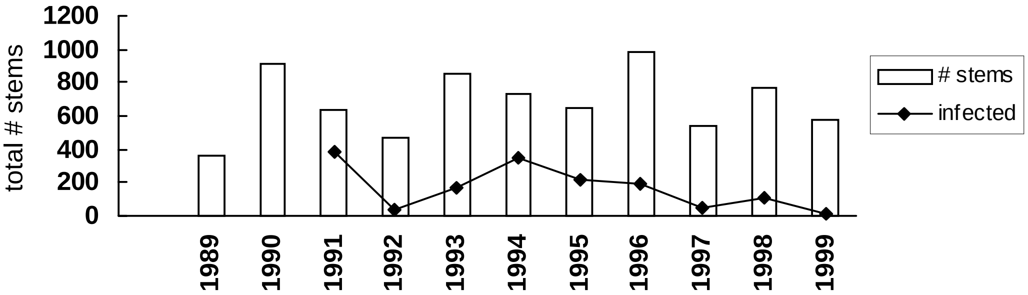

Figure 6.—Number of Goldenseal (*Hydrastis canadensis*) stems in subpopulation 1 from 1989-1999 with number of infected stems.

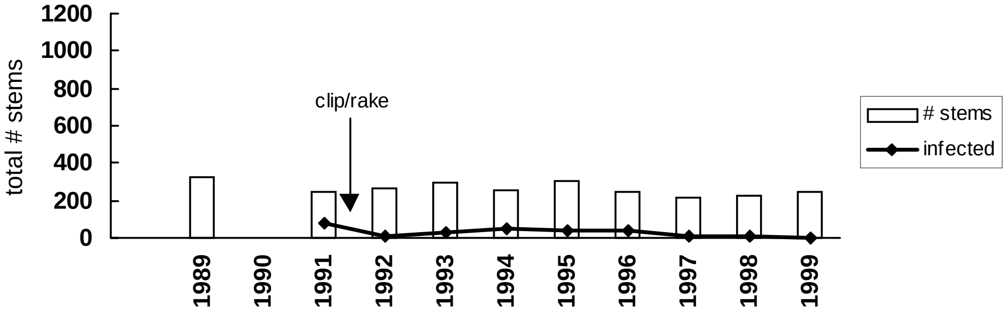

Figure 7.—Number of Goldenseal (*Hydrastis canadensis*) stems in subpopulation 2 from 1989-1999 with number of infected stems.

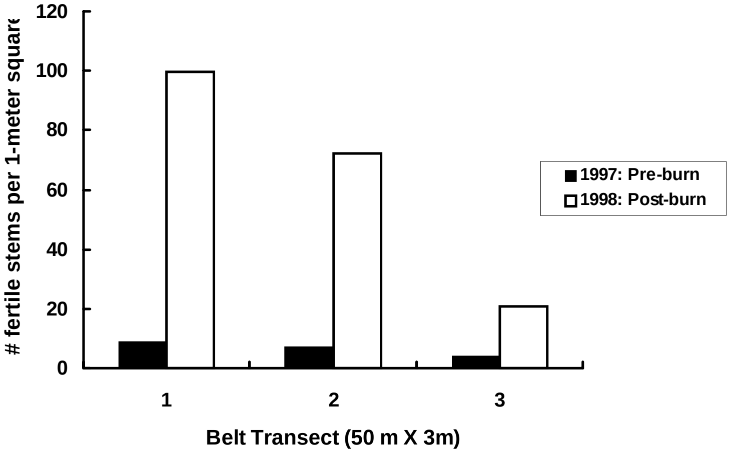

Figure 8.—Average number of fertile stems per 1-meter square in Indian grass population (*Sorgastrum nutans*) before fire (1997) and after fire (1998).

Bittercress

A 27 percent increase in rosettes was observed from 1998 to 1999, all within the burned portion of the study site. In addition, fertile rosettes showed a twofold increase (118 percent) after prescribed fire (Fig. 5).

Goldenseal

After our first treatment (clipping, etc.), a large reduction in diseased stems was documented in 1992 (90 percent). Subsequently, diseased stems in subpopulation 1 more than doubled (110 percent increase) from 1993 to 1994 (two years after our first treatment) (Fig. 6). However, the incidence of disease again decreased in 1995 without intervention. An additional 93 percent decrease in infected stems in this subpopulation was observed from 1998 to 1999 following our second treatment (prescribed fire) in March 1998 (Fig. 7). Data indicates both treatments (clip and rake, or burn) are effective management tools.

Indian Grass

Indian grass responded favorably to prescribed fire. Average number of fertile stems per 1-meter square increased in all three belt transects after treatment (Fig. 8). Transect 1 had the highest increase in average number of fertile stems per 1-meter square (8.94 in 1997 to 99.73 in 1998). Transects 2 and 3 had an average of 6.92 and 4.08 fertile stems in 1997, respectively. In 1998, the average fertile stems per 1-meter square increased to 72.05 and 20.9 in transects 2 and 3, respectively.

Conclusions

Although fires were historically uncommon in southern Appalachian cove forests (Harmon 1982), prescribed fire has been an effective management tool for some rare plant populations in GSMNP. Dormant-season burning to reduce leaf litter buildup was successful. Daisy-leaved moonwort and Bittercress populations increased after fire was experimentally introduced. Prescribed fire also reduced the incidence of a fungal infection in Goldenseal.

Raking was an effective tool at the Daisy-leaved moonwort and Goldenseal populations as well. Managing selected rare plant populations with prescribed fire and/or raking will continue to be a part of our Rare Plant Program.

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In Summary: Fire in the Evolution of the Eastern Landscape - A Timeline

Ed Buckner[1]

Over time fire has played many roles in the modification of the earth's landscapes. In the beginning there was only "natural" fire, but it came from several sources (i.e., volcanoes, spontaneous combustion, lightning, etc.). Then came a new source of fire that has drastically altered the Earth's landscapes - cultural fire. Some 100,000 years ago, as modern man evolved from earlier homo species, there was a seminal event in mans "rise to civilization" - it was his controlled use of fire. It is now recognized that this accomplishment was a primary factor that enabled Neanderthal man to move from Africa into the colder reaches of central Europe. Much later (e 12,000 years ago) this same tool enabled modern man to cross Berengia (the land bridge between Siberia and Alaska that was exposed as continental glaciation lowered ocean levels by capturing much of the Earth's water) and populate the Americas. The original people of the Americas were modern human beings whose arrival brought the first taste of "un-naturalness" to our landscape.

There is strong public sentiment in the East today to establish (restore??) "natural" forest conditions. Accepting the environmentalists stance that "natural" means without a human (cultural) influence, restoring this condition is an impossibility as it has never before existed with the current complement of trees. The deciduous forest moved northward into the Eastern landscape between five and eight thousand years ago as the Wisconsin Ice event ended. However, "man" had arrived several millennia earlier when boreal forests (spruce, larch, jack pine) extended as far south as Atlanta. His activities (especially setting fires) possibility aided in the northward migration of trees. Thus our present forest cover evolved in a culturally altered environment and many trees exhibit traits that adapt them to cultural modifications of the environment (e.g., cones that do not open until burned).

If it could be argued that prior to 1492 Native Americans lived "in harmony" with their environment and no effect on it, then the existence of a "natural" environment at that time could be valid. However, the intent of this presentation is to show that this was not the case; that throughout the Holocene (last 10,000 years) mans' activities have resulted in marked modifications in the Eastern landscape to the extent that, then or now, few places can claim to display "natural" conditions.

There were four preColumbian cultures that occupied the Eastern landscape prior to 1492. The original settlers were Paleo-Indian (10,000-8,000 BC), hunter-gathers who used fire to drive and trap the megafauna (wooly mammoth, mastodons, etc.) that served as a primary food source. These fires also maintained the open, grassy conditions essential for these animals (closed forests do not support grass). The following period -the Archaic (8,000-1,000 BC) saw the beginnings of agriculture with fire essential to control the aggressive native vegetation that threatened to close over their fields. Recent research indicates that landscape-scale fires of cultural origin were occurring in the Southern Appalachians during this Period. Next came the Woodland period (1,000 BC-800 AD) in which agriculture was refined and fire was widely used to open the landscape for hunting, travel and for protection. The Mississippian Period (800-cl, 600 AD) brought PreColumbian time to a close with Native American fire practices passed on to early white settlers.

Ecologists now know enough about forest dynamics and tree biology to model a close approximation of the "natural" forest condition for Eastern forests. For our present complement of trees, the "natural" forest condition would be a closed, multi-canopied, forest composed of shade tolerant hardwoods. The only conifer in the mix would be hemlock on moist, cool sites. The highly productive "pioneer" species characteristic of earlier stages of forest succession (the change in species composition over time, from light-demanding "pioneer" trees to shade tolerant "climax" species) would occur only as widely scattered individuals or in the rare areas where natural disturbances opened the forest. The characteristic trees of today's landscape (yellow pines, yellow-poplar, oaks, sweetgum, ash, hickory, etc.) that represent early- to mid-successional stages, would be largely absent. Examples of the final stage of forest succession (climax forest) may actually exist on a few sites that have not been disturbed for several centuries thus allowing succession to run its full course.

Shifts toward this condition are well underway on public lands in the East. Sites that once supported pure yellow pine or pine-hardwood stands are now converting to hardwoods, and oaks are being replaced by red maple and black gum. These shifts are due largely to 70+ years of highly effective fire protection. The recent reductions in harvesting and additions to wilderness (legislating protection from disturbance in perpetuity) will further erode the productivity of Eastern forests for both wildlife and forest products. Accepting that this trend will continue, the widespread use of prescribed fire offers the only hope for establishing (maintaining) the early successional stages that contribute significantly to the health and productivity of Eastern forests.

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[1] University of Tennessee (retired), Knoxville, TN 37919, ph: 423-524-8329, email: ebuckner@utk.edu

Poster Abstracts

Oak Seedling Response to Fire and Herbivory

A.S. Adams and L.K. Rieske
University of Kentucky, Department of Entomology, Lexington, KY 40546

We manipulated arthropod and mammalian herbivory levels on white oak seedlings using a combination of insecticide applications and fencing, and employed a split-split-plot design to assess the impact of single- and multiple-year burns on seedling growth over a two year period. Herbivory levels increased over time on all sites, but there was no significant difference in herbivore pressure on seedlings in single-year, multiple-year, and non-burned plots. Insecticide treated seedlings (+I) suffered less herbivore pressure than did non-insecticide treated seedlings (I)($P<0.0001$), and mammalian herbivory was significantly reduced by the presence of a fence (+F)($P<0.01$). Seedling relative height growth, shoot elongation, diameter growth, and specific leaf mass were greatest on once-burned sites, intermediate on twice-burned sites, and least on non-burned sites. Bud expansion was the only seedling performance parameter unaffected by burn treatment. We found differences in herbivory levels due to insecticide application, suggesting that the arthropod component was the more significant aspect of overall herbivory levels, but this herbivory did not impact seedling performance, and seedling performance parameters were more closely linked to mammalian herbivory. These results suggest that prescribed fire may be a practical component of a management program designed to enhance oak regeneration.

Prescribed Burning: Effects on Forest Bird Populations

Vanessa L. Artman
Department of Evolution, Ecology, and Organismal Biology, 1735 Neil Avenue,
Ohio State University, Columbus, OH 43210

Prescribed burning is being applied on an experimental basis in southern Ohio to restore and maintain oak-dominated forests. Effects of prescribed burning on bird communities have been given little consideration in previous research, but are of concern because long-term declines have been observed for many bird species within the region. My research addresses short-term changes in population levels and nesting productivity of forest bird species. Bird species diversity did not change as a result of burning, but population levels of three species, the hooded warbler, ovenbird, and worm-eating warbler, declined in burned areas. As expected from the reduced population levels, fewer active nests were found in burned areas than in unburned areas, but there were no differences in rates of nesting success between treatments. Predation of nest contents was the primary factor influencing the likelihood of nesting success. A wide variety of nest predators inhabit the forests, including snakes, squirrels, raccoons, and other birds, and their foraging tactics are so variable that nest predation events tend to be unpredictable. Prescribed burning did not appear to affect the array of nest predators in the community or the susceptibility or exposure of eggs or nestlings to these predators.

Ecological Effects of Fire on *Xerophyllum asphodeloides,* a Rare Appalachian Lily

Norman A. Bourg and Douglas E. Gill
Department of Biology, University of Maryland, College Park, MD 20742

Fire’s role as a biotic disturbance agent in Appalachian forests has been historically understudied. We performed a fire/canopy removal field experiment to determine the impact of these factors on survival and reproduction of turkeybeard (*Xerophyllum asphodeloides),* a rare lily associated with Appalachian mixed pine-oak forests. Turkeybeard is endangered or rare in portions of its range and is in the U.S. Center for Plant Conservation’s National Endangered Plant Collection. It is also thought to be fire-dependent; yet, there has been no experimentally validated ecological linkage to fire. An immense flowering response to our treatments occurred in the second and third post-manipulation years. Mixed-model ANCOVA analyses yielded a significant positive effect of fire on both the numbers of fertilized flowers and seeds. A significant positive effect of canopy removal on these variables was also found. Total number of flowers per inflorescence did not differ between control and treated plants. Inflorescences from another nearby undisturbed, shaded population showed virtually no fertilization or seed production. Many plants exhibited a cost of reproduction by partial vegetative dieback, but little complete mortality occurred. This suggests that fire is important for turkeybeard conservation, and also has implications for the study of disturbance regimes in Appalachian forests.

A Shelterwood-burn Technique for Regenerating Productive Upland Oak Sites

Patrick Brose[1] and David Van Lear[2]
[1]Research Forester, USDA Forest Service, Northeastern Research Station, Forestry Sciences Laboratory, Irvine, PA 16329, [2]Bowen Professor of Forestry, Clemson University, Department of Forest Resources, Clemson, SC 29634

Regenerating oak stands on productive upland sites is widely recognized by foresters as a major problem in hardwood management. Recent research indicates that oak regeneration is a more resilient sprouter following surface fires than its primary competitors on these sites if burning occurs 3 to 5 years after a partial overstory harvest. This combination of cutting followed by fire (the shelterwood - burn technique) mimics natural disturbances that have occurred in eastern North America for millennia and appears to be a viable approach to regenerating oaks on productive upland sites. This poster presents silvicultural guidelines for applying the shelterwood - burn technique on appropriate sites and discusses its benefits for private landowners and resource professionals.

Effects of Fire, Deer Herbivory, Seed Predation, and Canopy Gaps on Forest Regeneration: A Large-scale Experimental Approach

Rachel J. Collins and Walter P. Carson
Department of Biological Sciences, University of Pittsburgh, Pittsburgh, PA

Oaks have been a dominant group and readily regenerating in eastern deciduous forests for thousands of years. Interestingly, since the 1960s, oaks have failed to regenerate in many areas. This pattern is one of a number of symptoms characteristic of recent changes in forest understories (i.e. the decline in diversity of wildflowers and tree seedlings; and the increase in maple abundance). We are currently investigating the underlying mechanisms that control tree species regeneration in eastern forests using a large-scale experimental approach to test four prevalent hypotheses. We have experimentally manipulated fire, deer, seed predation and canopy gaps in 128, 400m^2 research plots in West Virginia and are monitoring the survival, growth and recruitment of the 28,000 seedling, sapling and canopy trees within these research plots. The fires were intense and topkilled up to 100% of small saplings and up to 60% of large saplings; however canopy tree mortality was only 4% and not different from controls. Fires strongly influenced the species composition of emergent seedlings. Fire by deer, and gap by deer interactions are emerging as important factors affecting growth and survival of seedlings. This study is just beginning to elucidate the underlying mechanisms that drive forest regeneration.

Effects of Shelterwood Harvesting and Prescribed Fire on Regeneration Structure

Wade Dorsey[1] and Mary Ann Fajvan[2]
[1]Doctoral Candidate, [2]Associate Professor, West Virginia University, Division of Forestry, Morgantown, WV 26506

In 1993, four, 10 acre Appalachian hardwood stands received a shelterwood seed cut. Preharvest overstory species consisted primarily of yellow poplar (40-50% of basal area) and oaks (30% of basal area); understory composition was dominated by red maple. Post-harvest basal areas averaged 70ft^2/a with oaks comprising the majority of the seed trees. After 6 growing seasons, black cherry, red maple and sassafras seedlings dominated the regeneration. Because regeneration composition did not reflect the pre-harvest overstory, a prescribed fire was conducted in November 1999 primarily to reduce the density of black cherry and red maple and to promote oak species. The effects of fire on herbaceous vegetation and soil nutrients are also being examined.

In the summer of 1999, the 10-acre treatment blocks were divided into two, 5-acre treatments: burned and unburned. In each 5-acre block, nine, milacre sampling plots were established and all woody and herbaceous vegetation <1" dbh was counted and height recorded. An additional milacre was established to measure only tree seedlings. All plots will be remeasured in the spring and summer of 2000 and 2001.

Soil samples were collected at 3 locations, adjacent to six of the vegetation sample plots, immediately prior to and after burning. Soil samples will be collected two additional times in spring and summer 2000. Litter and mineral soil samples will be analyzed for total N. P. K, Mg, Ca, and S. Mineral soil samples will also be analyzed for exchangeable or available N. P. K, Mg, Ca, and S. CEC, % BS, bulk density and pH.

Prescribed Fire Effects on Foliar Nutrients, Photosynthesis, and Growth of Ridgetop Maple and Oak Seedlings

N.L. Gilbert, M.A. Arthur, S.L. Johnson, S.K. Gleeson and B.A. Blankenship
University of Kentucky, Lexington, KY

Forest management with low-intensity prescribed burns aims to promote oak regeneration in upland forests of the Cumberland Plateau, where sixty years of fire suppression coincide with increasing red maple abundance. Our objective was to determine whether a single fire influences the ecophysiology of red maple, chestnut oak, and scarlet oak seedlings during the first year postburn. On a burned (March 1997) and an unburned ridgetop site, we measured foliar nutrients (N. P. K, Mg, and Ca) monthly during the 1997 growing season, photosynthesis at five light intensities (0, 100, 400, 1000, 1500 uEm-2s-1) in July, 1997, and stem diameter and height in May, 1997, and March and May, 1998. For all species, foliar N. P and K, net photosynthetic rates, and diameter and height relative growth rates were significantly higher on the burned site than the unburned site. Fire had little effect on the relative ecophysiology among species on the burned site compared to the unburned site. These data suggest that single low-intensity prescribed burns will not benefit the regeneration of oaks by enhancing their ecophysiological performance over that of red maple.

Chilton Creek: A Case Study of Landscape Fire Effects

George W. Hartman[1] and Blane Heumann[2]
[1]Fire Ecologist, Missouri Department of Conservation, 1110 S. College Ave., Columbia, MO 65201, [2]Lower Ozarks Preserves Manager, The Nature Conservancy, Missouri Ozarks Office, P.O. Box 960, Van Buren, MO 63965

The Chilton Creek Management Area is 5,657 acres located in Carter County, Missouri on the Ozark Plateau. Owned and managed by The Nature Conservancy, a 2,500-acre portion is divided into 5 burn units with a fire return interval for the various units from annual to 4 years The major objective of the studies is to document the effects of landscape-scale prescribed burning on the naturally occurring communities of the Ozark woodlands. Fuel loading, fire behavior, fire intensity and coverage measurements are also being recorded to relate fire activity to resource response. Prescribed burning was initiated spring, 1998.

The study to document fire effects on the herbaceous and woody components is being conducted cooperatively with the Missouri Department of Conservation. Nested plots (250) and sampling protocols mimic an adjacent long term study of timber harvesting techniques called the Missouri Ozark Forest Ecosystem Project (MOFEP). Prescribed fire management will be compared to even aged timber management, uneven aged management and controls (no active management)

Other studies being conducted with the restoration burning include stream hydrology and biota, differential mortality of young woody species, breeding bird populations, herpefauna, lichens, invertebrates including land snails and Orthoptera, soils, geology, and ecological land classifications.

Ethnoecology of Fire: An Experimental Approach in the Ohio Valley

Erin B. Hines[1], Jan Salick[1], Elaine Kennedy Sutherland[2], Todd F. Hutchinson[3] and Steve Sutherland[4]
[1]Department of Environmental and Plant Biology, Ohio University, Athens, OH 45701, [2]USDA Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory, PO Box 8089, Missoula, MT 59807, [3]USDA Forest Service, Northeasterm Research Station, 359 Main Road, Delaware, OH 43015, [4]USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, P.O. Box 8089, Missoula, MT 59807

Native Americans used fire to manipulate nature and directly benefit their survival. Certain plant species, many of which were useful to Native Americans as sources of food, fiber, dye, medicine, and game browse, are adapted to survive and even thrive in post-burn environments. Evidence suggests that Native Americans intentionally set fires to encourage growth and survival of such useful species. Data from a 5-year study conducted by the U.S. Forest Service in Southeastern Ohio to track the long-term effects of prescribed fire on forest composition were used to track changes in useful plant abundance under certain fire regimes. Preliminary analyses of pre-burn data establish covariates between burn treatments and moisture indices. Split-plot block ANOVAs of the 1999 post-burn data suggest an increase in many fern and graminoid species, along with an increase in *Vaccinium* and early-successional tree species. The results from this study have implications for modern forest management and global conservation. Fire increases diversity in both botanical and zoological systems by initiating patches of early succession in plant communities, thus increasing resources available for ecosystem interactions and possible human use. By encouraging the growth of both timber and non-timber forest products, resource managers can move towards more multiple-use forest management.

Managing Red-Cockaded Woodpeckers *(Picoides borealis)* Affects Breeding Bird Communities of Pine-Oak Forests in Southeastern Kentucky

Martina Hines
Kentucky State Nature Preserves commission

Breeding bird communities were surveyed on the Daniel Boone National Forest, Kentucky, using point counts, in mid May, late May and early June 1994 and 1995. Twenty-eight stands of Appalachian pine-oak forest were surveyed, of which 14 were managed for red-cockaded woodpeckers (RCWs). Management involved removal of midstories (2.5 to 17.5 cm dbh) and repeated prescribed burns. Results indicate that these management practices appear to have effects similar to that of timber harvests on species composition and abundance of breeding birds. Prescriptions for RCWs were beneficial to several species associated with early successional and edge habitat, including at least some neotropical migrants whose numbers are decreasing in eastern North America (e.g., indigo bunting, hooded warbler, and prairie warbler). Management for RCWs appears to decrease habitat quality for other neotropical migrants, particularly those associated with forest interiors.

Disturbance History of an Old-Growth Forest in Southeastern Ohio

Brian C. McCarthy and Darrin L. Rubino
Department of Environmental and Plant Biology, Ohio University, Athens, OH 45701

We performed a dendroecological investigation of Dysart Woods, an old-growth forest in southeastern Ohio, to determine stand age and the historic disturbance regime. The 23 ha woods is dominated by white oak, beech, and sugar maple. Because Dysart Woods is a preserve, our sample size was limited to ten white oak samples that yielded a 374-yr chronology. Of the samples with pith, it appears that the trees originated under closed canopy conditions. From 1625-1850, ring widths were consistently small and below the overall mean of 1.5 mm; this was followed by a nearly 100-yr period of above-average growth before leveling off in the last 50 years. Using a 10-yr running median technique, we detected 2.42 release events per tree per 100 years. This is consistent with other old-growth studies in the region and characteristic of a gap disturbance regime. With few exceptions, Resettlement fire is *not* recorded in a full basal slab from Dysart Woods. However, burning during the 1800s (period of settlement) was prolific and occurred every few years before diminishing in the early 1900s. While fire may have been an important ecological factor in these woods, it does not appear to have influenced the origin of white oak.

Response of Pin Cherry *(Prunus Pensylvanica* L.F.) to Fire, Canopy Disturbance, and Deer Herbivory on the Westvaco Wildlife and Ecosystem Research Forest

David W. McGill[1], Rachel J. Collins[2], and Walter P. Carson[2]
[1]Research Scientist, Westvaco Corporation, Appalachian Forest Research Center, Rupert, WV 25984,
[2]Doctoral Candidate and Associate Professor, respectively, Department of Biological Sciences, University of Pittsburgh, A234 Langley Hall, Pittsburgh, PA 15260

A study was established to assess the impact of fire, canopy disturbance, and deer herbivory on the germination and development of pin cherry *(Prunus pensylvanica* L.f.) in Appalachian hardwood stands located on the Westvaco Wildlife and Ecosystem Research Forest in Randolf County, West Virginia. In each of four stands, two blocks with four, 400 square meter treatment plots were established. One of four treatments were randomly applied to each treatment plot: 1) fence, gap; 2) fence, no gap; 3) no fence, gap; and 4) no fence, no gap. In each treatment plot, five one-square meter regeneration plots were randomly placed and permanently monumented. All four stands were burned in spring 1999. Over the course of the 1999 growing season, four separate regeneration measurements were made at approximately one-month intervals. Pin cherry germination showed a large amount of variation from stand to stand. Average initial pin cherry densities ranged from 23,500 to 63,000 seedlings per hectare (SPH) following prescribed fire. By the end of the growing season, seedling densities ranged from a low of 5,250 SPH in the No Fence/Gap treatment to 43,250 SPH in the Fence/Gap treatment. These differences were principally a function of deer herbivory. Average height in the Fence/Gap treatment was 12.6 cm greater than the next tallest treatment average—the Fence/No Gap treatment. The tallest pin cherries in the fenced treatments exceeded one meter in height.

Prescribed Fire and Herbicides Modify Soil Processes During Barrens Ecosystem Restoration

C.C. Rhoades, T. G. Barnes, and B.E. Washburn
University of Kentucky, Department of Forestry, Lexington, KY 40546

Prescribed fire has become a common tool of natural area managers for removal of invasive species and maintenance of barrens grassland communities. Certain target species however, resist fire and may require additional removal treatments. An effective option for removal of *Festuca arundinacea* (tall fescue) from areas that previously supported native warm season grasses combines spring prescribed fire and post-emergent herbicides. We studied changes in soil N and C dynamics following prescribed fire and herbicide application in a remnant barrens in westcentral Kentucky. Fire and herbicide effects were compared in a 2 way factorial design with 5 replicates. Fescue cover decreased and bare soil increased from near 88 and 2% in no-burn/noherbicide control plots to 12 and 28% in burned/herbicide plots. During the month following the fires, soil N movement measured with *in situ* exchange resins was 1.4-fold higher in burned compared to unburned soils. Net N mineralization was 2.8-fold higher in burned compared to unburned soils. Burned/herbicide soils mineralized more N than non-herbicide soils. Fire increased soil temperature and decreased soil moisture and CO2 efflux. Removal of non-native plants modify chemical, physical and biologic soil conditions that control availability of plant nutrients and influence plant species performance and community composition

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Experimental Studies of Oak Seed Responses to Fire Temperature in Appalachian Mixed-oak Litter

Cynthia L. Riccardi and Brian C. McCarthy
Department of Environmental and Plant Biology, Ohio University, Athens, OH 45701

The relationship between wildland fire intensity, fire temperature, and the response of plant and animal populations is complex. Few studies have examined the relationship between fire temperature at fine spatial scales and oak seed germination. To examine temperature patterns, we exposed aluminum tags painted with temperature-sensitive paints (38-427°C) and placed them at –1, +2, and +12 cm relative to the soil surface, in 1 x 1 m grids with 20 cm spacing distances, and burned in oak litter. Belowground (<30°C) and 12 cm (range: 150-225°C) sensors were spatially homogeneous across the experimental burn plots. Sensors in the leaf litter experienced the hottest and most variable temperatures (95-375°C) and greatest spatial variability. A subsequent study of 432 *Quercus prinus* seeds varied seed position (above- or below-ground), temperature (control, 38, 66, 150, 204, and 260°C), and exposure time (5 or 15 min). Time and temperature were highly significant ($P<0.001$). Seed exposure to temperatures above 150°C for more than 5 min yielded decreased germination. Temperatures at 204°C for 15 min usually resulted in complete mortality. Caching of *Q. prinus* seeds by gray squirrels may potentially increase seed survival during wildland fires. Further studies are underway to evaluate temperature and its effect on weevil *(Curculio)* populations.

Monitoring Scarab Populations Exposed to Prescribed Fire in Mixed-oak Forests of Southeast Ohio

A.E. Smith and D.J. Horn
Department of Entomology, The Ohio State University, Columbus, OH 43210

Scarabaeidae was selected to determine the effects of surface fire on the distribution and abundance of insect populations. Subfamilies Geotrupinae and Scarabaeinae were expected to show impact of fire, because they exploit resources, such as leaf litter and fungi, that are consumed by fire. Twelve 60-ha. study blocks were selected in 80- 120 year old mixed-oak forest according to similar exposure, soil type, elevation, and vegetation composition. Six blocks each were located in Vinton and Lawrence County, Ohio. Each block was equipped with a transect of twelve, 11-cm dia. unbaited pitfall traps. A pre-burn baseline survey was completed in 1995, and two blocks per site were burned in April 1996; one block was burned annually 1997-1999, and the other was burned in 1999. Three species were selected to model burning effects: *Ateuchus histeroides*, *Onthophagus striatulus*, and *Geotrupes.splendidus*. A multivariate analysis of repeated measures revealed no measurable impact of fire; however, significant yearly variation did exist. Data show a high interplot variation, which suggests that species could be habitat specific. This study yields insight into the diversity and distribution of Scarabaeidae, and whether forest plots can be adequately replicated.

Ground Beetle Responses to Prescribed Burning in Southern Ohio's Hardwood Forests

Chris Stanton, Foster F. Purrington, and David J. Horn
The Ohio State University, Department of Entomology, Columbus, OH 43210

The Forest Service has administered prescribed fire to hardwood forests in southern Ohio since 1996 in order to study the ecological response of the mixed-oak community to prescribed burning. One component of the project involved studying the effects of the fires on forest floor invertebrates. The primary objective of this study was to monitor ground beetle populations for any significant changes that may indicate critical effects of fire on the forest floor community.

Four replicates of mature, mixed-oak forests were subjected to annual spring burns, occasional spring burns, and no burns (control). Ground beetles were collected via pitfall traps established within each treatment block and operated from May to October each year.

Results from five years of monitoring do not indicate any significant effects of prescribed fire on ground beetle abundance, species richness, or diversity. Because the fires did not significantly affect ground beetle populations, prescribed burning should be considered a safe management tool for restoring and maintaining oak forests without devastating the forest floor community.

Yaussy, Daniel A., comp. 2000. **Proceedings: workshop on fire, people, and the central hardwoods landscape;** 2000 March 12-14; Richmond, KY. Gen. Tech. Rep. NE-274. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station. 129 p.

Contains 18 papers and 16 poster abstracts on the history of fire, fire ecology, fire and ecosystem management, and fire and the future presented at the workshop on fire, people, and the central hardwoods landscape.

Keywords: native burning, prescribed fire, prescribed burning, oak, mixed-oak, oak-hickory, barrens, ridgetop-pine, soil microbes, rare plants

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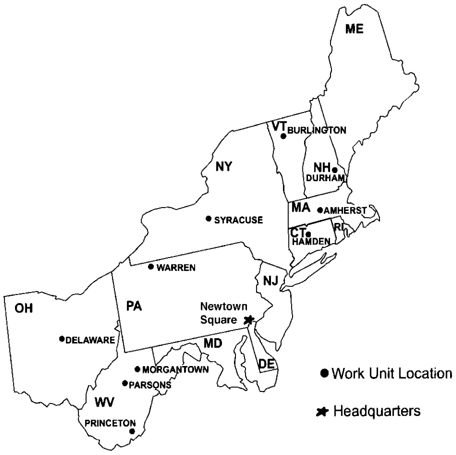

Headquarters of the Northeastern Research Station is in Newtown Square, Pennsylvania. Field laboratories are maintained at:

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